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USDA Forest Service, Intermountain Region

Objection Reviewing Office -SWIRL Project

Room 4403

324 25th Street

Salt Lake City, Utah 84401

RE: OBJECTION AGAINST THE SOUTHWEST IDAHO RESILIENT LANDSCAPE PROJECT

1. Objectors

Lead Objector Sara Johnson, Director, Native Ecosystems Council, PO Box 125, Willow Creek, MT 59760; phone 406-579-3286; sjjohnsonkoa@yahoo.com.

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Steve Kelly, Director, Council on Wildlife and Fish, PO Box 4641, Bozeman, MT 59772; phone 406-920-1381; troutcheeks@gmail.com.

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Kristine Akland, Senior Attorney, Center for Biological Diversity, PO Box 7274, Missoula, MT 59807; phone 406-544-9863; kakland@bioogicaldiversity.org.

Signed this 20th day for Objectors of May

Sara Johnson

Sara Johnson, NEC Director

2. Name and Location of Project being Objected to

Southwest Idaho Resilient Landscape Project on the Boise National Forest.

3. Responsible Official

Brant Peterson, Boise National Forest Supervisor

4. Attachments

We have included one attachment, attachment #1, with this objection; this attachment includes hard copies of various reports and/or publications that are cited in the objection.

5. Connection between Objection and Previous Public Involvement

On September 27, 2023, Objectors provided joint comments on the draft Environmental Assessment (EA) and draft Finding of No Significant Impact (FONSI) for the Southwest Idaho Resilient Landscape Project (hereafter "SWIRL project'). Other than giving a brief summary of the many issues and concerns, and legal violations we addressed for this project, we are incorporating "by reference" all of the above in order to avoid repetition. A major issue and concern first addressed in these 30-day comments was the failure of the agency to define habitat and population impacts to landbirds, including those associated with forests, woodlands, wooded-shrublands, shrublands, and riparian areas, including a host of bird species that have an identified conservation concern. We provided many specific examples of information that needed to be provided as per project impacts on these birds. In the final Decision and NEPA documents, however, this issue was not addressed, and we are carrying it forward into this Objection. Another major concern we raised in these 30-day comments was the failure of the agency to complete wildlife surveys during project planning, so that results would be available to the public. Inventory data and planned mitigation measures for wildlife are key information aspects of agency management of wildlife, and this information is required to be provided to the public as per the National Environmental Policy Act (NEPA). We are carrying this issue forward into this Objection, due to a lack of wildlife surveys in the project area. We also raised a concern about the lack of Forest Plan monitoring on prescribed burning impacts on wildlife. Without this monitoring, the agency lacks any basis for determining impacts of additional burning on wildlife, including a 20-year project. We have carried this issue forward into this Objection. We raised a concern about the likely vast increases in the nonnative annual grass, cheatgrass, that will occur with the prescribed burning. Failure of the agency to identify significant increases in this grass, along with irreversible impacts, including within Idaho Roadless Areas, was a violation of the NEPA. We have carried this issue forward into this Objection. We also raised a concern about the vast acres of habitat alterations planned in roadless lands without any actual habitat plans for any wildlife species. Implementing massive fuels treatment programs without habitat plans for wildlife will clearly have severe environmental impacts to almost all wildlife species, significant impacts that require completion of an Environmental Impact Statement (EIS). Evaluating this project as an EA is a clear violation of the NEPA. Severe impacts on the threatened wolverine were also identified. These issues were not addressed in the final EA and Decision, so they are being carried forward into this Objection. We also raised the issue about false claims of uncharacteristic fire to justify the massive fuels reduction program. We asked the agency to define the criteria to uncharacteristic fires and why they would be detrimental to wildlife. These issues were not addressed by the agency in the final EA and Decision, so we are carrying this issue forward into this Objection. We also raised a concern about how this massive reduction of vegetation on a huge portion of the Boise National Forest would affect ongoing climate change, as well as habitat conditions for wildlife on the Boise National Forest. In the final EA and proposed decision, the agency failed to demonstrate to the public why increasing effects of climate change and adverse impacts to wildlife represent a net public benefit. The agency, however, has continued to claim that fuels reduction programs that exacerbate climate change and degrade wildlife habitat are a net public benefit. This issue has been carried forward into this Objection. We raised a concern about Condition-based Management for the SWIRL project, as it prohibits effective public involvement by the public, including preventing the public from commenting on aspects of project implementation, because this information is not available to them. This issue is carried forward into this Objection. We raised a concern about the agency's false claims that burning wildlife habitat and whitebark pine stands in IRAs is needed to restore their habitat. To date, no actual information has been provided by the agency to demonstrate removing vast amounts of wildlife habitat qualifies as restoration for any wildlife species, and this issue has been carried forward into this Objection.

6. Suggested Remedies that would Resolve the Objection

Due to the extensive violations the proposed Condition-based Management for the SWIRL project will trigger, including violations of the NEPA, the Migratory Bird Treaty Act (MBTA), the Endangered Species Act (ESA), the National Forest Management Act (NFMA), and the Administrative Procedures Act (APA), Objectors believe this project cannot legally move forward. Any fuels management projects planned need to be limited to a project size where sufficient information can be provided to the public on resource management.

7. Statement of Reasons as per Legal Violations that will be Triggered by the SWIRL Project.

A. The proposed SWIRL Project will violate the National Environmental Policy Act (NEPA), the National Forest Management Act (NFMA) and the Migratory Bird Treaty Act (MBTA) by failing to define project impacts on birds.

The Project Wildlife Report identifies the following 60 bird species as present on the Boise National Forest. These include the following:

Management Indicator Species: Pileated Woodpecker, White-headed Woodpecker, and Black-backed Woodpecker.

Sensitive Species: Three-toed Woodpecker, White-headed Woodpecker, Boreal Owl, Flammulated Owl, Great Gray Owl, Mountain Quail, and Peregrine Falcon.

Priority Species in Forest Habitat: White-headed Woodpecker, Dusky Flycatcher, Lewis's Woodpecker, Black-backed Woodpecker, Varied Thrush, Western Tanager, Hammond's Flycatcher, Ruffed Grouse, Flammulated Owl, Northern Goshawk, Williamson's Sapsucker, Brown Creeper, Townsend's Warbler, Olive-sided Flycatcher, Sharp-shinned Hawk, and Blue Grouse.

Priority Species in Sagebrush/Woodland Habitat: Swainson's Hawk, Short-eared Owl, Lark Sparrow, Brewer's Sparrow, Long-billed Curlew, Mountain Quail, Loggerhead Shrike, Prairie Falcon, Sage Thrasher, Black-billed Magpie, Rock Wren, Sage Sparrow, Grasshopper Sparrow, Gray Flycatcher, Ferruginous Hawk, Plumbeous Vireo, Virginia's Warbler, Black-throated Gray Warbler, Greater Sage-Grouse, and Golden Eagle.

Priority Species in Riparian Habitat: Rufous Hummingbird, Black-chinned Hummingbird, Yellow Warbler, Calliope Hummingbird, Willow Flycatcher, MacGillivray's Warbler, and American Dipper.

The following bird species are also likely to occupy wooded shrublands on the Boise National Forest as per Gillihan et al. 2006: Black-chinned Hummingbird, Ferruginous Hawk, Ash-throated Flycatcher, Gray Vireo, Pinyon Jay, Juniper Titmouse, Townsend's Solitaire, Western Bluebird, Virginia's Warbler, Back-throated Gray Warbler, and Scott's Oriole.

The following bird species identified as species of the greatest need for conservation by the Idaho Department of Fish and Game likely occur on the Boise National Forest: Great Gray Owl, Golden Eagle, Olive-sided Flycatcher, Red Crossbill, Lewis's Woodpecker, Short-eared Owl, White-headed Woodpecker, Clark's Nutcracker, Grasshopper Sparrow, and Black Rosy-Finch.

The following bird species that have been identified as Birds of Conservation Concern (BCC) in the Great Basin Region by the U.S. Fish and Wildlife Service likely occur on the Boise National Forest: Calliope Hummingbird, Broad-tailed Hummingbird, Flammulated Owl, Short-eared Owl, Sage Thrasher, Cassin's Finch, Rufous Hummingbird, Northern Harrier, Long-eared Owl, Olive-sided Flycatcher, Back Rosy-finch, Cassia Crossbill, Evening Grosbeak, and Lewis's Woodpecker.

The habitat associations for the above birds that have identified conservation concerns demonstrate that the SWIRL project will have severe adverse impacts on these species due to habitat removal. These adverse impacts will occur to birds that rely on conifer seeds as forage, birds that rely on old growth forests as breeding habitat, birds that rely on snags for nesting habitat, birds that rely on insect infestations as forage, birds that rely on woodlands and sagebrush habitat for viability, and birds that rely on forest fires to provide habitat. Examples of these adverse impacts are provided below.

a. Birds that consume conifer seeds as forage

The following 38 bird species including 12 of conservation concern, that are known to consume conifer seeds and juniper berries (Smith and Balda 1947; Smith and Aldous 1979; Dobkin 1992; Gillihan et al. 2006) likely occur on the Boise National Forest: Cassin's Finch, Blue Grouse, Ruffed Grouse, Red-shafted Flicker, Lewis's Woodpecker, Stellar's Jay, Gray Jay, Clark's Nutcracker, Winter Wren, Robin, English Sparrow, Cowbird, Evening Grosbeak, Purple Finch, Pine Grosbeak, Redpoll, Goldfinch, Red Crossbill, Slate-colored Junco, Oregon Junco, Chipping Sparrow, Song Sparrow, Western Scrub Jay, Black-billed Magpie, Mountain Chickadee, Pinyon Jay, Hairy Woodpecker, White-headed Woodpecker, White-breasted Nuthatch, Red-breasted Nuthatch, Pine Siskin, Juniper Titmouse, Townsend's Solitaire, Mountain Bluebird, Western Bluebird, Bohemian Waxwing, and Cedar Waxwing. With forest thinning, the production of conifer seeds in forest stands will decline, and as such, so will this key forage resource for birds. One especially important conifer tree that will have an undefined level of removal is the limber pine. This tree produces pine seeds that have a very high value to birds (Pfister et al. 1977), such as the Clark's Nutcracker and Pinyon Jay, both species of conservation concern.

b. Birds that nest in cavities

The following 27 birds, including 8 of conservation concern as well as the MIS Pileated Woodpecker, known to nest in cavities within dead trees that likely occur on the Boise National Forest include: Mountain Bluebird, Western Bluebird, Black-backed Woodpecker, Pileated Woodpecker, Three-toed Woodpecker, American Kestrel, Black-capped Chickadee, Boreal Owl, Flammulated Owl, Brown Creeper, Downy Woodpecker, House Finch, House Sparrow, House Wren, Lewis's Woodpecker, Mountain Chickadee, Hairy Woodpecker, Pileated Woodpecker, Northern Pygmy Owl, Red-breasted Nuthatch, White-breasted Nuthatch, Pygmy Nuthatch, Northern Saw-whet Owl, Violet-green Swallow, Tree Swallow, Williamson's Sapsucker, Red-naped Sapsucker, and Vaux's Swift. These bird species require cavities for nesting. Dead trees in stands to be treated in the SWIRL project are considered "fuels," and will be reduced with treatment. Treated stands will differ from natural forest stands, and populations of cavity nesting birds will also be reduced. Natural forest stands in Region 1 of the Forest Service, including in adjacent Montana, have on average 12 snags per acre over 10 inches dbh (Bollenbacher et al. 2008).

c. Birds that feed on insect pests

The following 4 species of forest birds, including 3 of conservation concern and the MIS Pileated Woodpecker, known to feed in insects/arthropods associated with infested green trees, as well as standing and downed dead trees, including larvae of wood-boring beetles (buprestids, cerambycids) engraver beetles, and bark beetles (scolytids), codling moths, pupae of cercropia, and ants, likely occur within forest stands that occur on the Boise National Forest include: Black-backed Woodpecker, Three-toed Woodpecker (Goggans et al. 1987), Pileated Woodpecker (Bull et al. 2005) and the Hairy Woodpecker (*Id.*). The SWIRL project has a treatment objective to reduce insect and disease infestations, which will also reduce populations of these 4 woodpeckers.

d. Birds that depend upon old growth forests

The following 24 species of birds, including 11 species of conservation concern, that likely occur on the Boise National Forest in old growth forests include the Boreal Owl, Great Gray Owl, Brown Creeper, Chestnut-backed Chickadee, Flammulated Owl, Golden-crowned Kinglet, Hairy Woodpecker, Hammond's

Flycatcher, Williamson's Sapsucker, Hermit Thrush, Lewis's Woodpecker, Northern Goshawk, Pileated Woodpecker, Black-backed Woodpecker, Pine Grosbeak, Pygmy Nuthatch, Red-breasted Nuthatch, Swainson's Thrush, Three-toed Woodpecker, Townsend's Warbler, Varied Thrush, White-breasted Nuthatch, Vaux's Swift, and Winter Wren (USDA 2018). According to vague descriptions of proposed thinning treatments for the SWIRL project, trees up to 15 inches dbh will be cut down. Understory trees and shrubs will also be burned. These treatments will have significant impacts on old growth forests, and as well, have significant adverse impacts on associated species. The Boise National Forest has no protection requirements for old growth forests as defined by Hamilton (1993); old growth forests can be thinned and burned as long as some larger, older trees are retained. The impact of treatments in old growth forests has never been evaluated in the Boise Forest Plan Final Environmental Impact Statement (FEIS). The severe impacts old growth treatments on wildlife are thus not being addressed by this forest, in violation of the NEPA, the NFMA, and the Migratory Bird Treaty Act (MBTA).

e. Birds that depend upon shrublands and wooded-shrublands as habitat.

There are a host of birds that likely occur on the Boise National Forest that use wooded-shrublands and shrublands as essential habitat, and almost all of them have conservation concerns. These include the Loggerhead Shrike, Brewer's Sparrow, Sage Thrasher, Sage Sparrow, Ferruginous Hawk, Greater Sage-Grouse, Virginia's Warbler, Gray Flycatcher, Golden Eagle, Pinyon Jay, Townsend's Solitaire, Western Bluebird, Mountain Bluebird, Black-throated Gray Warbler, Scott's Oriole, Ash-throated Flycatcher, Black-chinned Hummingbird, Juniper Titmouse and Northern Goshawk. Slashing and burning wooded-shrublands and shrublands for the SWIRL Project will have severe adverse impacts on all these species. For example, the Loggerhead Shrike has suffered severe population declines (Audubon 2007), and is highly associated with sagebrush habitats (Cade and Woods 1996). Both the Brewer's Sparrow and Sage Thrasher are highly dependent upon large blocks of sagebrush habitat (Montana Partners in Flight 2000). Juvenile Northern Goshawks are highly dependent upon woodlands-shrublands during the fall when they disperse from parental territories (Wiens et al. 2006). The Golden Eagle as well as the Ferruginous Hawk are both highly dependent upon cottontails and jackrabbits that in turn use sagebrush habitats (Kochert et al. 1999; Knick and Dyer 1997; Smith et al. 1981). There are also a number of bird species that consume sagebrush as forage, such as the Greater Sage-Grouse, Dark-eyed Junco, Horned Lark, and White-crowned Sparrow, as well as mammals (jackrabbits, cottontails, ground squirrels) that eat sagebrush plants that are also important prey species for raptors (Welch 1999). Also, sagebrush plants can provide vast quantities of seeds that are eaten by birds, with up to 50 millions seeds per hectare produced some years (Owens and Norton 1992).

f. Birds that depend upon forest fires

The following species of forest birds are known to benefit from various levels of forest fires, from low to stand replacement burns (Hutto 1995; Hutto and Patterson 2016): stand-replacement burned areas used immediately by the Black-backed Woodpecker, Hairy Woodpecker, Three-toed Woodpecker, and Northern Flicker to feed on bark and wood-boring beetle populations, and species as the Cassin's Finch, Clark's Nutcracker, Red Crossbill and Pine Siskin come in to feed on conifer seed availability from cones that opened from the fire; also quick responders include cavity-nesting species as the Western and Mountain Bluebirds, birds using open areas for aerial foraging on insects, as the Western Wood Peewee, and birds that use the relatively predator-free nest sites at ground level for nesting from burned-out roots and root wads associated from wind-thrown trees, as the Rock Wren, Townsend's Solitaire, Dark-eyed Junco; also, the Olive-sided Flycatcher uses edges between burned and unburned forests for nesting and aerial foraging; several years following a stand-replacement fire, additional cavity nesting birds move ion, as the House Wren, as well as the Williamson's Sapsucker, who nests along the edges of burned/unburned forests; other birds that move in several years after a stand replacement fire include the White-breasted Nuthatch, who feeds on insects under sloughing bark from dead trees; in areas that burned at lower severity, "mixed responders" as the Brown-headed Cowbird, Red Crossbill, Red-breasted Nuthatch, Western Tanager, and

Ruby-crowned Kinglet respond positively to a forest habitat that is more suitable (dead trees) than a long-unburned green forest; finally, longer after severe fire has occurred and dense stands of conifer regeneration develop, bird species as the Orange-crowned Warbler, MacGillivray's Warbler, Calliope Humming Bird, and Lazuli Bunting use burned forests. Hutto and Patterson (2016) noted that other research has shown that species as the Ruby-crowned Kinglet, Yellow-rumped Warbler, and Hermit Thrush have been found to reach highest densities about 100 years post-fire, again demonstrating a dependence upon fire for providing suitable habitat. One objective of the SWIRL project is to reduce stand replacement fires. This objective is in direct conflict with the benefits of forest fires to many bird species.

The above tally of birds that likely occur on the Boise National Forest include approximately 60 that have a conservation concern. Although the agency claims that the SWIRL project will benefit all bird species in the long term, there is no supporting documentation as to how habitats will be improved for these bird species. There are no habitat objectives identified for any of these 60 bird species. Without habitat objectives, the agency has no basis for measuring impacts of vegetation treatments. The agency has not taken a "hard look" at how burning will impact these 60 species, in violation of the NEPA as well as the MBTA. A hard look at project impacts on 60 birds of conservation concern needs to include at least the following:

1. What is the expected change in carrying capacity for the birds that consume conifer seeds and juniper berries in forest and woodland treated areas, and why aren't changes expected to be significant?
2. What is the expected change in carrying capacity for forest and riparian birds due to a reduction of canopy and ground-level hiding cover for nesting and post-fledging survival of young birds, and why aren't changes expected to be significant?
3. What is the expected change in carrying capacity for forest and riparian birds due to a reduction of thermal cover that protects birds, nests, and

fledged young bird from severe weather events and summer heat, and why aren't changes expected to be significant?

4. What is the expected change in carrying capacity for forest, riparian, woodland, and shrubland birds due to the increase of summer temperatures that will result from vegetation thinning of these habitats, and why aren't changes expected to be significant?
5. What is the expected change in carrying capacity of forest, woodland, shrubland and riparian birds due to increases in the brood parasite, the brown-headed cowbird, due to vegetation thinning and fragmentation resulting from vegetation treatments, and why aren't changes expected to be significant?
6. What is the expected change in carrying capacity for forest birds due to reductions of forest-associated insects associated with green and dead trees, and logs, and why aren't changes expected to be significant?
7. What is the expected change in carrying capacity for forest birds due to a reduction in larger snags (at least 10 inches dbh) and recruitment of snags that create suitable nest sites, and why aren't changes expected to be significant?

Until the Boise National Forest completes the required "hard look" at slashing/burning impacts on 60 birds of conservation concern, the agency cannot claim that no significant impacts will be triggered on these species. This conclusion requires an actual analysis, which has never been done for the SWIRL Project.

B. The agency will violate the NEPA and the MBTA by failing to identify and evaluate the cumulative adverse impacts to birds from the proposed project, including mortality and habitat loss, or to propose measures that would reduce adverse impacts.

1. The agency did not estimate the number of birds that would be killed per acre with treatment activities, including slashing and burning trees and shrublands. There were not estimates made, for example, of how many bird nests and newly fledged birds would be destroyed with burning, both from the fire itself, or from the smoke created by the fire. Smoke has been demonstrated to be highly toxic to birds (Defiance Canyon Raptor Rescue 2022). Smoke has killed Bald Eaglets who were not able to escape from heavy smoke. This could easily happen as well to other forest raptors that will be nesting in treatment areas, including Boreal Owls, Flammulated Owls, and Great Gray Owls, all sensitive species on the Boise National Forest. Shrubland birds that are species of conservation concern, as Sage Sparrows, Sage Thrashers, Brewer's Sparrows, and Loggerhead Shrikes that nest in the spring, will also have their nests destroyed as well as their fitness degraded by toxic smoke. The agency "suggested" that bird nests would be protected during burning activities, but did not provide any specifics as to how this would be done, in violation of the NEPA. If the agency has taken a "hard look" at bird mortality that will be triggered from slashing and burning in the spring nesting season, they would have proposed an action alternative that required no spring burning. There was no such alternative proposed. Since slashing/burning is a planned activity, the associated unmeasured bird mortality is also a preventable activity - avoid burning in the spring. The agency is required by the MOU for the MBTA to measure the planned mortality of birds per acre of proposed treatments, and obtain a "take permit" from the U.S. Fish and Wildlife Service for this planned avoidable mortality.
2. The agency is required by the NEPA and the MBTA to evaluate cumulative impacts to landbirds, including planned mortality due to prescribed fire and smoke impacts, as well as the habitat loss that will occur due to burning. The total reduction in landbird populations from these cumulative impacts is required to be evaluated by the NEPA, along with assessing how these cumulative impacts will affect ongoing trends of landbirds. With many declining population trends (e.g., 64% of 67 species of western forest birds are in decline, and 56.5% of 62 species of aridland birds are in decline) are documented (Rosenberg et al. 2019). The North American Bird Conservation Initiative (2022) also reported on the loss of 1 in 4 landbirds

since the 1970s, which comes to a loss of 3 billion birds. This report noted that public aridlands are essential for conservation of these birds, including species as the Sage Thrasher that has exhibited accelerated declines since 2010. This report noted that about half of 46 species of western forest birds are in decline, including tipping point species as the Pinyon Jay and Rufous Hummingbird (species that have lost 50% of their populations in the last 50 years, and are expected to lose another 50% of their populations in the next 50 years); another recently declining western forest bird was identified as the Williamson's Sapsucker. Another tipping point species identified in this report is the Evening Grosbeak, while the Olive-sided Flycatcher was noted to have lost 50% of its population in the last 50 years.

C. The agency is violating the NEPA and the NFMA by failing to provide the public the wildlife survey results and mitigation measures to be implemented to protect nest sites of sensitive forest raptors.

The SWIRL project NEPA documents did not provide the results of any wildlife surveys for the project area. Instead, the agency stated that surveys and mitigation measures for sensitive forest raptors, as the Great Gray Owl, Flammulated Owl, and Boreal Owl will be done "post decision." As such, the public will not be able to see if valid and thorough surveys were actually done, or raise objections to invalid surveys and mitigation measures. Wildlife survey results and mitigation measure designs are key information to the public in order for them to understand agency commitments to management of wildlife. With wildlife surveys results being hidden from the public, the public has no means to understand specifically how sensitive forest raptors are being managed, if at all, in treatment areas. The agency claims that surveys will be done post decision does not provide the public with essential information on the management of wildlife in treatment areas. As well, the public is not able to see, or to comment on, what specific mitigation measures have been implemented in treatment areas to protect sensitive forest raptor nests, because these mitigation measures are never defined or demonstrated as per implementation in a project area. It is clear that one reason the agency is using condition-based management for the SWIRL project is that wildlife surveys and mitigation measures do not have to be provided to the public. It seems highly likely that wildlife surveys on 77,000 acres of planned treatment areas per year will not be done on many acres, since past burning projects have been only about 3,500 acres. The public is expected to believe that the agency will do high-quality, valid wildlife surveys on 22 times the acres per year previously burned. The NEPA documents for the SWIRL project do not include any notations of the huge increase in survey technicians that would be required to survey 77,000 acres of sensitive forest raptor sites per year. It is clear that the agency cannot demonstrate that key management requirements for wildlife (surveys and effective mitigation measures) will actually be done because these requirements have not been done before a decision is made, in violation of the NEPA.

We also note that the Boise Forest Plan requires that management actions (mitigation measures and location of sensitive wildlife species) regarding 3 standards for wildlife (e.g., WIST03, WIST04, WIST05) be determined "during project planning." The agency's decision to do wildlife surveys and develop mitigation measures post-project implementation are a violation of the Forest Plan.

D. The agency will violate the NFMA by failing to apply Forest Plan monitoring results to the SWIRL Project, and the NEPA by implementing a decision beyond the 5-year timeliness requirement of the NEPA.

The SWIRL project is stated to last for 20-25 years. Forest Plan monitoring results are required at yearly to 5-year intervals. The agency did not define in the SWIRL NEPA documents how upcoming Forest Plan monitoring results will be applied to project implementation post-decision. For example, the population monitoring of MIS, such as the Pileated Woodpecker and White-headed Woodpecker, including how prescribed burning is impacting their populations and habitat use, are not considered a requirement for implementing a 20-25 year project. It is not clear how forest plan monitoring requirements can be applied to this project or the project decision. If notable adverse impacts are documented for MIS or sensitive species, it is unclear how the public would be able to request changes in the project design post decision. As well, if new science is published that identifies severe impacts of prescribed burning on specific wildlife species, there is no procedure available to the public post-decision to affect ongoing project implementation. Condition-based management in effect shields the agency from any additional public involvement for 20 or more years, in violation of the NEPA.

E. The SWIRL Project will violate the NMFA and the NEPA regarding burning of Greater Sage-Grouse habitat.

The Boise Forest Plan has incorporated management direction for the Greater Sage-Grouse (hereafter "sage grouse") as per the 2015 Greater Sage-Grouse Record of Decision for Idaho, Southwest Montana, Nevada, Utah and Wyoming. Sage grouse habitat on the Boise National Forest occurs in the SWIRL project area, in the Snake Salmon and Beaverhead Priority Area 23, as well as in the Biologically Significant Unit (BSU) Idaho West Owyhee Conservation Area – Important. The SWIRL project area contains 57,334 acres of general sage grouse habitat (GHMT), and 21,122 acres of important sage grouse habitat (IHMA) as per the Wildlife Report at 31. These 78,456 acres of sage grouse habitat are approximately half of the total 159,443 acres on the Boise National Forest. There are 6 sage grouse leks within the Forest Service boundary, while the number of leks adjacent to the Forest Service within 3.1 miles is not provided. The Wildlife Report states that leks will be protected by a 3.l mile buffer from burning. No leks are mapped for the project area. No buffer areas are mapped for the project area. The public cannot determine if the 3.1 mile buffer from leks will be included in project implementation.

The Forest Plan Desired Condition (DC) for sage grouse identified in the ROD (2015) at Table 1 directs that at least 80% of nesting habitat having 15-25% canopy cover of sagebrush with plants 12-31 inches tall; at least 40% of late brood-rearing/summer habitat having 10-25% canopy cover, with plants from 16-32 inches in height; and with at least 80% of the winter habitat having at least a canopy cover of 10% and at least 10 inches in height. IN addition, the ROD includes standards for management of sage grouse habitat, including:

GRSG-FM-ST-042: do not use prescribed fire in 12-inch or less precipitation zones unless necessary to facilitate restoration of sage grouse habitat conditions consistent with desired conditions.

GRSG-FM-ST-043: restoration to desired conditions requires identification in NEPA document as to how burning moves habitat towards desired conditions.

GRSG-FM-GL-044: sagebrush removal including burning should be restricted unless it supports attainment of desired conditions or protects from wildfire.

GRSG-FM-GL-047: fuels treatments should be designed to maintain, restore or enhance sage grouse habitat.

The NEPA documents for the SWIRL Project do not demonstrate how the Forest Plan direction will be met by moving towards desired conditions identified in Table 1 of the ROD for sage grouse. As such, the agency cannot demonstrate to the public that this forest plan direction will be implemented for the SWIRL Project, in violation of the National Forest Management Act (NFMA).

The NEPA documents for the SWIRL project also fail to define how many acres of sage grouse habitat will be burned, or where these burning units occur. The Wildlife Report also provides false information on management of sage grouse habitat. It is claimed that "mosaic burning" of sagebrush will maintain sage grouse habitat, and possibly improve it. However, the current best science has defined the persistence of sage grouse as strongly related to the abundance and patch size (large) of "tall sagebrush" (Wisdom et al. 2011; Johnson et al. 2011). Creating a diversity of sagebrush age classes has been the long-standing justification for Forest Service sagebrush burning projects. This long-standing practice has certainly been an important contributing factor in the current status of this species. The North American Bird Conservation Initiative (2022) reported the sage grouse as a "tipping point species," species that have lost 50% of their populations in the last 50 years, and are on a trajectory to lose another 50% of their populations in the next 50 years. This report notes that for tipping point species, immediate science and conservation actions are needed to turn around declines. Somehow, more burning of sagebrush in the SWIRL project hardly addresses this critical conservation need. These agency claims that burning will improve sage grouse habitat are a violation of the NEPA.

F. The agency is violating the NEPA by failing to map and quantify areas dominated by cheatgrass in the SWIRL project area, to demonstrate that mitigation measures are working, or to identify the expected increase in fires due to cheatgrass promotion in treatment units.

There will be irreversible impacts on the Boise National Forest due to the vast prescribed burning that is planned in the SWIRL project area. Cheatgrass is the most widespread invasive species in North America with millions of acres converted to a monoculture of cheatgrass and tens of millions of acres at risk of infestation; a significant proportion of the public lands at risk from cheatgrass-fueled fire is managed by the Forest Service; multiple scientific studies demonstrate that cheatgrass invasion create larger and more frequent fires by creating continuity of fine fuels; the threshold for avoiding the ecological and economic consequences of cheatgrass infestation are at between 5% and 25% of the land area; multiple studies identify prevention of ground disturbance as the best way to limit the spread of cheatgrass; cheatgrass expands rapidly because it is a prolific seed producer, can germinate in spring and autumn giving it a competitive advantage over native grasses, is tolerant of grazing, and increases with fires; cheatgrass ultimately drains soils of available nitrogen, which helps cheatgrass to exclude native grasses and exhausts other soil nutrients needed by native plants; cheatgrass depletes soil water in the spring hindering survival of native seedling and subjecting adult native plants to moisture stress; for a litany of reasons, minimizing cheatgrass infestations and restoring infested lands to natural conditions should be a priority dictating the outcomes of land-use and land management decisions throughout the arid West; recent record-breaking grass fires in Texas, Hawaii an d Colorado reinforce conclusions regarding wildfire hazards created by cheatgrass (Forest News 2024).

The proposed SWIRL project will clearly create irreversible increases in cheatgrass, which not only displaces native plants, creates hazards for raptors (McCrary and Bloom 1984), and increases fire frequency (USDI 2015). The severe long-term impacts that the SWIRL project will have on infestations of cheatgrass across the Forest were never identified as significant, in violation of the NEPA. For example, the agency notes that existing areas of cheatgrass infestations will be "treated" before the area is burned. To date, the agency has not provided any evidence that cheatgrass infestations can be effectively removed, which seems apparent as the SWIRL project area is covered with an unknown acreage of cheatgrass infestations. Since cheatgrass appears to be extremely difficult to eradicate, it seems apparent that burning over a million acres of the Boise National Forest will create vast new infestations of cheatgrass that will never be eradicated. This significant irreversible impact was never evaluated in the SWIRL project NEPA documents. What is the expected acreage increase in cheatgrass, how does this compare to current levels, and what level of cheatgrass increases considered nonsignificant? Also, the agency needs to estimate how these increased cheatgrass infestations will affect fire frequency across the Boise National Forest. It is unclear why increasing cheatgrass is a valid fuels management strategy, and needs to be fully demonstrated to the public.

G. The agency is violating the NEPA by claiming that slashing and burning in whitebark pine stands represents "restoration" of this threatened species; the agency is violating the ESA by exceeding mortality limits set by the 4(d) rule.

Whitebark pine occurs in the Potential Vegetation Group 11; the Project EA at 20 notes there are more than 95,283 acres of whitebark pine across the Boise National Forest; potential negative effects of burning in whitebark pine area is noted to include damage or mortality to individuals, populations and seed banks burning pre-treatment activities, burn preparation, and prescribed fire application. The EA at 35 notes that large blocks of forest in PVG 11 may be burned, where prescribed fire activities could be integral to restoration of whitebark pine, which is dependent upon fire to reduce competition anj promote regeneration. This PVG 11, high elevation habitat, includes 25,995 acres of forest as per Table T-1 of the EA.

The agency provided no monitoring data of past whitebark pine treatments on the Boise National Forest to show burning whitebark pine has increased regeneration. This is highly unlikely, as Keane and Parson (2010) reported that logging and burning treatments had not resulted in hardly any whitebark pine regeneration in 40 years. Also notable is the U. S. Fish and Wildlife Service's 2023 standing analysis on whitebark pine, where it is questioned that whitebark pine depends upon fire for persistence. This report also notes that whitebark pine seedlings and saplings can tolerate shade conditions of forest for many years and then released when canopies open up with disturbances, such as pine beetles. The impact of planned burning activities on whitebark pine has recently been estimated on a National Forest in Region 1 of the Forest Service. The Shoshoni National Forest estimated that burning in whitebark pine stands and mixed conifer stands could result in the total estimated mortality of 1,310,082 whitebark pine (USDA 2024). This hardly qualifies as "restoration." Six et al. (2014) also noted that forest thinning and increased tree growth of remaining trees may increase susceptibility to mountain pine beetle attack in the future, due to increased growth. Overall, the Boise National Forest has failed to identify that burning and slashing treatments in whitebark pine stands will result in a massive loss of seedlings, saplings, and younger, smaller whitebark pine trees, and along with this loss, a massive loss of accrued genetic diversity over years of recruitment. The planned adverse impacts on whitebark pine will also impact various Idaho bird species of conservation concern, as the value of whitebark pine seeds to wildlife is very high, similar to that of limber pine.

The planned loss of up to millions of whitebark pine recruitment trees in the SWIRL project is also a violation of the ESA, as the USFWS 4(d) rule limits the number of whitebark pine trees that can be killed in agency management activities.

H. The agency will trigger multiple violations of the Idaho Roadless Rule.

The agency did not map the 5 different categories of Idaho Roadless Lands in the SWIRL project area. The agency did not define the specific management direction for each of these 5 categories of roadless lands, or what treatments would be consistent with each category. The agency did not map the Wildland Urban Interface of Community Protection Zones in any of these Idaho Roadless Lands. The implementation of the Idaho Roadless Rule was never defined, in violation of the NEPA.

The agency repeatedly claims that the proposed treatments in IRA lands will eventually benefit wildlife with habitat improvements. AS we noted previously in this objection, there are roughly 60 bird species with identified conservation concerns that do not have a single habitat objective identified for IRA treatments. Without a demonstration as to how treatments will benefit habitats for these 60 species of conservation concern, the agency is violating the NEPA by failing to support claims with documentation to the public. As we have noted, there is a highly likely outcome that habitat for all 60 bird species of conservation concern will be degraded, with associated declines of these species on the Boise National Forest. To claim that undisclosed population/habitat declines represents restoration in IRA is not only a NEPA violation, but also one of the MBTA.

The agency also repeatedly falsely claims that treatments in the IRAs will restore ecosystems. However, no further information was ever provided. It is not defined how slashing and burning will restore forests, wooded-shrublands, shrublands, riparian areas for wildlife.

The actual treatments that are planned within forested, woodlands, wooded-shrublands, shrublands and riparian areas is never described in any detail. Current and planned conditions for shrubs and trees is never defined. Current and planned basal area of trees and shrubs is unknown. Especially concerning is that some of the vegetation reports includes many, many photos of logging treatments, including some followed by prescribed burning. The agency actually never defines how many trees under 15 inches are expected to be cut down in each of the 11 vegetation types. Since this is not a commercial logging proposal, it is unclear why extensive cutting of trees is expected as per the Vegetation Condition-based Management Guide (e.g., Tables 1-15). Similar lack of any description of project treatments occurred for woodland, wooded-shrubland treatments, and shrubland treatments. The public has no idea as to what is actually planned for these vegetation types, in violation of the NEPA. This complete lack of information regarding what is planned for the 11 vegetation types also means that the public has never been able to provide input on these treatments, since we don't know what they entail. As one concern, it was noted that limber pine, a very important tree for wildlife, will not be protected in treatment areas. It is unknown how many acres of this high-value tree will be impacted as a result.

A similar lack of information deals with road management. There are no maps provided for roads that are currently closed in IRAs that will be opened, and likely "reconstructed," to allow heavy equipment access into IRAs. Reconstruction of roads in some of the Idaho Roadless Areas is not allowed.

Justification of heavy management activities in IRAs is based on an agency claim that vegetation across these millions of acres are "unnatural," or as defined by the agency, "departed" from historical conditions. The basis for this claim is never provided, for any of the 11 vegetation types on the Boise National Forest. It is a NEPA violation for the agency to claim that existing vegetation on the Forest is unnatural without supporting documentation, including unnatural for wildlife. This claim would require the agency to provide extensive documentation of the existing conditions of habitat types in each of the 11 vegetation groups. This information could easily be developed by comparing the basal areas of various habitat types in each vegetation group with basal areas defined for these vegetation types by established habitat type documents. In Montana, this is Pfister et al. (1977), that defines habitat types for Montana forests. For PSME/JUCO (Douglas-fir/juniper habitat types), the natural basal area is 139 square feet per acre. For PIFL/JUCO (limber pine/juniper), the natural basal area is approximately 139 square feet per acre. For ponderosa pine habitat types, the natural basal area ranges from 96-168 square feet per acre, and averages 130 square feet per acre. For Douglas-fir forest types, the natural basal area in 13 types ranges from 128-210 square feet per acre, and averages 180 square feet per acre. For 2 whitebark pine/subalpine fir habitat types, the basal area ranges from 199-247. For 5 spruce habitat types, the natural basal area ranges from 177-234, and averages 207 square feet per acre. For subalpine fir habitat types, 15 habitat types range from 175-268 square feet per acre, and average 230 square feet per acre. For lodgepole pine, 5 habitat types range from 130-193 square feet per acre, and average 166 square feet per acre.

The above habitat densities, defined as basal area, will not change over time, as these are the actual site potentials for each habitat type. Although stand densities will be reduced with fire, they will recover quite quickly as trees regenerate and grow. The composition of small and large trees per habitat type will change over time, but in general the total area covered by all trees will remain relatively constant over time. The agency claim that forests and woodlands across the Boise National Forest are unnatural is clearly false, in violation of the NEPA.

The false claim by the agency that habitats across the Boise National Forest are unnaturally dense is being used to justify fuels treatments to prevent uncharacteristic fire. Again, the agency did not provide any actual data as to current versus historic levels of vegetation in the 11 vegetation types. This would also include sagebrush and wooded-shrublands. There was no information provided to demonstrate that these habitats are unnaturally dense and require burning for restoration.

It was noted in agency NEPA reports for the SWIRL project that forest thinning will increase tree growth by reducing the density of competing trees; this will improve stand resilience. **<u>Are these treated forest stands in IRAs being treated for future timber harvest</u>**? This potential long-range plan was never disclosed to the public. It is unclear as to why increasing growth of trees would otherwise be an objective for treatment.

The agency did not provide any actual monitoring results, as photos, to show how past activities in IRAs look. The claim that where required, that the natural appearance of IRAs will be maintained with treatments has in fact never been demonstrate to the public. The claim that the public will not notice that various restoration activities have not occurred seems disingenuous. Some of the activities, besides cutting an undetermined number of large trees per acre in treatment units (up to 15 inches dbh), the following activities may occur:

-treatments may occur at any time of the year.

-there will be cross country travel with heavy equipment.

-activities will include machine piling, slashing of trees up to 15 inches dbh or larger, mastication of trees and woody debris, hazard tree removal, tree pruning, including whitebark pine, the use of chainsaws, chippers, skid steers, and feller bunches to achieve thinning. Tree stumps will be evident past treatment.

-fire lines 18-24 inches will scrape down to bare mineral soil.

-dozer lines may be 12 feet wide to allow for access by tracked heavy equipment including excavators or dozers.

-mulching involves the use of heavy equipment to grind, shred, chip or chop large woody material.

-avoidance of felling trees in stands that contain whitebark pine will occur.

-the overland use of ATVs up to heavy equipment, including dozer lines up to 12 feet wide, will leave visible signs of motorized activity, including likely soil erosion and cheatgrass infestations.

-mitigation measures for this heavy equipment include minimal turning or attempts to do "rolling turns" to reduce soil displacement.

-heavy equipment will strive to operate over masticated fuels to reduce soil compaction.

-track equipment operators will avoid making abundant sharp right angle turns; instead they will use gentle curved patterns with the lowest possible number of sharp angle turns.

Somehow, the agency is claiming that all this management activity within IRAs will not be evident to the general public, and that these protected areas will maintain an unmanaged appearance. The agency did not provide any actual studies that measure the level of landscape alteration that is tolerable to the general public. Without supporting documentation that roadless lands will continue to appear natural and undisturbed by the general public, claims of such are a NEPA and roadless area rule violation.

Attachment #1 for the Objection filed by NEC et al. on May 20, 2024 against the Southwest Idaho Resilient Landscape Project on the Boise National Forest.

Attachment #1 contains relevant portions of the following documents and/or publications cited in the Objection, including:

Audubon. 2007. #8 common bird in decline Loggerhead Shrike (*Lanius ludovicianus*). Birds and Science. State of the Birds – common birds in decline – Loggerhead Shrike. 10/5/2007.

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Loggerhead Shrike males may impale multiple prey items and adorn them with bird bills and feathers to attract a mate.

#8 Common Bird in Decline
Loggerhead Shrike
(Lanius ludovicianus)

French Name: *Pie-grièche migratrice*
Spanish Name: *Alcaudón verdugo*

Genus: *Lanius*
Species: *L. ludovicianus*
Order: *Passeriformes*
Family: *Laniidae*

Bird Image: Gary Stolz, FWS

Range Map

Rate of Decline: 71 percent in 40 years

Global Population: 2.9 million

Continental Population: 2.9 million now, 10 million 40 years ago

Watch List Status:

Appearance: A robin-sized gray bird with black wings, white wing-patches, a black m
black tail. A close look reveals a hooked beak.

Vocalization: Harsh *"bzeek, bzeek"* alarm call. Song is a very quiet combination of sh
clear notes, and harsh notes. Listen (*© Lang Elliot, Nature Sound Studio*).

Habitat: Short grass with isolated trees or shrubs, especially pastureland.

Range: Found year-round in most of Mexico and the southern half of the United State
breeding season only in eastern Washington and Oregon, the northern Great Plains, a
Midwest. No longer found in New England and disappearing from the Mid-Atlantic stat
northern portions of the Midwest.

Feeding: An opportunistic forager that consumes arthropods, amphibians, small repti
mammals, birds, and even roadkill and carrion. Often forages in recently plowed fields
lacks heavy talons with which to capture and hold larger prey, the Loggerhead Shrike
from a perch, attacks from behind, and impales prey on thorns or barbed-wire fences.
poisonous prey, including monarch butterflies and eastern narrow-mouthed toads, for
to allow the poison to break down.

Reproduction: Available cover is the most important criterion for nest site selection,
with thorns are preferred. The nest is usually well hidden and located on top of an exi
Mean clutch size is 5.4 eggs, and birds located at higher latitudes and farther west ter

I-2

Region One
Vegetation Classification, Mapping, Inventory and Analysis Report

$\bar{x} = \frac{\sum x}{n}$

Report 08-07 v2.0

12/19/2008

Estimates of Snag Densities For Eastside Forests in the Northern Region

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1.0 Introduction

In 2000, the Northern Region Snag Management Protocol provided optional snag retention standards which were based on using FIA data from western Montana forests. However, the Protocol specifically recognized that FIA data from northern Idaho and eastern Montana was not used in the Protocol, as it was not available at the time. FIA data is now available and the data for the eastside Forests in this paper provides the most current snag data available. Table 1 shows snags per acre across the entire land base between the three geographic areas of the Region. There is a statistically significant difference in the density of snags and large-live trees between these areas due to biophysical and climatic differences between the areas. This suggests that snag analysis and management plans pertaining to snags should be formulated by geographic area and not extrapolated from one area to another. Furthermore, the 2000 Protocol specifically provided that when local data are available or are considered better than the sources used in the Protocol, Forests have the option to use those data sets. This report provides a replacement for the Northern Region Snag Protocol for eastside Montana forests in Region 1. The snag information provided in this paper does not set forth mandatory or required direction but rather provides current snag information and analysis for consideration by the Forests.

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snags 10.0" DBH and larger, and all snags 20.0" DBH and larger are included in the estimate of snags 15.0" DBH and larger.

The total primary sampling units (PSUs) are the number of FIA plots within the domain of interest, such as wilderness/roadless or with a specified dominance group. The number of forested PSUs are the number of FIA grid locations that have at least a portion of the PSU with a "forested" condition. The information from the "forested" portion of the PSUs are used in the analysis.

3.0 Preliminary analysis of snag densities on Eastside Forests

We evaluated snag densities on the eastside Forests of Region 1 using a hierarchical approach.

3.1 Comparison of Snag Density within and outside of Wilderness and Roadless Areas

First, we looked at the density of snags within and outside of wilderness and roadless areas. Timber harvest and human access can have substantial effects on snag density and longevity (Wisdom and Bate, 2008; Russell et al. 2006). Exploring the density of snags in wilderness and roadless areas can provide insight to natural snag abundance and distribution on a Forest. These can be compared to paired field plots outside wilderness/roadless to help to understand differences between areas that have been influenced by management and unmanaged areas. There is some uncertainty how climate, a period of cool and moderate precipitation, and fire suppression from 1930-1985 has affected snag density and distribution in wilderness and roadless areas. Harris (1999) notes similar uncertainty concerning effects of fire suppression on creation of snags in unharvested areas of western Montana. However, even with some degree of uncertainty it is the best quantitative data available to represent natural forested systems. To date, there has been no known extirpation of cavity nesting species from eastside Forests, within or outside of roaded areas. It follows that, in general, analysis of the roadless portion of these Forests will represent an appropriate range of snag numbers and distribution to develop desired snag conditions for planning purposes.

As shown by Table 2, there are fewer snags in each of the diameter classes outside of wilderness and roadless areas for the eastside, in general, and for all of the Forests except the Helena. Furthermore, the larger the snag, the less common it is. This is largely due to less trees living to an older age, as trees age, they grow slower, never reaching very-large diameters, and the inability of systems to contain large old trees and snags due to various types of disturbance agents which kill and remove them over time.

Table 2: Mean snag densities per acre with 90% confidence interval, by diameter classes, inside and outside of wilderness/roadless areas for all eastside Forests and for each Forest.

	Area	Snags per Acre 10"+			Snags per Acre 15"+			Snags per Acre 20"+			Total Number PSUs	Number Forested PSUs
		Mean	90% CI-Lower Bound	90% CI-Upper Bound	Mean	90% CI-Lower Bound	90% CI-Upper Bound	Mean	90% CI-Lower Bound	90% CI-Upper Bound		
In Wilderness / Roadless	Eastside Forests	12.2	10.8	13.7	2.9	2.5	3.3	0.7	0.5	0.8	937	747
	Beaverhead Deerlodge	10.6	8.5	13.0	2.6	1.9	3.3	0.5	0.3	0.8	325	253
	Custer	12.7	7.0	19.4	3.8	1.8	6.2	1.0	0.4	1.6	84	44
	Gallatin	17.6	14.4	21.1	4.7	3.7	5.9	1.3	0.9	1.8	221	170
	Helena	10.1	5.9	15.0	0.9	0.3	1.6	0.2	0.0	0.4	85	79
	Lewis & Clark	10.3	7.7	13.2	2.2	1.5	3.1	0.4	0.2	0.7	222	201
Outside Wilderness / Roadless	Eastside Forests	4.4	3.5	5.3	1.1	0.8	1.4	0.2	0.1	0.3	538	428
	Beaverhead Deerlodge	2.9	1.9	4.0	0.7	0.4	1.0	0.2	0.1	0.3	222	189
	Custer	3.2	1.5	5.2	1.1	0.4	1.8	0.0	0.0	0.2	111	61
	Gallatin	7.4	4.2	11.0	2.5	1.2	3.9	0.4	0.1	0.9	64	53
	Helena	5.0	2.7	7.6	1.2	0.4	2.1	0.4	0.1	0.8	64	59
	Lewis & Clark	6.7	3.6	10.6	0.9	0.3	1.6	0.1	0.0	0.3	77	66

3.2 Estimates of Snag Density by Habitat Type Groups

Second, estimates of large-snag density, by aggregations of habitat types (Pfister, 1977), referred to as habitat type groups, commonly used for eastside vegetation assessments (Appendix B) were derived (Table 3). Each of these habitat type groups have similar biophysical and disturbance regime characteristics that determine snag abundance ranges during various stages of succession. Habitat type groups were used instead of Vegetative Response Units (VRUs) because habitat type groups are a consistent classification utilized across all eastside Forests for planning and analysis.

Within wilderness/roadless areas, some habitat type groups contain similar densities of large snags (e.g., warm and very dry, warm and dry). Habitat groups with similar snag densities are shaded in Table 3. Each of these shaded habitat type groups have characteristic disturbance regimes that are generally different between groups and contribute to snag abundance, during various stages of succession, in different ways. One of the differences is the numbers of snags produced. The warm groupings have fewer snags, most likely due to frequent, low- to mid-severity fire that tended to produce a relatively constant level of snags at low numbers. The cool group, with a characteristic fire regime that tended to have less frequent, but with more severe fires, produced pulses of snags, and generally a greater quantity of snags, especially early in the forest succession cycle. Then as stands aged, the density of snags increased, until another high-severity stand replacing fire occurs. The cold types tend to produce high snag densities as characteristic disturbance regimes produced persistent snags over a long periods due to colder climates, where decomposition rates are slower, and the period of time between stand replacing events were likely the longest. One again, we see that the larger the snag, the less common they are within the forest. Individual Forest's snag densities, by these habitat type groups are displayed in Appendix C, Table 1.

Table 3: Mean snag density per acre and 90% confidence interval, by diameter class, inside and outside of wilderness/roadless areas by initial habitat type groups, for all eastside Forests.

Area	Habitat Type Group	Snags per Acre 10"+			Snags per Acre 15"+			Snags per Acre 20"+			Total # PSUs	# Forested PSUs
		Mean	90% CI Lower Bound	90% CI Upper Bound	Mean	90% CI Lower Bound	90% CI Upper Bound	Mean	90% CI Lower Bound	90%CI Upper Bound		
In Wilderness / Roadless	Warm & Very Dry	5.1	2.8	7.9	1.4	0.7	2.3	0.3	0.1	0.6	105	105
	Warm & Dry	5.8	1.8	10.9	1.4	0.5	2.6	0.3	0.0	0.6	44	44
	Warm & Moist	4.7	2.1	7.8	1.1	0.2	2.2	0.3	0.0	0.9	31	31
	Cool & Moist	12.2	7.6	17.5	2.5	1.0	4.3	0.5	0.1	1.0	68	68
	Cool & Dry to Moist	14.0	11.0	17.1	3.0	2.2	4.0	0.6	0.4	0.9	207	207
	Cool & Moist to Wet	16.2	9.8	23.6	2.6	1.3	4.3	0.5	0.1	0.9	36	36
	Warm to Cool & Dry	13.7	8.7	19.4	3.5	2.0	5.3	0.8	0.3	1.4	45	45
	Cool & Wet	18.3	12.0	25.1	5.5	3.2	8.2	1.5	0.5	2.9	39	39
	Cold & Dry to Wet	18.9	14.2	24.1	4.4	3.0	6.1	1.1	0.6	1.6	95	95
	Cold & Dry	11.5	6.8	16.9	3.3	2.0	4.6	1.1	0.6	1.6	60	60
Outside Wilderness / Roadless	Warm & Very Dry	2.0	1.2	3.1	0.8	0.4	1.2	0.2	0.0	0.3	124	124
	Warm & Dry	3.1	1.5	4.9	0.9	0.3	1.7	0.4	0.1	0.8	53	53
	Warm & Moist	2.5	0.8	4.5	0.4	0.0	1.0	0.2	0.0	0.5	39	39
	Cool & Moist	3.4	1.5	5.5	0.8	0.0	1.8	0.2	0.0	0.4	51	51
	Cool & Dry to Moist	5.2	3.1	7.5	0.8	0.3	1.5	0.1	0.0	0.3	97	97
	Cool & Moist to Wet	6.6	0.0	17.0	1.2	0.0	4.6	0.0	0.0	0.0	5	5
	Warm to Cool & Dry	12.5	6.1	20.0	3.2	1.3	5.6	0.9	0.1	2.0	19	19
	Cool & Wet	6.7	1.5	13.6	1.6	0.0	4.1	0.0	0.0	0.0	15	15
	Cold & Dry to Wet	16.6	5.9	30.4	4.0	1.4	7.0	0.4	0.0	1.1	15	15
	Cold & Dry	11.5	0.0	31.2	2.0	0.0	7.7	0.0	0.0	0.0	5	5

USDA

United States Department of Agriculture

Forest Service

April 2018

Glacier Loon Fuels Reduction and Forest Health Project

Draft Decision Notice

Swan Lake Ranger District, Flathead National Forest
Missoula County, Montana

OLD GROWTH ASSOCIATED WILDLIFE

INTRODUCTION

Old growth is defined in Amendment 21 of the Forest Plan as "a community of forest vegetation that has reached a late stage of plant succession." The generic description is as follows:

- The age of the dominant cohort of trees is significantly older than the average time interval between natural disturbances (interval will vary depending upon forest cover type and habitat type);
- Forest composition and structure are different from younger stands;
- Rates of change in composition and structure of the stand are slow relative to younger forests;
- There is a significant showing of decadence (wide range of defect and breakage in both live and dead trees).

In The Dictionary of Forestry (Helms 1998), old growth forests are described as having:

- Large trees for the species and site;
- Accumulations of large dead standing and fallen trees;
- Decay or breakage of tree tops, boles, or roots;
- Multiple canopy layers;
- A wide variation in tree size and spacing; and
- Canopy gaps and understory patchiness.

The characteristics of old growth forest described above provide habitat for many plant and animal species. Old growth forests are an important component of biological diversity. For the purpose of this discussion, old growth associated species includes any wildlife species that use the various attributes of old growth forests for some or all of their ecological needs. These needs could include nesting, denning, security, or foraging habitat. For some species, closed canopy old growth provides snow capture and reduces snow depths, insulates the animals from cold winds, and provides protection from predators. Some species, such as the fisher, are strongly tied to canopy cover and mature forest structure for the majority of their habitat needs. More open canopies, or open understories, provide foraging opportunities for prey and predator species alike. Wildlife may use interior old growth habitat as shelter from sun, heat, dryness, or wind and old growth cover may provide protection from predators. Some old growth associated wildlife species need only a portion of their home range to be in old growth; examples include the Canada lynx, northern goshawk, and American marten. Other species such as southern red-backed voles, chestnut-backed chickadee, Swainson's thrush, and northern flying squirrels, have relatively small home range sizes (less than 400 acres), with the necessary proportion of this home range being in old growth unknown.

The following table displays 31 old growth associated species that may be found in the Swan Valley, along with their associations with various old growth habitat characteristics (USDA 1999b).

TABLE 3- 87 HABITAT REQUIREMENTS OF OLD GROWTH ASSOCIATED WILDLIFE SPECIES. (BASED ON WARREN 1998 AND LRMP AMENDMENT 21 FEIS).

SPECIES	COVER TYPE IN AFFECTED AREA	CANOPY	EDGE	LARGER PATCHES	SNAG	DOWN LOG	OCCURRENCE
American Marten	Mixed mesic, lodgepole, spruce/fir forests	Closed	-	+	X	X	Known current
Bald Eagle (S)	Mixed mesic forests, near large lake or river	Open		+	X		Known current
Black-backed Woodpecker (S)	Lower Montane & Montane; post-fire or insect-epidemic forests	Open			X		Known current
Boreal Owl	Mixed mesic and spruce/fir forest mosaic	Closed			X	X	Known current
Brown Creeper	Mixed mesic, lodgepole, and spruce/fir forests	Closed	-		X		Known current
Canada Lynx (T1)	Mixed mesic, lodgepole, and spruce/fir forests; gentle terrain		+2	+	X3	X	Known current
Chestnut-Backed Chickadee	Mixed mesic and spruce/fir forests, especially cedar-hemlock	Closed	-4		X		Known current
Fisher (S5)	Mixed mesic and lodgepole forests	Closed				X	Known current
Flammulated Owl (S, F6)	Lower Montane and Montane, single-story.	Open			X		Known current
Golden-crowned Kinglet	Mixed mesic, lodgepole, and spruce/fir forests	Closed		+	X		Known current
Hairy Woodpecker	Mixed mesic, lodgepole, and spruce/fir forests	Open			X	X	Known current
Hammond's Flycatcher (F)	Mixed mesic and spruce/fir forests	Closed					Known current
Harlequin Duck (S)	Swift mountain streams, riparian old growth (weak association)	Open				X	Known current
Hermit Thrush	Dry mixed mesic and spruce/fir forests	Open		+			Known current
Lewis' Woodpecker	Lower Montane ponderosa pine and old burns	Open			X		Known current
Northern Flying Squirrel	Mixed mesic and lodgepole forests			+	X	X	Known current
Northern Goshawk	Single or multistory old growth; clear forest floor	Closed		+	X		Known current
Pileated Woodpecker	Mixed mesic forests	Closed		+	X	X	Known current
Pine Grosbeak	Mixed mesic, lodgepole, and spruce/fir forests						Known current
Pygmy Nuthatch	Large single-story ponderosa pine and mixed mesic forests	Open			X		Known current
Red-Breasted Nuthatch	Mixed mesic, lodgepole, and spruce/fir; relatively dry	Open		+	X		Known current
Silver-haired Bat	Mixed mesic and lodgepole forests; caves and snags				X		Suspected
Southern Red-backed Vole	Mixed mesic, lodgepole, and spruce-fir forest				X	X	Known current
Swainson's Thrush (F)	Mixed mesic and lodgepole forest with shrub understory			+			Known current
Tailed Frog	Cold, high gradient headwater streams					X	Known current

TABLE 3- 87 HABITAT REQUIREMENTS OF OLD GROWTH ASSOCIATED WILDLIFE SPECIES. (BASED ON WARREN 1998 AND LRMP AMENDMENT 21 FEIS).

Species	Cover Type in Affected Area	Canopy	Edge	Larger Patches	Snag	Down Log	Occurrence
Three-toed Woodpecker	Mixed mesic, lodgepole, and spruce/fir forests; post-fire				X		Known current
Townsend's Warbler	Mixed mesic and lodgepole forest; dense understory	Closed	-	+			Known current
Varied Thrush	Mixed mesic and spruce/fir forests, especially cedar-hemlock	Closed		+			Known current
Vaux's Swift (F)	Mixed mesic and spruce/fir forests; large hollow snags				X		Known current
White-breasted Nuthatch	Large single-story ponderosa pine	Open			X		Known current
Winter Wren	Mixed mesic and spruce/fir forests, especially cedar-hemlock		-	+	X		Known current

[1] T = Threatened
[2] + = Positive correlation (where known)
[3] X = Important Habitat Component
[4] - = Negative correlation (where known)
[5] S = Sensitive
[6] F = Forest-dwelling Neotropical migrant with apparently declining populations

ANALYSIS AREA

SPATIAL BOUNDS

The effects analysis area for direct, indirect, and cumulative effects to old growth associated wildlife species is the Glacier Loon Project Area (37,320 acres). This area is large enough to include the home ranges of old growth associated species, and is representative of the effects of fire, natural tree mortality, timber harvest, and road management across the landscape. At the same time, this analysis area is small enough to not obscure the effects of the alternatives. A multi-scale assessment has also been conducted to address habitat diversity concerns.

TEMPORAL BOUNDS

The length of time for effects from the proposed fuels reduction and forest health treatments is approximately 1 to 5 years. This is based on the probable contract length for the proposed project, and the timeframes for related activities.

DATA SOURCES, METHODS, AND ASSUMPTIONS USED

Data used included stand exams, field surveys of snags and down woody logs, old growth surveys, project area field visits, research literature, and GIS and dataset information for features, such as general forest attributes, habitat type, and forest type.

MEASUREMENT INDICATORS

The effects analysis will focus on:

1. Effects to old growth habitat, and
2. Potential effects to old growth associated wildlife species.

April 2018

SNAG AND DOWN WOODY ASSOCIATED SPECIES

INTRODUCTION

Snags, broken-topped live trees, down logs, and other woody material are required by a wide variety of species for nesting, denning, roosting, perching, feeding, and cover (Bull et al. 1997). Snags and down, dead, material are also used for communication purposes:

- Singing, (songbirds),
- Drumming (grouse and woodpeckers),
- Calling (squirrels, jays, birds of prey), and
- Sight recognition posts.

Small mammals and birds use standing and down dead material for food storage and for hunting. Down logs and stumps are important for travel, both below the snow in the winter, and as travel cover throughout the year. It is estimated that about one third of the bird and one third of the mammal species that live in the forests of the Rocky Mountains use snags for nesting or denning, foraging, roosting, cover, communication, or perching. On the Flathead National Forest, at least 42 species of birds and 10 species of mammals are dependent on dead wood habitat for nesting, feeding, or shelter (USDA 1999b). The more mobile species that depend on dead wood habitat include black bears, Canada lynx, wolverines, marten, fisher, bats, woodpeckers, and small owls. Less mobile species that depend on dead wood include snowshoe hares (the primary prey of Canada lynx), red-backed voles (the primary prey of marten, fisher, boreal owl, and several other species), shrews, bryophytes, lichen, fungi, and protozoa. As down woody material further decays, it plays an important role in nutrient cycling, soil fertility, and erosion control.

Snags and their management have become a major conservation issue in managed forests across the western United States. Biologists have recognized for a long time that snags and down woody material provide important wildlife habitat, but only in the last decade or so have managers begun to understand that not only is tree decay an important ecological process that affects wildlife habitat (Bull et al. 1997), but snags and dead wood are an essential, important part of the larger ecosystem. The number, species, size, and distribution of available snags strongly affect snag-dependent wildlife. An insufficient number of suitable snags may limit or eliminate populations of cavity-using species (Saab et al. 1998; Thomas et al. 1979).

Although various sizes of snags and down woody are used, larger birds and mammals require larger dead trees. The larger-diameter down trees provide stable and lasting structure and offer better protection from weather extremes (Bull 2002). Longer down woody pieces provide better runways, shelter, and under-snow access.

Several wildlife species that use snag and down woody habitats on the Flathead National Forest are USFS Region One Sensitive Species, including the bald eagle, black-backed woodpecker, fisher, flammulated owl, Townsend's big-eared bat, and wolverine. One of the TES Species on the Flathead National Forest, the Canada lynx, has a strong habitat association with down woody material (denning).

Snags are essential habitat for at least 42 species of birds and 10 species of mammals in Montana. Table 3- 88 displays specific habitat relationships and Montana NHP rankings for wildlife species in Montana associated with snag, "defective" live tree, or down woody habitat.

TABLE 3- 89 SPECIES THAT USE SNAGS, "DEFECTIVE" LIVE TREES, AND/OR DOWNED LOGS.

Species	Global & State Ranks (MTNHP 2009) *	Snag DBH (inches)	Snag Height (feet)	Downed Logs?	Occurrence
American Kestrel (N)	G5, S5B	17	20		Known current
Bald Eagle (S)	G5, S3, SOC	25	40		Known current
Barred Owl (former MIS)	G5, S4	25	30		Known current
Barrow's Goldeneye	G5, S4, potential SOC	25	10		Known current
Big Brown Bat	G5, S4	17	20		Known current
Black-backed Woodpecker (S)	G5,S3 (SOC)	17	10		Known current
Black-capped Chickadee	G5, S5	9	10		Known current
Bobcat	G5, S5	-	-	yes	Known current
Boreal Chickadee	G5,S3 (SOC)	9	10		Known current
Boreal Owl (former S)	G5, S4	17	10		Known current
Brown Creeper	G5, S3, SOC	15	20		Known current
Bufflehead	G5, S5B	17	10		Known current
Canada lynx (T)	G5, S3, SOC	-	-	yes	Known current
Chestnut-backed Chickadee	G5, S4	9	10		Known current
Common Goldeneye	G5, S5	25	10		Known current
Common Merganser	G5, S5B	17	10		Known current
Dark-eyed junco	G5, S5B	-	-	yes	Known current
Downy Woodpecker	G5, S5	11	10		Known current
Fisher (S)	G5, S3, SOC	25	30	yes	Known current
Flammulated Owl (S, N)	G4, S3B, SOC	17	10		Known current
Great Horned Owl	G5, S5	25	30		Known current
Hairy Woodpecker	G5, S5	17	20		Known current
Harlequin Duck (S)	G4, S2B, SOC	-	-	yes	Known current
Hooded Merganser	G5, S4, potential SOC	17	10		Known current
House Finch	G5, S5	15	10		Known current
House Sparrow	G5, undesired species	15	20		Known current
House Wren (N)	G5, S5B	15	10		Known current
~~Lewis' Woodpecker~~	~~G4, S2B (SOC)~~	~~17~~	~~30~~		~~Known current~~
Little Brown Myotis	G5, S4	17	10		Known current
Long-eared Myotis	G5, S4	17	10		Known current
Long-legged Myotis	G5, S4	17	10		Known current
Long-tailed Weasel	G5, S5	-	-	yes	Known current
Marten (former MIS)	G5, S4	17	20	yes	Known current
Mountain Bluebird	G5, S5B	15	10		Known current
Mountain Chickadee	G5, S5	9	10	yes	Known current
Northern Alligator Lizard	G5,S3 (SOC)	-	-	yes	Known current
Northern Flicker	G5, S5	17	10		Known current
Northern Flying Squirrel	G5, S4	17	20		Known current
Northern Goshawk (former S)	G5,S3 (SOC)	-	-	yes	Known current
Northern Hawk Owl	G5, S4, potential SOC	25	10		Known current
Northern River Otter	G5, S4	-	-	yes	Known current
Northern Waterthrush (N)	G5, S5B	-	-	yes	Known current
Osprey	G5, S5B	17	40		Known current

TABLE 3- 89 SPECIES THAT USE SNAGS, "DEFECTIVE" LIVE TREES, AND/OR DOWNED LOGS.

SPECIES	GLOBAL & STATE RANKS (MTNHP 2009) *	SNAG DBH (INCHES)	SNAG HEIGHT (FEET)	DOWNED LOGS?	OCCURRENCE
Painted Turtle	G5, S4	-	-	yes	Known current
Pileated Woodpecker (former MIS)	G5,S3 (SOC)	25	60		Known current
Pygmy Nuthatch	G5, S4	17	30		Known current
Pygmy Owl	G5, S4	17	30		Known current
Raccoon	G5, S5	25	10		Known current
Red-breasted Nuthatch	G5, S5	17	20		Known current
Red-naped Sapsucker (N)	G5, S4B	17	20		Known current
Rubber Boa	G5, S4	-	-	yes	Known current
Ruffed Grouse	G5, S4	-	-	yes	Known current
Saw-whet Owl	G5, S4	17	20		Known current
Silver-haired Bat	G5, S4, potential SOC	17	20		Known current
Southern Red-backed Vole	G5, S4	-	-	yes	Known current
Spruce Grouse	G5, S4	-	-	yes	Known current
Striped Skunk	G5, S5	-	-	yes	Known current
Swainson's Thrush (N)	G5, S5B	-	-	yes	Known current
Tailed Frog	G5, S4	-	-	yes	Known current
Three-toed Woodpecker	G5, S4	17	20		Known current
Tree Swallow (N)	G5, S5B	15	20		Known current
Vaux's Swift (N)	G5, S4B	25	40		Known current
Violet-Green Swallow	G5, S5B	15	20		Known current
Western Bluebird	G5, S4B	15	10		Known current
Western Jumping Mouse	G5, S4	-	-	yes	Known current
Western Screech Owl	G5, S3, potential SOC	17	20		Known current
Western (Townsend's) Big-eared Bat (S)	G4, S2 (SOC)	?	?		Known current
White-breasted Nuthatch	G5, S4	17	20		Known current
Williamson's Sapsucker (N)	G5, S4B	17	20		Known current
Wilson's Warbler (N)	G5, S5B	-	-	yes	Known current
Wolverine (S)	G4, S3 (SOC)	-	-	yes	Known current
Wood Duck	G5, S5B	25	10		Known current
Yuma Myotis	G5, S3, potential SOC	17	10		Known current

T=Threatened; S=Sensitive Species; N-Neotropical migratory bird; Natural Heritage Program Rank: G=species range-wide (global); S=state wide; 2=At risk because of very limited and/or declining numbers, range, and/or habitat, making it vulnerable to global extinction or extirpation in the state. 3=Potentially at risk because of limited and/or declining numbers, range, and/or habitat, even though it may be abundant in some areas. 4=Uncommon but not rare (although it may be rare in parts of its range), and usually widespread. Apparently not vulnerable in most of its range, but possibly cause for long-term concern. 5=Common, widespread, and abundant (although it may be rare in parts of its range). Not vulnerable in most of its range. B=State rank modifier indicating breeding for a migratory species. SOC=Montana Species of Concern.

ANALYSIS AREA

SPATIAL BOUNDS

The Glacier Loon Project Area was considered for the evaluation of direct and indirect effects on snag and down woody associated species. This approximately 37,320-acre area is large enough to include the home ranges of several individuals or pairs of a species, and is representative of the effects of fire, natural tree mortality, timber harvest, and road management across the landscape. The actions proposed in the alternatives that could directly or indirectly affect snag or down woody associated wildlife species are contained within this area. The Upper Swan Valley was considered in the cumulative effects analysis. A multi-scale assessment was also conducted to address habitat diversity concerns for dead tree dependent species (USDA 2006).

TEMPORAL BOUNDS

The length of time for effects from the proposed fuels and forest health treatments is approximately 1 to 5 years. This is based on the probable contract length for the proposed project, and the timeframes for related activities.

DATA SOURCES, METHODS, AND ASSUMPTIONS USED

Data used included project area field visits, research literature, and GIS and dataset information for features, such as general forest attributes, habitat type, and forest type.

MEASUREMENT INDICATORS

The effects analysis will focus on:

1. Effects to snag and down woody habitat, and
2. Potential effects to snag/down woody associated wildlife species.

AFFECTED ENVIRONMENT

HISTORICAL CONDITION

Forest ecosystems in the western United States have adapted in response to disturbances such as wildfire, insects, disease, and windstorms. Snags and down woody material have always occurred on the landscape, a direct result of these disturbance factors, either on a large scale, or on a very small scale, as individual trees grow old and die. Ritter and others have described snag populations as occurring in either "pulses" of snags following a large disturbance event, or as "continuous" populations of scattered individuals (Ritter et al. 2000).

Historically, in the Swan Valley, snag habitat and down woody material, though always present in varying amounts, experienced greater "pulses" across the landscape and in localized areas as a result of natural disturbances. Warmer and drier areas historically underwent more frequent, lower-intensity fires, and typically supported fewer snags and large down logs than cooler and moister environments, where the stands reached climax conditions before experiencing stand-replacing fire.

EXISTING CONDITION

The Northern Region of the Forest Service estimated snag densities for Western Montana Forests by

United States Department of Agriculture

Forest Service

Pacific Northwest Research Station

Research Paper PNW-RP-564
February 2005

USDA

Short-Term Effects of Fuel Reduction on Pileated Woodpeckers in Northeastern Oregon—A Pilot Study

Evelyn L. Bull, Abe A. Clark, and Jay F. Shepherd

Introduction

Widespread outbreaks of the western spruce budworm (*Choristoneura occidentalis*) and Douglas-fir beetle (*Dendroctonus pseudotsugae*) in the 1980s and early 1990s caused heavy tree mortality in stands with grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) in northeastern Oregon (Gast et al. 1991). The outbreaks resulted in high tree mortality that increased the fuel loading and the risk of stand-replacement wildfires (Gast et al. 1991). The increase in wildfires nationally in the early part of this century resulted in an increased emphasis on reducing fuels within forest stands. Currently, the primary methods of reducing fuels are the mechanical removal of trees and coarse woody debris (standing or down dead wood) and prescribed burning.

Certain wildlife species in northeastern Oregon, including small mammals, amphibians, rubber boas (*Charina bottae*), martens, bears, and woodpeckers, depend heavily on the coarse woody debris that is removed during these fuel reduction treatments (Tiedemann et al. 2000). American martens (*Martes americana*) frequently capture prey in accumulations of logs, and use rest and den sites in logs and snags (Bull and Heater 2000). Ants found in dead wood are the primary prey of pileated woodpeckers (*Dryocopus pileatus*) (Bull et al. 1992, Torgersen and Bull 1995). Other log-dwelling and log-associated insects, mainly ants and yellowjackets, comprise 24 percent of black bear (*Ursus americanus*) diet in northeastern Oregon (Bull et al. 2001).

Inadequate amounts and kinds of dead wood could affect the beneficial role that foliage-foraging ants and other forest-floor arthropods have in maintaining forest health. Ants also serve an important role in forest health in preying on the western spruce budworm, one of the most important forest-defoliating insects in the Pacific Northwest (Torgersen et al. 1990). Many other species of forest-floor arthropods are predators of the western spruce budworm, and some parasites of the western spruce budworm depend on the forest floor for a portion of their life cycle (Tiedemann et al. 2000). Populations of all forest-floor arthropods were significantly lower in areas that had been harvested and burned in comparison to adjacent undisturbed forests 3 years after treatment (Fellin 1980). Prescribed burning severely affected groups of forest-floor fauna by directly killing the organism or by removing woody material and forest floor that are required by these insects for food and shelter (Fellin 1980).

Although few studies directly have investigated the results of fuel reduction treatments on wildlife in the Pacific Northwest, evidence suggests that some species are negatively affected by this type of management, at least in the short term, whereas others are unaffected or benefit. Numbers of red squirrels (*Tamiasciurus hudsonicus*), snowshoe hares (*Lepus americanus*), and southern red-backed voles

Results

The objective of the fuel reduction treatments was to decrease stem density and coarse woody debris, so it was logical that significant differences occurred in the number of logs, snags, and stumps among the three treatments ($X^2 = 208.01$, 4 df, $P < 0.01$) with the highest number of each occurring in the control stands (fig. 1).

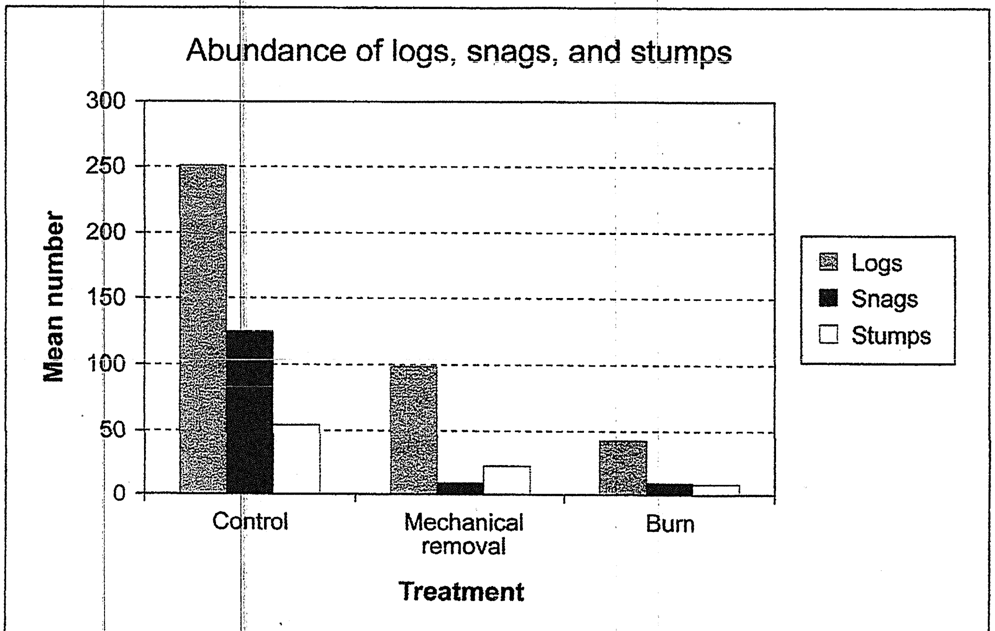

Figure 1—Mean number of logs on 1000 m of transects in seven stands and the mean number of snags and stumps in a 1-ha plot in seven stands in control, mechanical removal, and prescribed burn treatments on the Starkey Experimental Forest and Range in northeastern Oregon, 2004.

Logs

We tallied 1,764 logs in the control plots, 693 in the mechanical removal plots, and 299 in the prescribed burn plots. Significant differences occurred among the treatments in log large-end diameter ($F = 20.59$, $P < 0.01$), length ($F = 329.16$, $P < 0.01$), species ($X^2 = 48.36$, $P < 0.01$), and wood condition ($X^2 = 49.84$, $P < 0.01$). Logs remaining after the mechanical removal treatment were significantly larger (mean = 33 cm, standard error = 0.54) than in either the control (mean = 29 cm, standard error = 0.31) or burn treatments (mean = 29 cm, standard error = 0.75). Logs in the control treatment were significantly longer (mean = 9 m, standard error = 0.12) than those in the burn and mechanical removal treatments (burn: mean = 5 m, standard error = 0.29; mechanical: mean = 4 m, standard error = 0.13). In all treatments, 83 to 91 percent of the logs were in the intermediate decay class, with the highest proportion of logs with advanced decay (16 percent) occurring in the mechanical removal treatment.

65/879 = 93%

Snags

We tallied 879 snags in the control plots, 65 in the mechanical removal plots, and 63 in the prescribed burn plots; there were no significant differences in snag numbers between the two fuel reduction treatments (fig. 1). Significant differences occurred in snag diameter ($F = 40.66$, $P < 0.01$), height ($F = 13.51$, $P < 0.01$), species ($X^2 = 16.29$, $P < 0.01$), and condition ($X^2 = 10.51$, $P < 0.01$) among the treatments. Although the stands with either fuel reduction treatment had fewer snags than the control, a higher percentage of the snags that occurred in these treatments were larger than 50 cm d.b.h. compared to the control treatments. Snag d.b.h. differed significantly among all the treatments. The mean d.b.h. of snags was 35 cm (standard error = 0.63) in the control treatments, 46 cm (standard error = 2.88) in the prescribed burn treatments, and 55 cm (standard error = 3.17) in the mechanical removal treatments. Sixty-three percent of the snags in the control treatments were less than 35 cm d.b.h., whereas 35 percent of those in the prescribed burn treatments and 25 percent of those in the mechanical removal treatments were this size. Snags in the control treatment were significantly shorter (mean = 9.3 m, standard error = 0.17) than those in the prescribed burn/mechanical removal treatments (mean = 12.4 m, standard error = 0.90) or the mechanical removal treatments (mean = 11.3 m, standard error = 0.87), which is consistent with snags in the control treatments being smaller in diameter. Wood condition differed significantly among the treatments where the burn treatments contained 11 percent of recently dead trees compared with 2 percent and 4 percent in the mechanical removal and control treatments, respectively.

Stumps

We tallied a total of 379 stumps in the control plots, 154 in the mechanical removal plots, and 57 in the prescribed burn plots (fig. 1). Significant differences occurred in stump diameter ($F = 14.74$, $P < 0.01$), height ($F = 8.35$, $P < 0.01$), and species ($X^2 = 27.28$, $P < 0.01$) among the treatments. Although the stands with either fuel reduction treatment had fewer stumps, the average diameter was 38 cm (standard error = 2.24) in the prescribed burn treatments, 41 cm (standard error = 1.78) in the mechanical removal, and 32 cm (standard error = 0.80) in the control treatments. Stumps in the control stands were significantly taller (mean = 2.2 m, standard error = 0.04) than those in the prescribed burn treatments (mean = 2.0 m, standard error = 0.11) or the mechanical removal (mean = 1.9 m, standard error = 0.06) largely because most of the stumps in the treated stands were a result of the harvesting treatment.

Presence of Woodpecker Foraging and Ants

Pileated Woodpeckers—We detected 534 dead wood substrates (i.e., logs, snags, and stumps) with pileated foraging. The majority (71 percent) of the foraging activity in all substrates involved large rectangular or deep excavations where pileated woodpeckers appeared to be foraging on ants or wood-boring beetle (cerambycids and buprestids) larvae, 17 percent were shallow excavations where bark beetles likely had been present, and the remainder were small, isolated shallow excavations.

Overall, we detected 216 logs with recent pileated woodpecker foraging; 200 of those logs were on transects and 16 did not cross a transect. The majority of these logs with foraging occurred in the control treatment (fig. 2). Pileated foraging in logs was significantly more abundant in the control and mechanical removal treatments than in the prescribed burn treatments (control versus prescribed burn: $X^2 = 4.34$, 1 df, $P = 0.04$; mechanical removal versus prescribed burn: $X^2 = 3.82$, 1 df, $P = 0.05$). Pileated woodpeckers selected substrates with ants. Forty-three percent of transect logs had ants, whereas 62 percent of logs with pileated woodpecker foraging had ants ($X^2 = 33.83$, 1 df, $P < 0.01$). Ants were significantly more abundant in logs in the control and mechanical removal treatments than in the prescribed burn treatments ($X^2 = 28.92$, $P < 0.01$). Ants occurred in 46 percent of transect logs in the control, 44 percent in the mechanical removal treatments, and in 29 percent in the prescribed burn treatments. The presence of charring on logs influenced the occurrence of ants and pileated woodpecker foraging activity. Logs with ants had

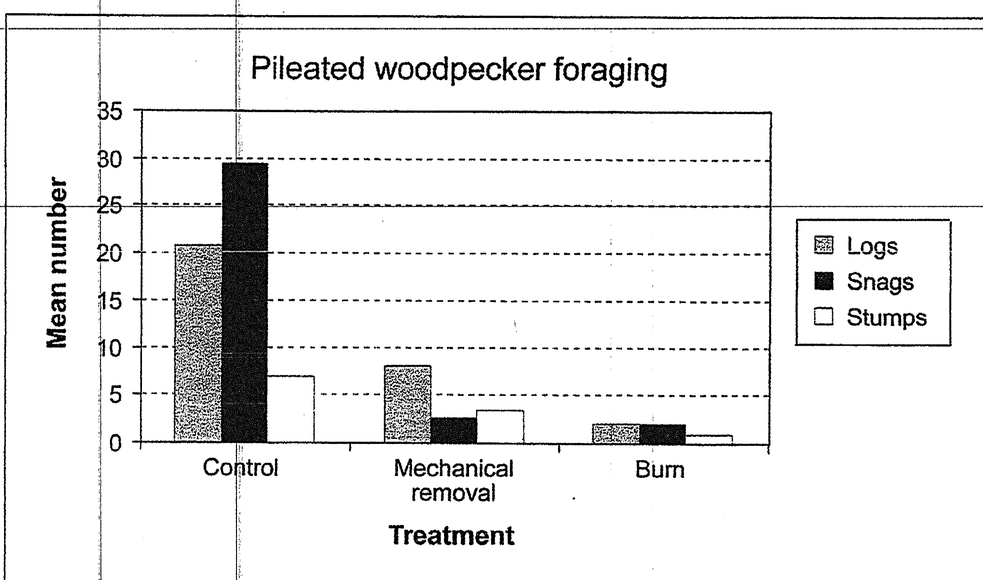

Figure 2—Mean number of logs, snags, and stumps with pileated woodpecker foraging in 1-ha plots in seven stands in control, mechanical removal, and prescribed burn treatments on the Starkey Experimental Forest and Range in northeastern Oregon, 2004.

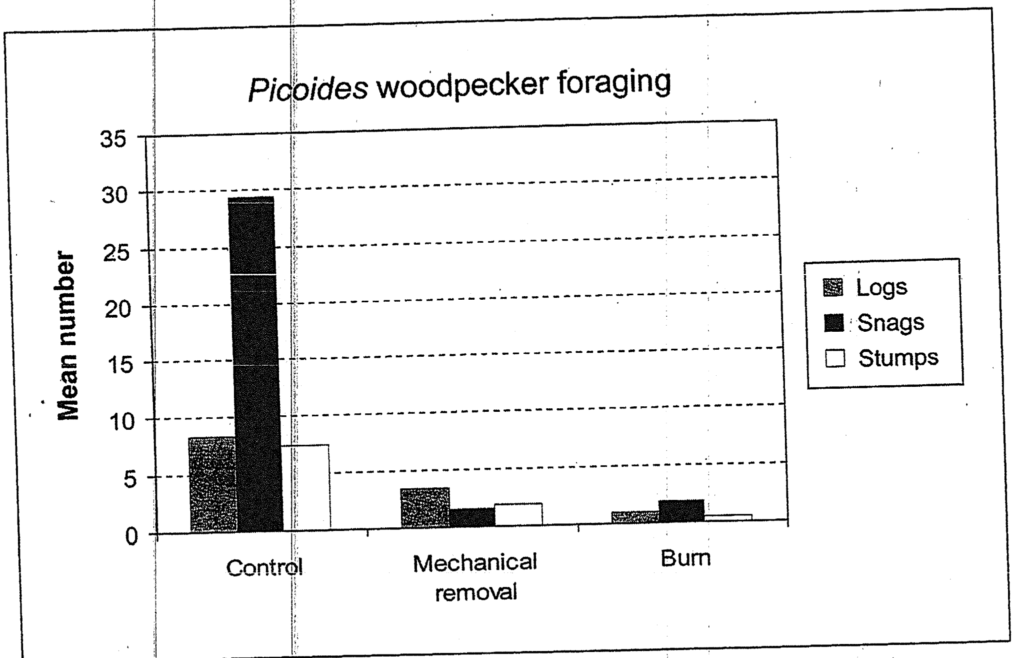

Figure 3–Mean number of logs on 1000 m of transects in seven stands and the mean number of snags and stumps in a 1-ha plot with foraging evidence of *Picoides* woodpeckers in control, mechanical removal, and prescribed burn treatments on the Starkey Experimental Forest and Range in northeastern Oregon, 2004.

$P = 0.04$) and stumps ($X^2 = 6.04$, 1 df, $P = 0.01$) with and without evidence of ants. *Picoides* foraged more in snags and stumps without ants than snags and stumps with ants. In snags, 62 percent of the foraging occurred with no evidence of ants versus 38 percent with ants. In stumps, 56 percent of the foraging occurred with no evidence of ants versus 44 percent with ants. *Picoides* foraging in logs did not differ with or without evidence of ants. The percentage of charring in the prescribed burn treatments differed significantly between stumps with *Picoides* foraging (mean of 22 percent charring) and stumps without *Picoides* foraging (mean of 56 percent charring) ($t = 3.21$, 40 df, $P < 0.01$).

Discussion

Both the control treatments and the mechanical removal treatments provided significantly more foraging habitat for pileated woodpeckers, whereas the prescribed burn treatments provided significantly less. The higher incidence of ants in the control treatments and mechanical removal treatments explains the greater use by these woodpeckers. In this study area, pileated woodpeckers are known to forage primarily on ants, specifically *Camponotus* and *Formica* (Bull et al. 1992). The higher incidence of pileated woodpecker foraging in substrates with ants suggests that their foraging is not random and that they were able to detect ants in substrates. Consequently, their foraging is more efficient than random selection of substrates, and the energy they expend foraging is therefore minimized.

A significant difference in foraging habitat of the smaller *Picoides* woodpeckers was not detected among the treatments (fig. 3), except in their avoidance of charred stumps in the prescribed burn treatments. Unlike the pileated woodpecker, these woodpecker species do not concentrate their foraging extensively on ants, likely because the *Camponotus* and *Formica* ants are inside dead wood, which would be largely unavailable to these woodpeckers. Both the black-backed and hairy woodpeckers forage extensively by bark scaling and pecking on trees in Starkey and seldom excavate in the interior wood (Bull et al. 1986). Black-backed woodpeckers are frequently associated with recent stand-replacement fires (Hutto 1995, Kotliar et al. 2002, Saab and Dudley 1998), and recent mortality of trees killed by burning would result in an increase in bark beetles and wood-boring beetles, which the *Picoides* species of woodpeckers forage on. Black-backed woodpeckers feed extensively on larvae of wood-boring beetles (buprestids, cerambycids), engraver beetles, and bark beetles (scolytids; Dixon and Saab 2000, Murphy and Lehnhausen 1998). Hairy woodpeckers forage on larvae of bark beetles, codling moth, pupae of cecropia (*Hyalophora cecropia*), as well as ants (Jackson et al. 2002).

In a comparison of prescribed burning and harvesting in southeastern British Columbia, Machmer (2002) found no difference in woodpecker foraging among treatments after partial harvesting and burning in the short term (1 to 2 years). After treatments, she detected increases in insect species diversity in Buprestidae, Cerambycidae, and Scolytidae, which are all species that inhabit dead and dying wood and serve as prey for woodpeckers. The increase in insects was most pronounced in the burn only treatment, followed by harvest and burning, and then by harvest only. Prescribed burns in northern Idaho resulted in most of the small woody fuels and 15.5 t/acre (34 746 kg/ha) of large woody fuels being consumed (Reinhardt et al. 1991). In spruce (*Picea* spp.) ecosystems in Alaska, Werner (2002) found that both fire and timber harvest attracted woodborers and bark beetles the first year after disturbance, but populations then decreased to levels below those in undisturbed sites.

The absence of management activities in the control treatments in our study resulted in an abundance of logs, snags, and pileated woodpecker foraging (figs. 1 and 2). Because no logs or snags were removed from the control treatments, the average diameter and height or length of snags and logs were smaller than in the fuel reduction treatments where the smaller snags and logs were targeted for removal. However, the control treatments actually contained a larger number of large-diameter snags and logs compared to the fuel reduction treatments. The control treatments also contained the highest number of logs and snags with all species of ants, including *Camponotus* and *Formica*, which provided the most foraging habitat for pileated woodpeckers.

Although foraging by pileated woodpeckers in mechanical removal treatments was not as common as in the control treatments, there was significantly more foraging than in the prescribed burn treatments. The presence of *Camponotus* and *Formica* ants in the mechanical removal treatment provided prey for pileated woodpeckers. The removal of standing trees alone did not prevent pileated woodpeckers from using the stands. The larger diameter logs and snags found in these stands, as well as in the burn treatment, were retained during harvesting owing to management guidelines and to their low susceptibility to wildfire and high value to wildlife. The shorter length of logs and the higher incidence of logs in the advanced decay class found in this treatment likely resulted from the logging equipment running over the logs and breaking them.

The lower occurrence of ants in logs, snags, and stumps in the prescribed burn treatment suggests that the burning either directly eliminated the ants or rendered the habitat unsuitable for ants. In the burn treatment, the logs in the advanced decay class would probably have been consumed. Our observation that logs with ants had less charring than logs without ants suggests that charring on logs may make the logs less attractive to ants or that ants were extirpated from the general area by fire. The lower abundance of ants in the burn treatments resulted in significantly less pileated woodpecker foraging activity.

In our study, the average number of snags was about the same in both fuel reduction treatments (fig. 1), although the higher number of recently dead snags in the prescribed burn treatments suggests that some live trees were killed and some existing snags were consumed. Trees continued to die more than a year after the burn owing to woodborer and bark beetle activity in them. The recent mortality of live trees created foraging habitat for *Picoides* woodpeckers because woodborers were present in the dead and scorched live trees in the 2003 prescribed burn treatments. We did not detect woodborers still present in the 2001 or 2002 prescribed burn treatments.

Prescribed burning in this study area did not allow the degree of control in retaining coarse woody debris that the mechanical reduction treatment allowed. In one of the 43 burned stands on Starkey, more than 10 pileated woodpecker nest trees located in a previous study (Bull and Holthausen 1993) were consumed by the fire in an 18-ha area. A snag used by great gray owls (*Strix nebulosa*) for nesting was burned in one of the stands used in this study. Although some combustible fuel was removed from the base of some snags as recommended by Conner and Locke (1979), the nest trees burned. Weather conditions can change rapidly and reduce the amount of control that managers have over a prescribed burn and result in the loss of logs, snags, and live trees that were designated for retention.

The results of this pilot study indicate that it is feasible to determine the effect of fuel reductions on pileated woodpecker foraging in relatively small plots. It would be beneficial to conduct a similar study over a much larger geographic area to determine if the results of this study are applicable on a landscape scale and over the long term. It would be beneficial to know when *Camponotus* and *Formica* colonize areas that had prescribed burn treatments. If maintaining biodiversity and management indicator species is an objective, it is important to know the consequences of fuel reduction treatments on specific wildlife species, particularly those that depend on the coarse woody debris that is removed during these treatments. Alternative measures may be available to ensure the retention of structures that specific wildlife species depend on and allow managers to meet multiple objectives simultaneously. Additional research is needed on the long-term effects of fuel reduction treatments on specific wildlife species, amounts and kinds of fuels to retain for wildlife, and additional measures that can be taken to protect specific habitat structures.

Acknowledgments

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English Equivalents

When you know:	Multiply by:	To find:
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Kilometers (km)	.62	Miles
Hectares (ha)	2.47	Acres
Kilograms per hectare (kg/ha)	.89	Pounds per acre
Square meters per hectare (m^2/ha)	4.37	Square feet per acre
Celsius (°C)	1.8 + 32	Fahrenheit

Changes in Distribution and Abundance of the Loggerhead Shrike

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Abstract: *The Loggerhead Shrike* (Lanius ludovicianus) *was once widely distributed and common over most of North America, occupying an exclusive breeding range with no other shrikes. Although it occurs in a wide variety of plant associations, this shrike is generally found in landscapes characterized by widely spaced shrubs and low trees interspersed with short grasses, forbs, and bare ground, habitats which include deserts, scrub lands, savannas, and some agricultural settings. The Loggerhead Shrike seems to have been always most abundant in the southern and western portions of its range, with high breeding densities from Florida across the gulf states to Texas and throughout the arid regions of the West. A northeastward expansion in range occurred in the late 1800s in association with deforestation and agriculture. A similar north central expansion occurred in the 1900s with agricultural development of the northern Great Plains and aspen parklands. Contraction and decrease in numbers have been noted in parts of its range since the 1940s, concurrent with the regrowth of forests, loss of pasturelands, and intensive row-crop agriculture. More recently, Christmas Bird Count data and Breeding Bird Survey data have revealed an overall downward trend across the continent at least since 1966, although numbers are stable or increasing in some locations. Field studies generally implicate alterations in habitat structure and loss of habitat as factors responsible for changes in breeding distribution and overall abundance. Nevertheless, considering its entire distribution in North America and its historical expansions and contractions of range associated with habitat changes, the Loggerhead Shrike does not appear threatened with extinction as a species. We favor a hands-on approach to management of the critically endangered subspecies,* L. l. mearnsi *and* L. l. migrans, *however, and recommend extensive preservation of the natural scrub desert, shrub-steppe, western oak savanna, and southern savanna vegetation types, which appear to be optimal, core habitats for this species, as well as fostering land-use practices that favor shrikes in agricultural and suburban landscapes.*

Cambios en la Distribución y Abundancia del Verduguillo

Resumen: *En el pasado, el verduguillo* (Lanius ludovicianus) *era común y estaba ampliamente distribuido en casi toda América del Norte, ocupando un rango de reproducción exclusivo. Aunque ocurre en una amplia variedad de asociaciones vegetales, esta especie se encuentra generalmente en paisajes caracterizados por arbustos y árboles bajos ampliamente espaciados separados por pastos cortos, hierbas y suelo desnudo: desiertos, matorrales, sabanas y algunos campos agrícolas. El verduguillo parece haber sido más abundante en las porciones sur y occidental de su rango, con altas densidades reproductivas desde Florida hasta Texas y las regiones áridas del Oeste. A finales del siglo XIX occurió una expansión hacia el noreste asociada con deforestación y agricultura. En este siglo ocurrió una expansión similar hacia el norte con el desarrollo agrícola de las planicies del norte y de los bosques de álamo. Desde la década de 1940 se ha notado la contracción y reducción de números en partes de su rango concurrentes con la regeneración de bosques, la pérdida de pastizales y la agricultura intensiva. Más recientemente, datos de censos de Navidad y de Aves Anidantes revelan la disminución como una tendencia general, a pesar de que los números son estables o crecientes en algunas localidades, por lo menos desde 1966. En estudios de campo, los cambios de distributión y abundancia generalmente son atribuidos a alteraciones o pérdida de hábitat. A pesar de ello, considerando su rango de dis-*

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Other frequently mentioned sources of mortality that appear to have increased in recent decades are collision with vehicles (Robertson 1930; Miller 1931; Cadman 1985; Luukkonen 1987; Gawlik 1988; Flickinger 1995) and predation by domestic/feral cats (Gawlik & Bildstein 1990; Scott & Morrison 1990). Both factors are enhanced by the shrike's attraction to roadways. Fence lines and roadsides are natural avenues for cats and other predators, and nest losses in some areas are higher along roads and fences (Yosef 1994).

Discussion

Much of the recent contraction in range and consequent decrease in shrike numbers can be accounted for by successional changes in habitats (e.g., reforestation of abandoned agricultural lands) or by outright habitat loss from various human activities (e.g., intensive rowcrop agriculture, conversion of sage-steppe to exotic grassland or cropland, urbanization, and suburbanization). These are rather easily visualized processes, if not readily manageable.

What is not so easily understood and consequently is perhaps more troubling is the continuing disappearance of shrikes in regions where apparently suitable habitat remains, as well as the patchy distribution of shrike populations in some extensive habitats (e.g., sagebrush-steppes; Woods 1994). Peakall (1995) has chronicled a parallel case involving the total extirpation of the Red-backed Shrike (*Lanius collurio*) in Great Britain. In considering these problems we should not rule out the possibility that there are subtle features of habitat to which shrikes respond but that human perception of "suitable habitat" fails to detect (Prescott & Collister 1993; Cuddy 1995); in other words, truly suitable habitat may actually be patchy. Graber et al. (1974) refer to a two-phase decline in Illinois—an initial, gradual reduction related to change in habitats followed by a second, more rapid decline seemingly unrelated to habitat. Possibly, as suitable habitat becomes fragmented, factors related to site fidelity and dispersal in shrikes may further exacerbate strains placed on already declining populations (Lande 1987; Cade 1995). Shrikes are not strongly philopatric (Woods 1994), and this could lead to reduced success in locating a mate when breeding aggregations are widely separated and scarce.

On the other hand, shrike social and territorial behavior may require greater contact with neighboring shrikes than is provided when breeding habitat is overly disjunct and one or only a few pairs can settle in a given area. Darling (1952) and other students of avian territoriality (reviewed by Stamps 1988) noted the tendency for territorial birds to attract conspecific individuals to their neighborhoods, with the result that avian territories are often clustered even though apparently suitable, unoccupied habitat exists nearby. Conspecific attraction and territorial aggregation have been confirmed by several recent studies of birds (Mayfield 1960; Alatalo et al. 1982; Herremans 1993) and of lizards and other animals (Stamps 1988).

Many Loggerhead Shrikes nesting in the sage shrub-steppes of southern Idaho fit this pattern, too (Woods 1994). It is not clear, however, whether this pattern of breeding dispersion represents a normal condition for viable populations or is instead a reflection of deteriorating populations, although we lean toward the former view for the following reasons. It may be that nesting shrikes seldom reach the carrying capacity of their breeding habitats—even in the best of circumstances—because of normally high winter mortality, a condition that appears to apply to the Northern Shrike in North America (Cade 1967, 1995). In this case one would expect to see behavioral adaptations that promote conspecific attraction and territorial aggregation. In Europe Great Gray Shrikes (*Lanius excubitor*) assemble in "group-meetings" at the margin of breeding territories in late winter and early spring during territorial establishment and pairing (Schon 1995). These meetings involve frequent changes in location of individuals and partners (typically four to six birds), loud calling, wing and tail-flashing displays, and high flights across territorial borders. We have seen similar behavior among Loggerhead Shrikes at our Wilson Creek study site in southern Idaho (Woods & Cade unpublished notes), and we suggest that this behavior both attracts potential settlers to the neighborhood and functions as a way for shrikes to locate high-quality nesting habitat by cuing on early, presumably more experienced arrivers, in a manner similar to Stamps's (1987) proposal for *Anolis* lizards. These group-meetings may also have evolved as a way to counter the disadvantage conferred by weak philopatry in finding a mate.

Consideration of these and similar behavioral and ecological variables in the design of future studies of breeding shrikes should aid our understanding of the recent declines. Such information could also lead to new ways of managing metapopulations in fragmented landscapes (Smith & Peacock 1990), and in large, continuous tracts of habitat.

Is the Loggerhead Shrike a globally threatened species? It has been extirpated from some of its range and is currently in decline in other areas. However, when it is viewed over its entire distribution in North America, and when its historical expansions and contractions of range associated with habitat changes are considered, the Loggerhead Shrike does not appear to be threatened with foreseeable extinction as a species. In much of the portion of its range where populations are disappearing, the shrike seems to be in a category with several declining bird species that occupy seral stages of vegetation, and noted declines may represent a progression in avian communities in response to changing habitats (Askins 1993; Hagan

1993; Litvaitis 1993; Trauger & Bocetti 1993). Furthermore, over nearly 30% of its breeding range shrike populations may be stable or increasing according to our analysis of the BBS map produced by Sauer et al. (1995).

Even so, we favor a hands-on approach to management of the species, especially for distinct populations or subspecies that meet the criteria of "critically endangered" (IUCN 1994, based on Mace & Lande 1991 and subsequent modifications), for instance, *L. l. mearnsi* on San Clemente Island (Morrison et al. 1995) and the eastern *L. l. migrans*. To the degree that these subspecies represent unique genetic adaptations to local or regional environments, priorities should be set for active management. For example, shrikes can be bred in captivity (Cade 1992; Azua & Lieberman 1995). Restoration through the release of captive-bred birds and other methods of augmenting numbers in conjunction with aggressive removal of goats and predator control around shrike nests are proving feasible on San Clemente Island (Morrison et al. 1995). A similar program is urgently needed to save the rapidly disappearing *migrans* population in Ontario, the last holdout of the eastern migrant shrikes. The International Shrike Working Group and the IUCN/SSC Re-introduction Specialist Group have urged Canadian authorities to establish a captive population while a significant genetic representation of this stock still exists.

In addition, acclimating shrikes to some suburban environments is possible. Shrikes have adjusted to suburban areas of southern California, Arizona, Texas, Florida, and some other places, but we have observed few shrikes nesting around human habitations or farms in Idaho. Sage-steppe shrikes appear to be less tolerant of close human association, but through the release of hand-reared birds it could be possible to establish shrikes in suburban and farmstead environments, which have rapidly encroached into the sagebrush landscape in the past several decades, in much the same way that captive-bred and released Peregrine Falcons (*Falco peregrinus*) have become established in urban environments (Cade et al. 1996).

In considering habitat management for shrikes, Gawlik and Bildstein (1993:356) note that reduced pasturelands may now be limiting shrikes in many parts of their range, and they call for management to include ". . . a patchwork of short disturbed grassland and sparsely vegetated cropland at the scale of individual territories." They suggest this habitat mosaic could be achieved either by interspersing small monocultures of rowcrops and pastureland to create "among-field diversity," as occurs in South Carolina, or by managing large blocks of pastureland to provide a "within-field diversity" of bare areas, short grass, and taller vegetation. This habitat interspersion can be achieved by common land-use techniques, grazing, burning, plowing, and mowing. It should be noted that fire can have both beneficial and harmful impacts on shrike habitats. Fire often creates or maintains suitable grasslands; however, it has been one of the principal factors involved in the conversion of sage shrub-steppe, an optimum habitat for shrikes, to exotic cheatgrass (*Bromus tectorum*) stands devoid of shrubs (Knick & Rotenberry 1995) and shrikes.

"Short grass" is, of course, a relative term, which may be more appropriate to eastern and tall-grass prairie environments. In western, short-grass prairie shrikes are associated more with taller grass stands ≥20 cm (Prescott & Collister 1993), somewhat deceptively termed "tall grass" by these authors. ("Short grass" had an average height of only 15.8 cm.) Furthermore, Chavez-Ramirez et al. (1994) caution that management practices that favor shrikes in agricultural landscapes may be inappropriate for application to natural grasslands because shrikes use these habitats in different ways, a point also emphasized by Prescott and Collister (1993).

Shrikes occurring along roadways are frequently killed by automobiles (Flickinger 1995), and attention also needs to be given to management for appropriate habitat away from heavily traveled roads to help alleviate this problem. Yosef (1996) has summarized all these and other possible management techniques, including fencing old shelterbelts and other shrubby formations against destruction by livestock, increasing the number of hunting perches on deficient territories, and reclaiming strip mines by keeping them in early successional stages instead of allowing them to revert to forest.

To these recommendations we would also add extensive preservation of the natural scrub desert, shrub-steppe, and western and southern savanna vegetation types, which appear to be optimal, core habitats for this species, from which shrike populations can disperse into appropriate seral habitats as they become available. If increased effort can be devoted to maintaining suitable habitats within an overall ecosystem approach to landscape management, it is reasonable to believe that the Loggerhead Shrike will continue to exist as an interesting, viable, and functional component of the North American biota, even though its distribution and abundance may change locally and regionally, as they no doubt have done repeatedly since the likely origin of the species sometime in the Pleistocene Epoch.

Acknowledgments

We thank S. Robinson and two anonymous reviewers for helpful comments on our manuscript and their suggestions for including additional information. Our ideas for this review developed from discussions with colleagues at the First International Shrike Symposium at the Archbold Biological Station, Lake Placid, Florida, the published proceedings of which we recommend to the reader for fuller accounts and documentation of topics summarized in our review (Yosef & Lohrer 1995*a*).

Cal Fire burns next to Bald Eagle nest, eaglets die (/stories/2022/4/15/2092201/-Cal-Fire-burns-next-to-Bald-Eagle-nest-eaglets-die)

Defiance Canyon Raptor Rescue (/users/defiance canyon raptor rescue)
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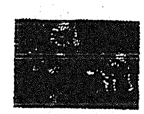
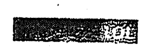
Liz Cheney: The House Jan. 6 committee has uncovered a 'broad,' 'well-organized' conspiracy
by Aldous J Pennyfarthing
★ 262 188



One More Dead Russian General in Ukraine
by GaryNaham
★ 156 92

Tweets of the Week May 29-Jun



Bald eagle chick, dead in nest tree after Cal Fire control burn next to the nest in 2021. Cal Fire has not committed to stop burning by the nest this year. Who would think it would be a problem for public agencies to adhere to laws that protect wildlife?

Climate change impacts have been worsening for years, raising temperatures and exacerbating fire danger in California and the world. In many cases though, trees and other plants are being treated as enemies to be annihilated, rather than as the ecosystems that enable life on earth to exist.

In California, both Cal Fire and PG&E are being given exemptions from any environmental review for their “fuel reduction” or “vegetation management” programs. In this time when thousands of scientists worldwide are screaming and waving red flags about biodiversity and climate catastrophe, the impacts of these projects are being ignored, particularly to wildlife and habitat.

RSS (/user/Defiance Canyon Raptor Rescue/rss.xml)

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TAGS

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#Environment (/tags/Environment)

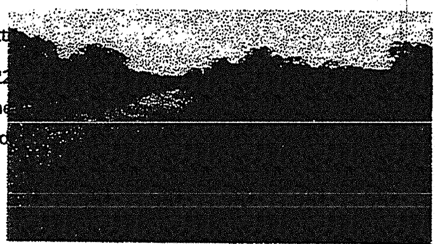
#Fire (/tags/Fire)

#Nature (/tags/Nature)

#Raptors (/tags/Raptors)

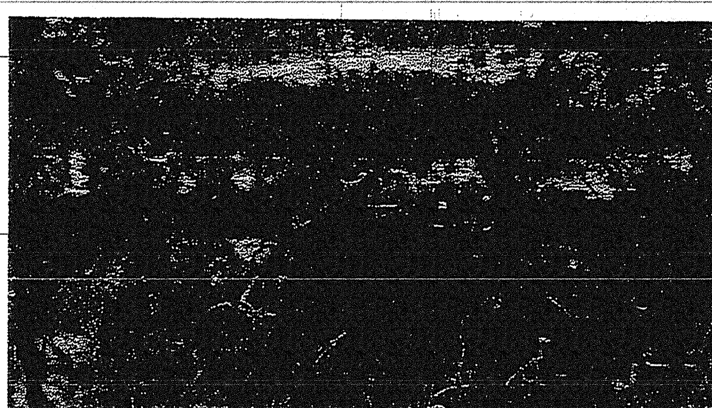
#Wildlife (/tags/Wildlife)

What has happened over the past several years to a bald eagle nest east of Red Bluff, California illustrates what is occurring because of these exemptions from any oversight. The fear of fire is being exploited to the detriment of the natural world, rather than substantive actions being implemented to reduce the emissions which are causing climate disaster.



Hwy 36, east of Red Bluff. The eagle nest is to the right (south side). This is the roadside Cal Fire has burned in 2020 and 2021 when the eagle nest was occupied. There is little reason to burn here, and many reasons not to.

The eagles' nest to the south of the highway, circled in red.



Parent eagle with young eaglet in front (little grey head) in nest tree, April 2022.

Local residents have been watching this nest since 2020. A photographer from Red Bluff was going out to the nest every day in 2021. At the end of May, the photographer saw a notice that there was going to be a control burn by the nest in a few days. She contacted a local eagle group, who called Cal Fire to tell them about the nest which was occupied by two young eaglets. The eagle group left a message and received a message back from Cal Fire saying their biologist said it was fine to be burning near the nest. The eagle group called back to get the biologist's name, but received no answer then or later. It wasn't "fine".

The burn was done on June 1st. This nest is approximately 100 feet down a ravine from the highway. The eaglets were probably only 6-7 weeks old, 4 or 5 weeks from being able to fly.

The photographer was standing next to the nest during the burn and taking photographs. The Cal Fire people were slightly to the east of the nest. The smoke and flames can be seen on the south side of the highway, on the same side as the nest.

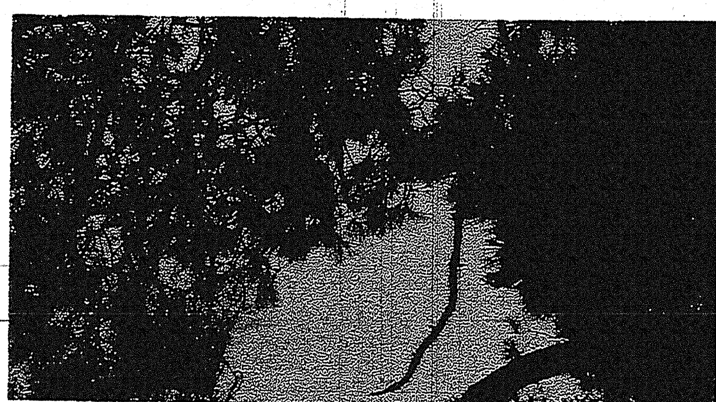
Cal Fire burning next to eagle nest, 2021. How much extra CO2 is being emitted by extra equipment use and burning unnecessarily?

The photographer went to check the nest a few days later and saw one adult perched above the nest, but could see no eaglets.

The next morning, the photographer took a photo which shows a dead eaglet hanging from the nest. The photographer contacted me (Marily Woodhouse from Defiance Canyon Raptor Rescue). We went to search for the other eaglet, in the hope it was still alive.

Dead eaglet hanging from the nest.

Both the adults were at the top of the tree, above the nest. I went down the ravine to the nest tree to walk around beneath it to search for the second eaglet and the adults stayed in the tree top. I walked to the southeast side of the tree and looked up and saw the other eaglet hanging dead in the tree, below the nest about 10'.



The second eaglet, dead below the nest.

We reported the deaths to US Fish and Wildlife and CA Department of Fish and Wildlife, but never received any notification of any action taken.

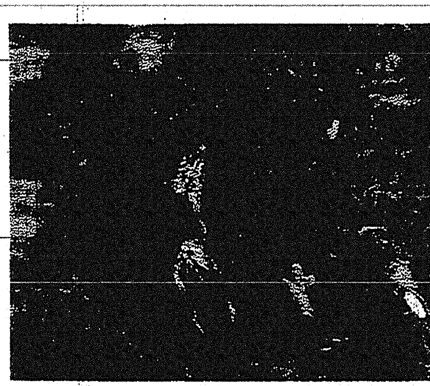
A State Wildlife Health Lab biologist wrote to us later that:

"A bird's respiratory system is more sensitive to toxins, including smoke, than a mammal's respiratory system. This is because birds have a higher oxygen demand than mammals and a bird's lungs are 10 times more efficient at capturing oxygen. The rapid efficiency of gas exchange in bird lungs makes them more susceptible to inhaled toxic agents, including smoke. Inhaled toxins, such as smoke, can cause irritation and damage the respiratory system. It also can

compromise the immune system, making the bird more susceptible to infections. This is especially true in young birds in the nest that are unable to escape the smoke. Smoke inhalation toxicity in birds is caused by irritant gases (aldehydes, hydrogen chloride, and sulfur dioxide), particulate matter, and nonirritant gases (carbon monoxide, carbon dioxide, and hydrogen cyanide) released by combustion."

There was a burn done next to the Dales Station nest in 2020 also. I was called upon to rescue an eaglet who got out of the nest before he could fly that year. It was several days before the burn was done that year, so he was away from the nest when the burn occurred. His sister was still in the nest during the burn. I received a call from Dales Station, less than a mile from the nest, in August, 2020 about an eaglet who had been on the ground for 3 days, standing next to a shallow pool of Paynes Creek. My determination was that it was the female from the nest. She was open-mouthed breathing with a raspy noise. She died a few hours after she was caught and transported. The Wildlife Lab report said: "This was a juvenile female in poor nutritional condition with no fat reserves and minimal pectoral muscle development. Internally, there was evidence of an extensive infection. The visible infection resembled avian tuberculosis which is caused by the bacterium Mycobacterium avium. It's widespread in the environment in soil and dust and is usually an opportunistic infection. Depending on where the lesions are in the bird, gives an idea of how it entered the body. The lesions in this bird were primarily in the air sacs suggesting it was inhaled."

The male who had been in care was released in 2020. A first year eagle was seen back at the nest in 2021. Judging by his and the adults' behavior, it was the male who was in care away from the nest during the burn in 2020.



2020 bald eaglet being released. He was away from the nest in rehab care during Cal Fire's control burn in 2020, which probably saved his life.

I had occasion to contact Cal Fire in February 2022 about another issue. I had just been informed that the Dales Station bald eagle nest was occupied, so mentioned it in the hope of preventing another burn next to the nest. Cal Fire and its employees are public servants. It is their job to uphold state and federal laws, which include protection of wildlife, but the answer from a Cal Fire employee contained only dismissive, condescending remarks, clearly refusing to take steps to ensure any protections were implemented.

Many letters, calls, and emails have ensued since February (most unanswered). I made maps from Cal Fire's own fire database showing how rarely the area around the nest has burned. The ravine area there is extremely rocky and is grazed by cattle. And then there are the State and Federal laws that protect nesting birds. Still, Cal Fire will not commit to refrain from burning by the nest again this year.

Last week a biologist from a PG&E contractor company working in Greenville (a town that burned in the Dixie fire last year) called Raptor Rescue because they wanted us to take eggs from a nest in a tree they wanted to cut down. I explained the multitude of reasons that was a bad idea, along with it being illegal for them to do. The man said "We have an exemption". How many nesting birds are being destroyed in California due to these stupid, thoughtless exemptions and the complete lack of oversight which is occurring?

There have got to be protections enforced. Apparently that won't happen without widespread public outrage.

Here are some state employees to contact if you will help tell them there is a problem with their practices:

George Morris, Cal Fire Northern Region Unit Chief (530) 224-2445 (They would not give out his email address)

Dave Russell, Cal Fire Tehama/Glenn Unit Chief (530) 528-5199 dave.russell@fire.ca.gov

Tina Bartlett, Regional Manager CDFW, (530) 225-2300 tina.bartlett@wildlife.ca.gov

People often focus on individual species, but we believe every species is important, whether it is on a man-made list or not. Habitat fragmentation and loss have significant impacts on wildlife. Defiance Canyon Raptor Rescue works to rescue, rehabilitate, and return raptors to their wild lives, along with our work to protect watersheds and forests of California.

www.thebattlecreekalliance.org (http://www.thebattlecreekalliance.org/)

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Decline of the North American avifauna

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Species extinctions have defined the global biodiversity crisis, but extinction begins with loss in abundance of individuals that can result in compositional and functional changes of ecosystems. Using multiple and independent monitoring networks, we report population losses across much of the North American avifauna over 48 years, including once common species and from most biomes. Integration of range-wide population trajectories and size estimates indicates a net loss approaching 3 billion birds, or 29% of 1970 abundance. A continent-wide weather radar network also reveals a similarly steep decline in biomass passage of migrating birds over a recent 10-year period. This loss of bird abundance signals an urgent need to address threats to avert future avifaunal collapse and associated loss of ecosystem integrity, function and services.

Slowing the loss of biodiversity is one of the defining environmental challenges of the 21st century (*1–5*). Habitat loss, climate change, unregulated harvest, and other forms of human-caused mortality (*6, 7*) have contributed to a thousand-fold increase in global extinctions in the Anthropocene compared to the presumed prehuman background rate, with profound effects on ecosystem functioning and services (*8*). The overwhelming focus on species extinctions, however, has underestimated the extent and consequences of biotic change, by ignoring the loss of abundance within still-common species and in aggregate across large species assemblages (*2, 9*). Declines in abundance can degrade ecosystem integrity, reducing vital ecological, evolutionary, economic, and social services that organisms provide to their environment (*8, 10–15*). Given the current pace of global environmental change, quantifying change in species abundances is essential to assess ecosystem impacts. Evaluating the magnitude of declines requires effective long-term monitoring of population sizes and trends, data which are rarely available for most taxa.

Birds are excellent indicators of environmental health and ecosystem integrity (*16, 17*), and our ability to monitor many species over vast spatial scales far exceeds that of any other animal group. We evaluated population change for 529 species of birds in the continental United States and Canada (76% of breeding species), drawing from multiple standardized bird-monitoring datasets, some of which provide close to fifty years of population data. We integrated range-wide estimates of population size and 48-year population trajectories, along with their associated uncertainty, to quantify net change in numbers of birds across the avifauna over recent decades (*18*). We also used a network 143 weather radars (NEXRAD) across the contiguous U.S. to estimate long-term changes in nocturnal migratory passage of avian biomass through the airspace in spring from 2007 to 2017. The continuous operation and broad coverage of NEXRAD provide an automated and standardised monitoring tool with unrivaled temporal and spatial extent (*19*). Radar measures cumulative passage across all nocturnally migrating species, many of which breed in areas north of the contiguous U.S. that are poorly monitored by avian surveys. Radar thus expands the area and the proportion of the migratory avifauna that is sampled relative to ground surveys.

Results from long-term surveys, accounting for both increasing and declining species, reveal a net loss in total abundance of 2.9 billion (95% CI = 2.7-3.1 billion) birds across almost all biomes, a reduction of 29% (95% CI = 27-30%) since 1970 (Fig. 1 and Table 1). Analysis of NEXRAD data indicate a similarly steep decline in nocturnal passage of migratory biomass, a reduction of 13.6 ± 9.1% since 2007 (Fig. 2A). Reduction in biomass passage occurred across the eastern U.S. (Fig. 2, C and D), where migration is dominated by large numbers of temperate- and boreal-breeding songbirds; we observed no consistent trend in the Central or Pacific flyway regions (Fig. 2, B to D, and table S5). Two completely different and independent monitoring techniques thus signal major population loss across the continental avifauna.

Species exhibiting declines (57%, 303/529) based on long-term survey data span diverse ecological and taxonomic groups. Across breeding biomes, grassland birds showed the largest magnitude of total population loss since 1970—more than 700 million breeding individuals across 31 species— and the largest proportional loss (53%); 74% of grassland species are declining. (Fig. 1 and Table 1). All forest biomes experienced large avian loss, with a cumulative reduction of more than 1 billion birds. Wetland birds represent the only biome to show an overall net gain in numbers (13%), led by a 56% increase in waterfowl populations (Fig. 3 and Table 1). Surprisingly, we also found a large net loss (63%) across 10 introduced species (Fig. 3, D and E, and Table 1).

A total of 419 native migratory species experienced a net loss of 2.5 billion individuals, whereas 100 native resident species showed a small net increase (26 million). Species overwintering in temperate regions experienced the largest net reduction in abundance (1.4 billion), but proportional loss was greatest among species overwintering in coastal regions (42%), southwestern aridlands (42%), and South America (40%) (Table 1 and fig. S1). Shorebirds, most of which migrate long distances to winter along coasts throughout the hemisphere, are experiencing consistent, steep population loss (37%).

More than 90% of the total cumulative loss can be attributed to 12 bird families (Fig. 3A), including sparrows, warblers, blackbirds, and finches. Of 67 bird families surveyed, 38 showed a net loss in total abundance, whereas 29 showed gains (Fig. 3B), indicating recent changes in avifaunal composition (table S2). While not optimized for species-level analysis, our model indicates 19 widespread and abundant landbirds (including 2 introduced species) each experienced population reductions of >50 million birds (data S1). Abundant species also contribute strongly to the migratory passage detected by radar (*19*), and radar-derived trends provide a fully independent estimate of widespread declines of migratory birds.

Our study documents a long-developing but overlooked biodiversity crisis in North America—the cumulative loss of nearly 3 billion birds across the avifauna. Population loss is not restricted to rare and threatened species, but includes many widespread and common species that may be disproportionately influential components of food webs and ecosystem function. Furthermore, losses among habitat generalists and even introduced species indicate that declining species are not replaced by species that fare well in human-altered landscapes. Increases among waterfowl and a few other groups (e.g., raptors recovering after the banning of DDT) are insufficient to offset large losses among abundant species (Fig. 3). Importantly, our population loss estimates are conservative since we estimated loss only in breeding populations. The total loss and impact on communities and ecosystems could be even higher outside the breeding season if we consider the amplifying effect of "missing" reproductive output from these lost breeders.

Extinction of the Passenger Pigeon (*Ectopistes migratorius*), once likely the most numerous bird on the planet, provides a poignant reminder that even abundant species can go extinct rapidly. Systematic monitoring and attention paid to population declines could have alerted society to its pending extinction (*20*). Today, monitoring data suggest that avian declines will likely continue without targeted conservation action, triggering additional endangered species listings at tremendous financial and social cost. Moreover, because birds provide numerous benefits to ecosystems (e.g., seed dispersal, pollination, pest control) and economies (47 million people spend 9.3 billion U.S. dollars per year through bird-related activities in the U.S. (*21*)), their population reductions and possible extinctions will have severe direct and indirect consequences (*10, 22*). Population declines can be reversed, as evidenced by the remarkable recovery of waterfowl populations under adaptive harvest management (*23*) and the associated allocation of billions of dollars devoted to wetland protection and restoration, providing a model for proactive conservation in other widespread native habitats such as grasslands.

Steep declines in North American birds parallel patterns of avian declines emerging globally (*14, 15, 22, 24*). In particular, depletion of native grassland bird populations in North America, driven by habitat loss and more toxic pesticide use in both breeding and wintering areas (*25*), mirrors loss of farmland birds throughout Europe and elsewhere (*15*). Even declines among introduced species match similar declines within these same species' native ranges (*26*). Agricultural intensification and urbanization have been similarly linked to declines in insect diversity and biomass (*27*), with cascading impacts on birds and other consumers (*24, 28, 29*). Given that birds are one of the best monitored animal groups, birds may also represent the tip of the iceberg, indicating similar or greater losses in other taxonomic groups (*28, 30*).

Pervasiveness of avian loss across biomes and bird families suggests multiple and interacting threats. Isolating spatio-temporal limiting factors for individual species and populations will require additional study, however, since migratory species with complex life histories are in contact with many threats throughout their annual cycles. A focus on breeding season biology hampers our ability to understand how seasonal interactions drive population change (*31*), although recent continent-wide analyses affirm the importance of events during the non-breeding season (*19, 32*). Targeted research to identify limiting factors must be coupled with effective policies and societal change that emphasize reducing threats to breeding and non-breeding habitats and minimizing avoidable anthropogenic mortality year-round. Endangered species legislation and international treaties, such as

Table 1. Net change in abundance across the North American avifauna, 1970-2017. Species are grouped into native and introduced species, management groups (landbirds, shorebirds, waterbirds, waterfowl), major breeding biomes, and nonbreeding biomes (see data S1 in (*18*) for assignments and definitions of groups and biomes). Net change in abundance is expressed in millions of breeding individuals, with upper and lower 95% credible intervals (CI) shown. Percentage of species in each group with negative trend trajectories are also noted. Values in bold indicate declines and loss; those in italics indicate gains.

Species Group	Number of Species	Net Abundance Change (Millions) & 95% CI			Percent Change & 95% CIs			Proportion Species in Decline
		Change	LC95	UC95	Change	LC95	UC95	
Species Summary								
All N. Am. Species	529	**-2,911.9**	-3,097.5	-2,732.9	**-28.8%**	-30.2%	-27.3%	57.3%
All Native Species	519	**-2,521.0**	-2,698.5	-2,347.6	**-26.5%**	-28.0%	-24.9%	57.4%
Introduced Species	10	**-391.6**	-442.3	-336.6	**-62.9%**	-66.5%	-56.4%	50.0%
Native Migratory Species	419	**-2,547.7**	-2,723.7	-2,374.5	**-28.3%**	-29.8%	-26.7%	58.2%
Native Resident Species	100	*26.3*	7.3	46.9	*5.3%*	1.4%	9.6%	54.0%
Landbirds	357	**-2,516.5**	-2,692.2	-2,346.0	**-27.1%**	-28.6%	-25.5%	58.8%
Shorebirds	44	**-17.1**	-21.8	-12.6	**-37.4%**	-45.0%	-28.8%	68.2%
Waterbirds	77	**-22.5**	-37.8	-6.3	**-21.5%**	-33.1%	-6.2%	51.9%
Waterfowl	41	*34.8*	24.5	48.3	*56.0%*	37.9%	79.4%	43.9%
Aerial Insectivores	26	**-156.8**	-183.8	-127.0	**-31.8%**	-36.4%	-26.1%	73.1%
Breeding Biome								
Grassland	31	**-717.5**	-763.9	-673.3	**-53.3%**	-55.1%	-51.5%	74.2%
Boreal forest	34	**-500.7**	-627.1	-381.0	**-33.1%**	-38.9%	-26.9%	50.0%
Forest Generalist	40	**-482.2**	-552.5	-413.4	**-18.1%**	-20.4%	-15.8%	40.0%
Habitat Generalist	38	**-417.3**	-462.1	-371.3	**-23.1%**	-25.4%	-20.7%	60.5%
Eastern Forest	63	**-166.7**	-185.8	-147.7	**-17.4%**	-19.2%	-15.6%	63.5%
Western forest	67	**-139.7**	-163.8	-116.1	**-29.5%**	-32.8%	-26.0%	64.2%
Arctic Tundra	51	**-79.9**	-131.2	-0.7	**-23.4%**	-37.5%	-0.2%	56.5%
Aridlands	62	**-35.6**	-49.7	-17.0	**-17.0%**	-23.0%	-8.1%	56.5%
Coasts	38	-6.1	-18.9	8.5	-15.0%	-39.4%	21.9%	50.0%
Wetlands	95	*20.6*	8.3	35.3	*13.0%*	5.1%	23.0%	47.4%
Nonbreeding Biome								
Temperate North America	192	**-1,413.0**	-1,521.5	-1,292.3	**-27.4%**	-29.3%	-25.3%	55.2%
South America	41	**-537.4**	-651.1	-432.6	**-40.1%**	-45.2%	-34.6%	75.6%
Southwestern Aridlands	50	**-238.1**	-261.2	-215.6	**-41.9%**	-44.5%	-39.2%	74.0%
Mexico-Central America	76	**-155.3**	-187.8	-122.0	**-15.5%**	-18.3%	-12.6%	52.6%
Widespread Neotropical	22	**-126.0**	-171.2	-86.1	**-26.8%**	-33.4%	-19.3%	45.5%
Widespread	60	-31.6	-63.1	1.6	-3.7%	-7.4%	0.2%	43.3%
Marine	26	**-16.3**	-29.7	-1.2	**-30.8%**	-49.1%	-2.5%	61.5%
Coastal	44	**-11.0**	-14.9	-6.7	**-42.0%**	-51.8%	-26.7%	68.2%
Caribbean	8	-6.0	1.4	-15.7	12.1%	-2.8%	31.7%	25.0%

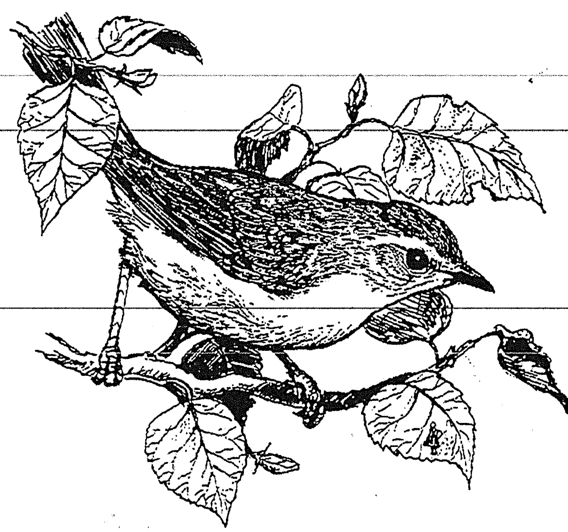
Neotropical Migrant Landbirds in the Northern Rockies and Great Plains

A Handbook for Conservation and Management

by

David S. Dobkin

The High Desert Museum
Bend, Oregon



United States
Department of Agriculture

Forest
Service

Northern Region

Schrasker B-84
GTT- B-115
Brewers 119
Sage-122

92

Cor-B-53

WISS B-45

Cal
Base-Soah-
Sousa- D-19.
Salvage -A-14 !!

CASSIN'S FINCH

Carpodacus cassinii
Fringillidae

Summer, Permanent, or Winter Resident

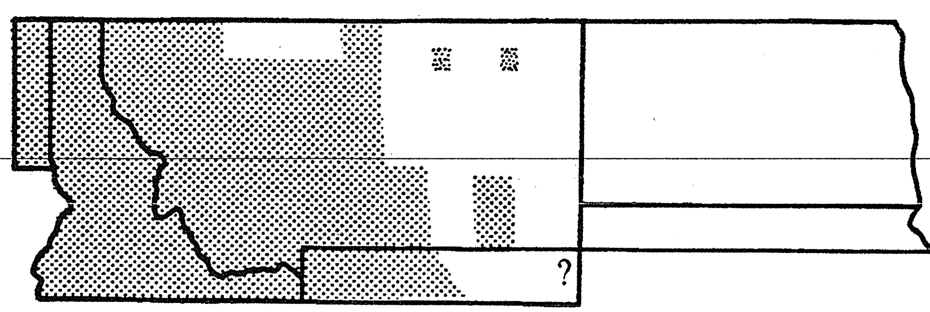
WINTERING AREA: 5

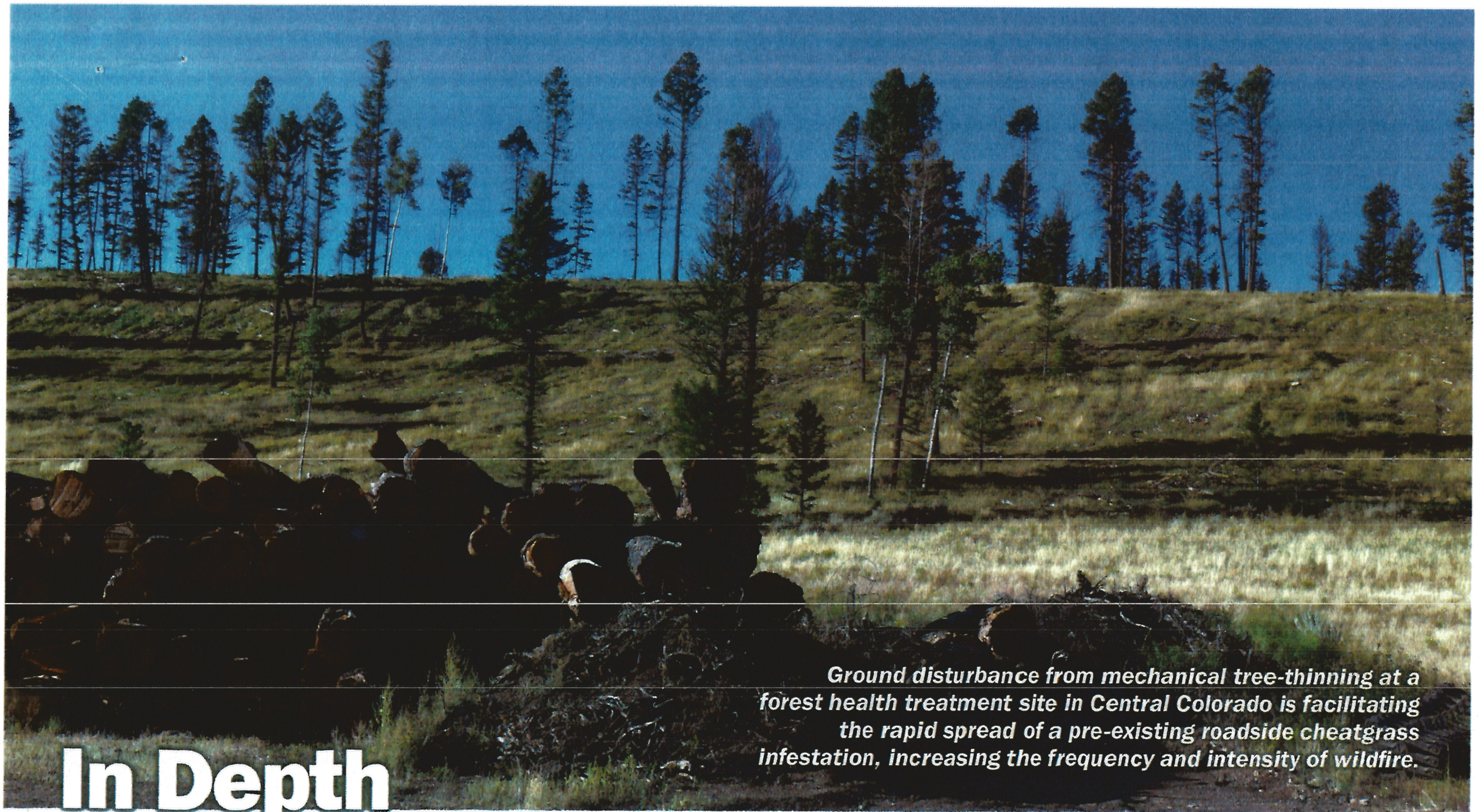
HABITAT REQUIREMENTS: Drier montane coniferous forests and woodlands, especially of ponderosa pine. Nests in coniferous trees.

FEEDING: Dines primarily on seeds of conifer trees, also takes insects, buds and berries. Forages on the ground and by gleaning from foliage in trees and shrubs.

STATUS AND MANAGEMENT: Numbers have been highly erratic in Idaho but appear to be increasing there; numbers have been more stable in Montana but appear to be declining slightly. In the West as a whole, numbers show a small but significant increasing trend. Prefers older rotation-age stands (Mannan and Maslow, 1984) and harvest units (Moore, 1992) over old growth. Cassin's Finch is a nomadic, semi-colonial breeder with resultant fluctuations in local population numbers.

FURTHER READING: Hejl et al., 1988; Mewaldt and King, 1985; Samson, 1976.





Ground disturbance from mechanical tree-thinning at a forest health treatment site in Central Colorado is facilitating the rapid spread of a pre-existing roadside cheatgrass infestation, increasing the frequency and intensity of wildfire.

In Depth

Cheatgrass

'One of the most significant ecological crises facing land managers in the arid West.'

A report published in January, *Cheatgrass Invasions: History, Causes, Consequences, and Solutions*, by Western Watershed Projects is the source of the above quote. Authored by Erik Molvar, Roger Rosentreter, Don Mansfield, and Greta Anderson, the new report draws on a century of research and data supporting a firm scientific consensus that this invasive species fuels a "livestock-cheatgrass-fire cycle" which "now prevails across much of the public lands of the western United States." As a result, those lands are now "susceptible to larger and more frequent fires."

Cheatgrass is the most widespread invasive weed in North America with millions of acres converted to cheatgrass monoculture and tens of millions of acres at risk of infestation. This annual grass from Eurasia was introduced to North America in the 1800s. Spread by railroads, vehicles, and livestock, it colonized lands that had been disturbed and degraded, mainly from overgrazing cattle. Molvar et al. provide a comprehensive review of scientific research on cheatgrass and evaluate solutions to restore healthy native ecosystems.

A significant proportion of the public lands at risk from cheatgrass-fueled fire is managed by the Forest Service, an agency currently spending billions of tax dollars to "mitigate wildfire risk" by cutting down trees. These logging projects don't address readily combustible fine fuels like cheatgrass, even though the risk is well-documented. The Boy Scouts

> ***Today the honey-colored hills that flank the northwestern mountains derive their hue not from the rich and useful bunchgrass and wheatgrass which once covered them, but from the inferior cheat.... The cause of the substitution is overgrazing. When the too-great herds and flocks chewed and trampled the hide off the foothills, something had to cover the raw eroding earth. Cheat did.***
>
> — *A Sand County Almanac*, Aldo Leopold, 1949

understand fine fuels, which they call tinder: "Thin, dry material that ignites instantly with a match. It's the basis of every fire. Examples include dead, dry grasses...."

Cheatgrass produces two crops per year, providing dead, dry grasses in summer and fall. The spring crop of cheatgrass dies off by early summer, leaving "the basis of every fire" available for easy ignition at the height of fire season. According to *Cheatgrass and Wildfire* (Colorado State University Extension) "A typical cheatgrass fire on flat terrain with wind speeds of 20 miles per hour may generate flame lengths up to eight feet in height," significantly putting cheatgrass in the category of "ladder fuel." Increase the wind speed, and a cheatgrass fire becomes unstoppable — like the million-acre grass fire that recently burned in Texas.

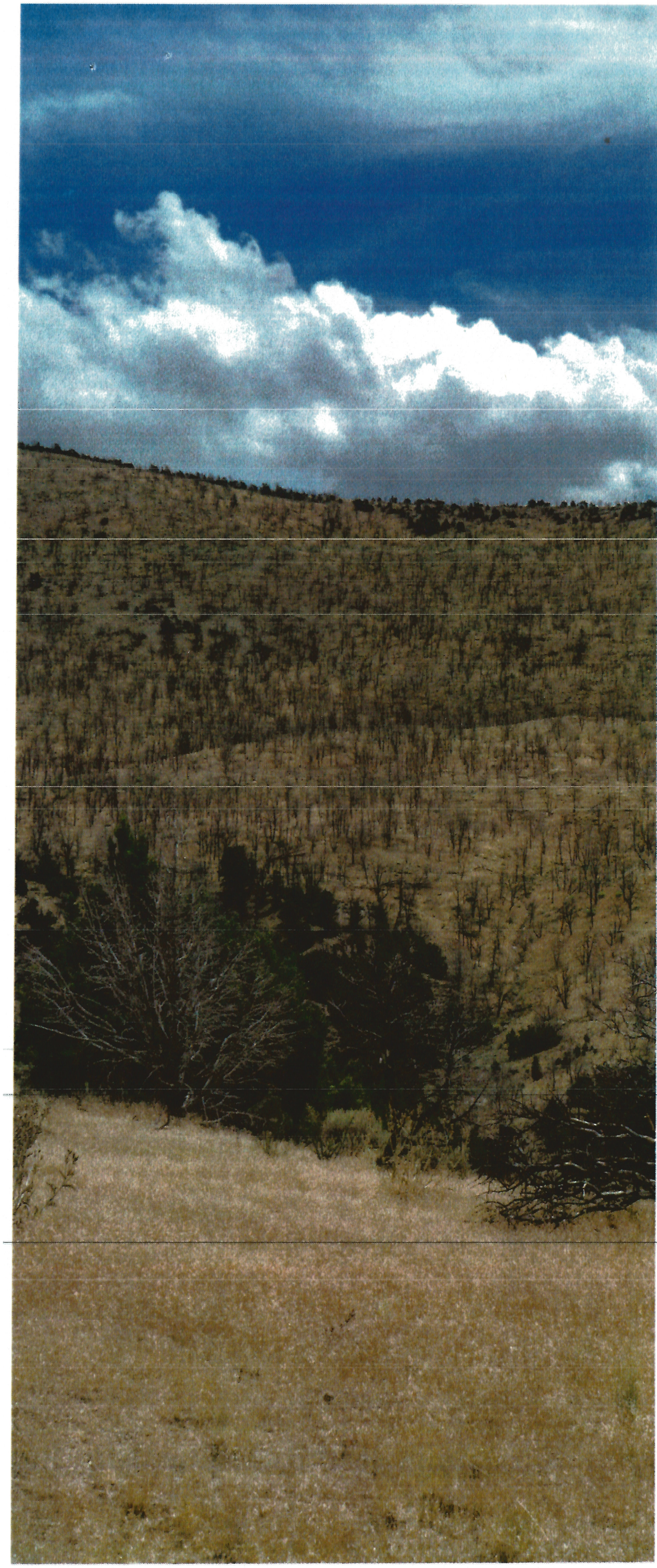
Multiple scientific studies cited in the cheatgrass report demonstrate that "cheatgrass invasion creates larger and more frequent fires by creating continuity of fine fuels." Anything from a roadside cigarette butt to a hot tailpipe on an ATV can ignite cheatgrass and spark a wildfire. And cheatgrass seeds are adept at surviving fire; therefore, cheatgrass fires often lead to establishment of a cheatgrass monoculture. "The costs and difficulties of combating both further cheatgrass expansion or retention — and minimizing the frequent fires that result — are high from both the ecological and the economic perspectives." The science cited in the report puts the threshold for avoiding the ecological and economic consequences of cheatgrass infestation at between 5% and 25% of land area.

The cumulative advantages of this invasive weed over native bunch grasses make cheatgrass a formidable opponent. As the research demonstrates, two key factors facilitate cheatgrass dominance over native plant species:

- Ground disturbance.
- Seed spread.

Livestock grazing continues to cause ground disturbance, and the authors note, "Reduction or elimination of livestock grazing achieves results on a sufficiently large scale, but full restoration can take decades." They also warn against prescribed fire and fuel-break construction, which "risk a worsening of cheatgrass infestations."

For wildfire mitigation and containment activities, the report recommends avoiding the use of "ground-disturbing equipment," which "creates a seedbed for cheatgrass." The bulk of Forest Service funding for wildfire mitigation goes to mechanical tree-thinning, which employs ground-disturbing equipment like masticators, skidders, and feller bunchers. These



Cheatgrass now dominates a former pinyon-juniper woodland following a wildfire in Nevada's White Pine Range.

mechanical "forest-health treatments" not only create conditions favorable to cheatgrass infestation, but the machinery used can introduce cheatgrass seeds, causing new infestations. Thinning trees also removes tree canopy, which provides more sunlight on the ground, further supporting the spread of cheatgrass.

Multiple studies identify prevention of ground disturbance as the best way to limit the spread of cheatgrass. Native ground cover in the arid West often consists of a "biological soil crust" (lichens and mosses) and "perennial bunchgrasses," which are more resistant to ignition than cheatgrass. The combination of biocrust and bunchgrasses also creates a synergy that resists cheatgrass invasion. Soil-disturbing machinery destroys the biocrust and damages native grasses, inviting cheatgrass establishment; then, cheatgrass outcompetes native bunchgrasses.

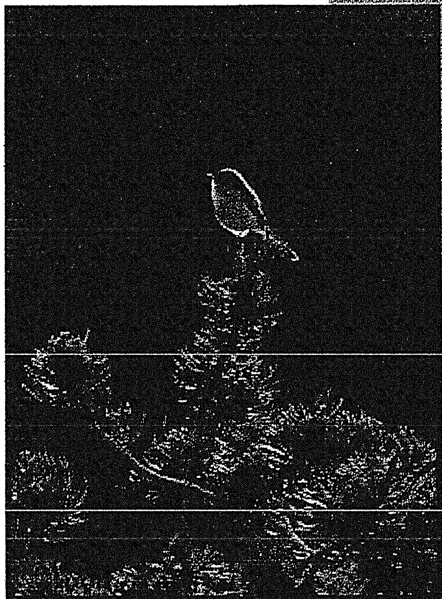
As part of a wildfire mitigation project, this masticator was used to grind entire trees into mulch in Central Colorado. Ground-disturbing heavy equipment such as this can spread cheatgrass seeds, damages native plants, and destroys the beneficial fungi network in soil, creating optimal conditions for invasive cheatgrass to take root.

Soil disturbance also damages the soil's symbiotic fungal network, which supports native plant species, including trees, and it can take up to a decade for these fungi – i.e., mycorrhizae – to recover from mechanical disturbance. Native plant species rely on mycorrhizae, which enhance nutrient uptake, but cheatgrass can thrive without the fungi. Cheatgrass also expands rapidly "because it is a prolific seed producer, can germinate in spring and autumn giving it a competitive advantage over native grasses, is tolerant of grazing, and increases with fires," according to a 1996 report — *Cheatgrass: The invader that won the West.*

Less than a year after masticators shredded mature pinyon-juniper forest in Central Colorado, fine fuels have spread. Citing established science, the cheatgrass report by Molvar et al. recommends, "Prevent pinyon-juniper removal in areas where woodlands are mature" to prevent cheatgrass infestation.

Other studies show that cheatgrass "can outcompete native grasses for water and nutrients because it is already actively growing when native plants are initiating growth." Cheatgrass "ultimately drains soils of available nitrogen, which helps cheatgrass exclude native grasses" and exhausts other soil nutrients needed by native plants. The science also shows that cheatgrass "depletes soil water in spring much more rapidly than native species," preventing the survival of native seedlings and subjecting adult native plants to moisture stress.

For a litany of reasons, minimizing cheatgrass infestations and restoring infested lands to natural conditions should be "a priority dictating the outcomes of land-use and land management decisions throughout the arid West." With their cheatgrass report, Molvar et al. add more scientific weight to the arguments against mechanical forest-thinning for fire mitigation. Recent record-breaking grass fires in Texas, Hawaii and Colorado reinforce their conclusions.



Sharing the Land with Pinyon-Juniper Birds

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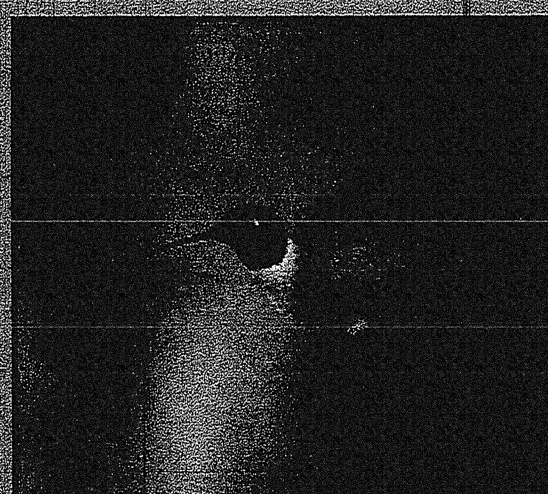

Scott W. Gillihan
Rocky Mountain
Bird Observatory

ROCKY MOUNTAIN BIRD OBSERVATORY

State of Utah
Department of Natural Resources
Division of Wildlife Resources

National Fish and Wildlife Foundation

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Sharing the Land with Pinyon-Juniper Birds

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Blue-gray Gnatcatcher (top)—Tony Leukering, RMBO
Landscape—Glenn Giroir, RMBO
Black-chinned Hummingbird (bottom)—© Brian Small / www.briansmallphoto.com

The mission of Rocky Mountain Bird Observatory is to conserve Rocky Mountain and Great Plains birds and their habitats through research, monitoring, education, and outreach.

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Sharing the Land with Pinyon-Juniper Birds

next season - Ap thru July -

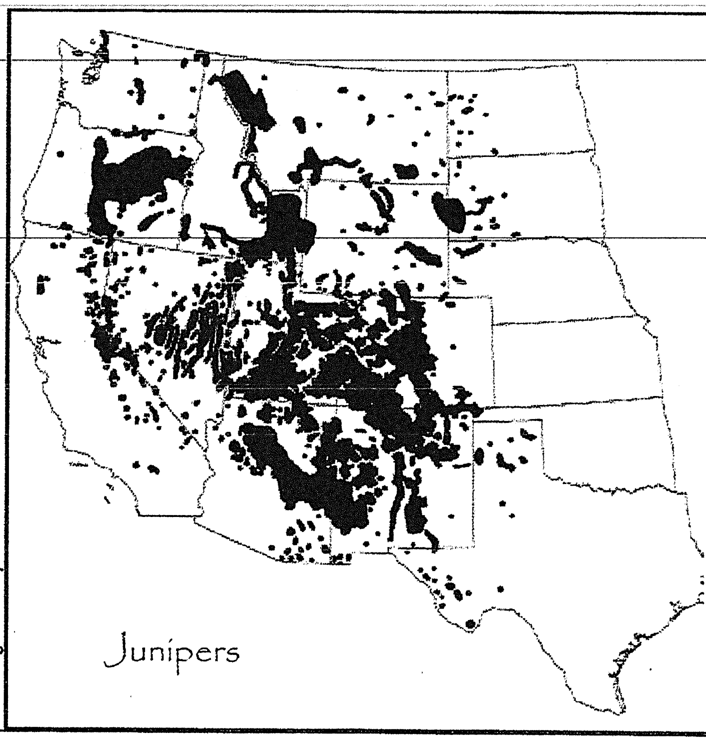
Why Share?

The pinyon-juniper woodland is a widespread ecosystem of the North American West. Estimates of its current extent vary widely, depending on how the habitat is defined and delineated. A recent estimate is 55.6 million acres (22.5 million ha; Mitchell and Roberts 1999). It is widely regarded that the extent of pinyon-juniper is increasing as some grasslands and shrublands are being transformed by pinyon-juniper encroachment. This expansion has been facilitated by a combination of climatic changes, fire suppression and, in some areas, overgrazing, which has removed the grassy understory that ordinarily would carry a fire. In some areas, pinyon-juniper woodlands are moving back into areas that formerly were woodlands but had been cleared in the late 1800s and early 1900s to meet the demand for pinyon-juniper wood products. In other areas, juniper is expanding into grasslands and shrublands where it had never been recorded before. Some researchers maintain that, at least in some areas, pinyon-juniper occurs in a dynamic equilibrium with adjacent vegetation types and that the expansion is part of a natural cycle independent of or, at best, facilitated by, human activities (Belsky 1996; Swetnam et al. 1999). These and other topics are still being debated; a number of conferences and symposia have been held to bring together researchers and managers who work in pinyon-juniper in order to share knowledge and identify information gaps that still exist (e.g., Gifford and Busby 1975; Aldon and Loring 1977; Everett 1987; Aldon and Shaw 1993; Monsen and Stevens 1999). What has come out of those gatherings is an appreciation for the habitat itself and a recognition of the need for more information.

While the body of knowledge regarding pinyon-juniper woodlands grows steadily, some synthesis of the existing knowledge is in order so that landowners and land managers can go about their business of caring for the land. One area where such a synthesis is needed, where an information vacuum exists, is in the area of managing pinyon-juniper woodlands to benefit birds.

Across a broad spectrum of habitat types, no wildlife group is as species-rich, as visible, or as vocal as birds. Pinyon-juniper woodlands are no exception. The pinyon-juniper bird community, especially in mature stands, contains a high number and variety of birds --- more than 70 species are known to breed in pinyon-juniper woodland --- although perhaps no more than 20-30 species may occur at any one site (Balda 1987; Balda and Masters 1980; Paulin et al. 1999). Pinyon-juniper woodlands support one of the highest proportions of obligate or semi-obligate bird species among forest types in the West (Paulin et al. 1999). Species closely tied to pinyon-juniper (scientific names of all species mentioned in the text are listed in the Appendix) include Black-chinned Hummingbird, Ash-throated Flycatcher, Cassin's Kingbird, Gray Flycatcher, Western Scrub-Jay, Pinyon Jay, Juniper Titmouse, Bushtit, Bewick's Wren, Northern Mockingbird, Blue-gray Gnatcatcher, Gray Vireo, Black-throated Gray Warbler, Lark Sparrow, and Black-chinned Sparrow (Balda and Masters 1980). However, little information is available on management practices that benefit bird communities in pinyon-juniper woodlands.

Birds can be useful indicators of biological integrity and ecosystem health (Hutto 1998). They fill this role because they comprise a diverse group of specialists that occupy a broad range of habitats, are sensitive to environmental changes, and reflect the abundance and diversity of other organisms with which they coexist. Responses by bird communities to environmental



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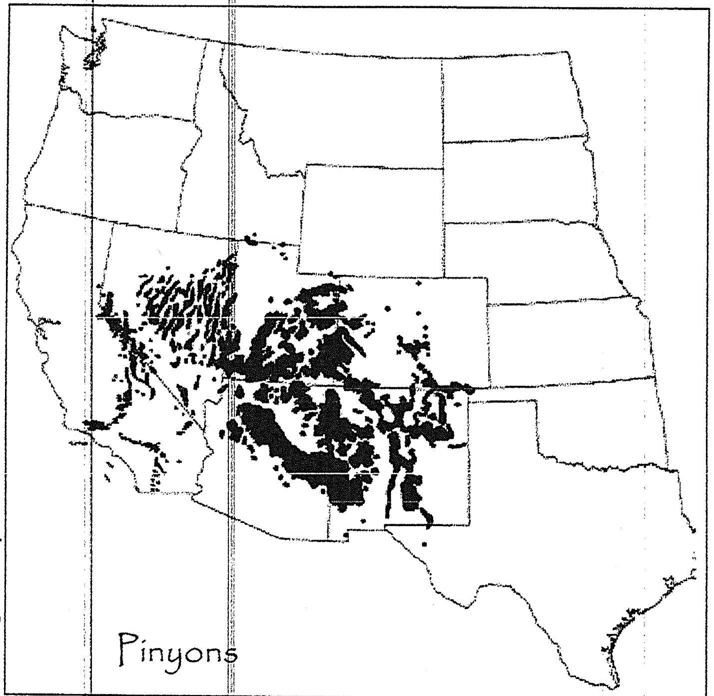


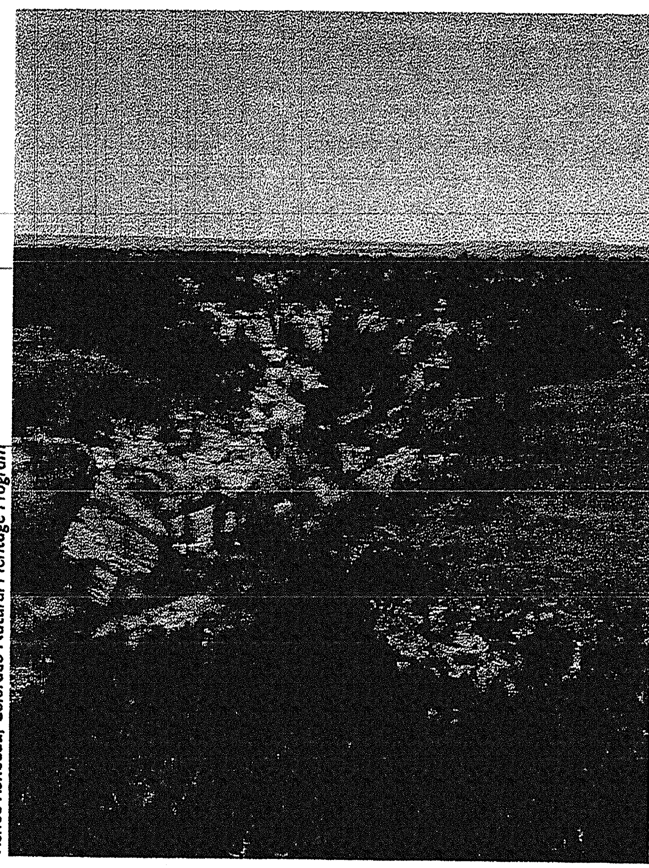
US Geological Survey 1999

changes are well documented, and changes in bird communities often reflect the effects of resource management, conservation, restoration, and environmental degradation. There is a strong and growing interest, both nationally and internationally, to manage and conserve bird populations. Examples of this interest include the numerous bird conservation plans that have recently been completed (e.g., Brown et al. 2001, Donaldson et al. 2000, Kushlan et al. 2002, North American Waterfowl Management Plan Committee 1998; Rich et al. 2004).

Birds are a tremendous economic resource in and of themselves. Birds provide untold billions of dollars in ecosystem services as pollinators and seed dispersers of ecologically and economically important plants, and as predators on insect pests in forested and agricultural landscapes. Birds are also the basis for a recreation-related economic boon. A recent federal report found that 46 million birdwatchers across the United States spent $32 billion in 2001 on bird watching and related activities (U.S. Fish and Wildlife Service and U.S. Census Bureau 2002). This spending generated $85 billion in overall economic output and $13 billion in federal and state income taxes, and supported more than 863,000 jobs. Communities in pinyon-juniper country shared in this economic vitalization since certain bird species can only be seen in pinyon-juniper habitats and others reach their highest densities in pinyon-juniper.

Pinyons and junipers rely heavily on animals for seed distribution, a dependence that makes the pinyon-juniper woodland unique. Other forest types may require animals to disperse some seeds of some plant species, but in pinyon-juniper, animals are critical to the dispersal of the seeds of the dominant tree species. Birds, in particular, are responsible for this important role in the maintenance of pinyon-juniper woodlands. This unique ecological relationship is yet another reason for landowners and land managers to share pinyon-juniper woodlands with birds and other wildlife.

The purpose of this document is to provide information on the management of pinyon-juniper woodlands that will benefit individual bird species and bird communities while still using the woodlands for other purposes. This information will be useful to private landowners and natural resource managers on public lands in developing more comprehensive management strategies that benefit the long-term health and productivity of pinyon-juniper communities throughout the western U.S.

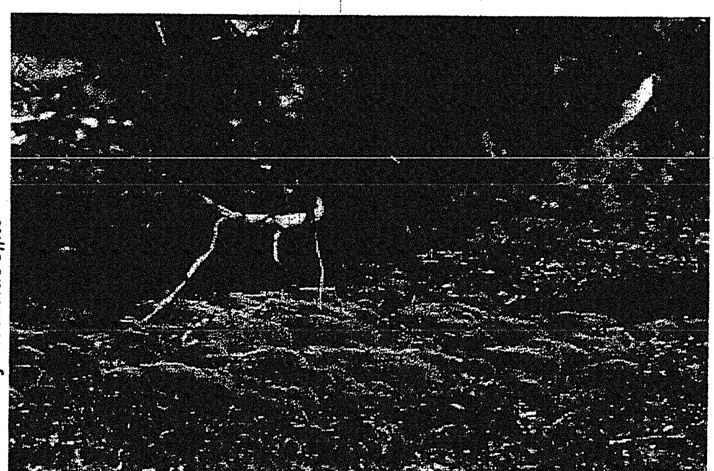


Renee Rondeau, Colorado Natural Heritage Program

Ecology of Pinyon and Juniper Woodlands

Pinyons are low-growing, rounded pines with one to five needles per bundle (fascicle), depending on species. The pinyon pine is represented by about 11 species, the two most widespread species in the West being the singleleaf pinyon and the Colorado pinyon. The other nine species are found primarily in Mexico, with some ranges extending into the southern portions of California, Arizona, New Mexico, and Texas (Lanner 1981).

Junipers are also conifers but with overlapping scales for leaves and seed-bearing cones that resemble small berries. Junipers associated with pinyon-juniper woodlands include about 17 species, including some in Mexico. The three species most commonly associated with pinyons in the West are alligator juniper, one-seed juniper, and Utah juniper. Other widespread species are Rocky Mountain juniper and western juniper (Lanner (California Partners in Flight 2002; Neel 1999; Parrish et al. 2002); species associated with pinyon-juniper range from Joshua tree and barrel cactus at low elevations to limber pine and bristlecone pine at high elevations (Tausch 1999a).



Moss and biological soil crust in a pinyon-juniper woodland

Temperature and moisture play leading roles in the distribution of pinyon and juniper. Pinyon-juniper woodlands occur where annual precipitation is just 8-20 inches (200-500 mm) (West et al. 1975). That precipitation is delivered as winter snow, spring rain, and infrequent summer thunderstorms. Although mixed stands of pinyon and juniper abound, pure stands of either also occur. Junipers are more tolerant of drought and cold and therefore often dominate at the lower elevations and sometimes at the higher elevations, although the increased moisture usually associated with higher elevations tends to favor pinyons (West 1999).

At higher elevations, junipers may drop out entirely, and pinyons grow larger and in denser stands (Pieper 1977). Both pinyons and junipers have wide elevational ranges, occurring from 2,700 to 11,000 feet (820-3,400 m)

The soils in some pinyon-juniper woodlands are held in place by a biological soil crust --- a complex community of mosses, lichens, fungi, and algae forming a soil surface layer up to several inches thick. This gray-green crust is firm enough to hold soil in place if undisturbed, but easily breaks down under traffic from humans, vehicles, or livestock (Lanner 1981). The crust is predominantly moss within the dripline of trees, and algae beyond the dripline. The importance of biological soil crusts and their role in promoting soil productivity and preventing erosion has only recently been recognized and additional information is needed to more fully understand its role in pinyon-juniper communities (Belnap et al. 2001, Belnap and Lange 2001, Ladyman and Muldavin 1996).

Pinyon-Juniper Animals

Both pinyons and junipers provide food for wildlife species. Mammalian consumers of pinyon seeds include deer mouse, pinyon mouse, Abert's squirrel, rock squirrel, cliff chipmunk, Hopi chipmunk, Uinta chipmunk, Colorado chipmunk, desert woodrat, Stephen's woodrat, white-throated woodrat, Mexican woodrat, bushy-tailed woodrat, black bear, and desert bighorn. Avian consumers include Clark's Nutcracker, Pinyon Jay, Mexican Jay, Western Scrub-Jay, Steller's Jay, and Juniper Titmouse. Some insects consume pinyon needles, as do mule deer occasionally. Bark beetle larvae (mountain pine beetle and pinyon engraver beetle) and rodents (especially porcupines) consume pinyon pine phloem. Weakened or dead trees may also become host to woodborers or other insects. Other insect consumers include various sawfly species, pitch midges, and gall midges (Lanner 1981). In pinyon-juniper woodlands, the importance of pinyon pines to birds is clear --- the number of breeding bird species increases as the number of pinyon pines increases (Masters 1979).

While pinyons provide food for many animals, the animals in turn provide a service to the trees by dispersing their seeds. Most pine species have winged seeds to facilitate dispersal; the seeds are released from the cone and carried away from the parent tree by the wind. However, pinyon seeds are wingless and enclosed in pockets within the cone, which holds them in place. Rather than being dispersed by wind or gravity, pinyons rely on rodents and certain bird species to remove and disperse their seeds. Birds gather seeds in late summer and early fall and cache them in the soil for later consumption when few other food sources are available. Birds fail to relocate some of the cached seeds, which then may sprout and grow. Seeds dispersed away from the parent trees may give rise to a new stand of pinyons.

BLM-Grand Junction Field Office

Mature juniper with nest cavity

Birds carry the seeds in their bill or esophagus. Some species are better adapted for carrying seeds than others; the Western Scrub-Jay can carry only 4 or 5 pinyon seeds, the Steller's Jay can carry up to 18, and the Pinyon Jay up to 50. The champion of seed-carrying capacity is the Clark's Nutcracker, which can carry up to 95 seeds thanks to a special pouch under its tongue (Lanner 1981). An individual jay or nutcracker may cache thousands or even tens of thousands of seeds each season, up to 13 miles (21 km) away from the parent tree (Lanner 1981; Vander Wall and Balda 1981). It has been estimated that a flock of 250 Pinyon Jays can cache 4.5 million pinyon seeds during a five-month period (Ligon 1978).

Junipers also provide food for wildlife. If more suitable browse is not available, mule deer will eat juniper foliage, sometimes browsing it extensively. Juniper berries, which consist of a hard-coated seed enclosed in a fleshy outer covering, are eaten by mammals such as rabbits, gray fox, black bear, coyote, striped skunk, and a variety of rodents, and by birds such as Western Bluebird, Mountain Bluebird, Townsend's Solitaire, American Robin, Bohemian Waxwing, and Cedar Waxwing (Lanner 1981; Chambers et al. 1999; White et al. 1999). These animals serve an important role in the future of local juniper populations by dispersing the seeds. When eaten, the hard-coated juniper seed often passes through the digestive system intact and is excreted, falling to the ground where it may germinate and become established. Removing the outer flesh increases by a factor of 10 the probability that a juniper seed will germinate. The junipers themselves facilitate this process by producing berries that are conspicuously colored blue or red and making them readily accessible on the outer layers of foliage.

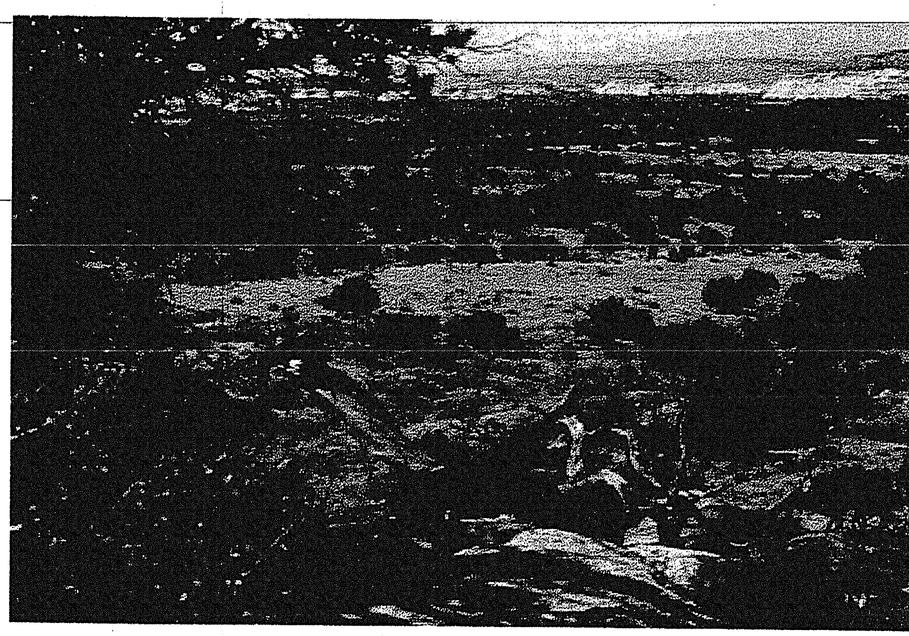
Birds are the primary seed-dispersal mechanism for some juniper species (Chambers et al. 1999). Birds are effective because they deposit seeds under woody vegetation --- a suitable site for germination and seedling growth. Bird-facilitated dispersal is particularly important for reestablishing junipers within woodlands that have been burned or killed by insects or drought, provided some dead trees remain standing. Also, since seeds are usually deposited singly or in small clusters, the chances of density-dependent seed depredation are reduced. However, juniper seeds must be covered by soil to germinate, something not accomplished by the birds. Bird-dispersed seeds must be buried by some other means such as trampling, frost heaving, soil deposition, or rodent caching. Mammals distribute juniper seeds, too, sometimes traveling a mile or more before depositing their seed-ladened scats.

For animals that rely on these trees for food, mixed pinyon-juniper woodlands provide overlapping seed supplies. Singleleaf pinyons produce seeds every two to three years and Colorado pinyons produce seeds every five to seven years, although some seeds may be produced every year. Junipers typically produce seeds annually, although there is variation among individuals and years (Tueller and Clark 1975; Janetski 1999). Dry years seem to trigger heavy juniper berry crops, while big crops are rare in wet years.

Animals make use of the trees in other ways, too. Bats use cavities, loose bark, and broken limbs as maternity roosts. Woodrats make nests of shredded pinyon bark or juniper bark fibers (Lanner 1981). Rock squirrels use the tree cavities as caching sites of pinyon nuts and juniper berries. Birds build nests in the foliage or in tree cavities, in some cases using juniper bark fibers. Insectivorous birds search the bark and foliage of pinyon-juniper trees for insects, and raptors perch in the trees while hunting. Downed logs are used as cover by invertebrates, small mammals, and reptiles. Big game species use pinyon-juniper stands for thermal cover, especially where junipers are plentiful.

The Changing Landscape

Mature pinyons may grow to be 600 years old and mature junipers may surpass 1,000 years. Even so, in many areas the trees are less than 140 years old --- roughly the time that has passed since mining and other Euro-American settlement impacts have occurred. Little information is available about pinyon-juniper woodlands prior to Euro-American settlement. Many researchers believe that woodlands of juniper or juniper with pinyon were less common than now and in some areas a more savanna-like landscape prevailed in which the trees were scattered throughout a grass- or shrub-dominated landscape. Pinyon-juniper woodlands and even forests (where tree crowns touch) certainly existed, but they may have been largely confined to areas protected from fire, such as steep and/or south-facing slopes, rocky areas, areas with shallow soils, or other situations that precluded growth of the fine fuels needed to carry fires (Roundy and Vernon 1999; West 1999). Native Americans set fires to improve game habitat by clearing dense brush and possibly to protect their homes by reducing dangerous fuel loads; those fires and lightning-caused fires likely served to control tree expansion into grassy or shrubby areas.



BLM-Grand Junction Field Office

Pinyon-juniper has expanded downward in elevation into areas with deeper soils and, historically, more fine fuels and higher fire frequency. This expansion was facilitated by the removal of fine fuels through heavy grazing and by the active suppression of wildfires (Roundy and Vernon 1999). In contrast, a study in a portion of the Colorado Plateau (Mesa Verde National Park) suggested that pinyon-juniper densities at higher elevations with naturally lower fire frequency have probably not increased greatly over the last 100 years and that the increases observed are probably the result of climatic changes rather than fire suppression (Romme et al. 2003). Fire intervals in much of the Mesa Verde area have likely always been on the order of hundreds of years and infrequent, severe, stand-replacing fires have likely always been the norm, rather than the exception (Romme et al. 2003). Fire history in pinyon-juniper woodlands is poorly understood, and much research remains before a clear picture of pinyon-juniper ecology can emerge (Baker and Shinneman 2004).

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HABITAT USE BY THREE-TOED AND BLACK-BACKED WOODPECKERS, DESCHUTES NATIONAL FOREST, OREGON

by

Rebecca Goggans, Rita D. Dixon, and L. Claire Seminara

Nongame Project Number 87-3-02

Oregon Department of Fish and Wildlife

U.S.D.A. Deschutes National Forest

ABSTRACT

Patterns of habitat use for home ranges, foraging, nesting, and roosting, were described for three-toed (Picoides tridactylus) and black-backed (Picoides arcticus) woodpeckers on the Deschutes National Forest, Oregon, during April-September, 1986 and 1987. A severe mountain pine beetle epidemic had created an abundance of dead and dying trees, and an agressive pest management and timber salvage program had created a patchwork of logged areas, primarily shelterwood cuts, on the study area.

All nests excavated by three-toed and black-backed woodpeckers were in portions of lodgepole pine (Pinus contorta) trees with heartrot. Evidently, both species require soft wood for excavating cavities, because of morphological adaptations associated with 3 toes on each foot. Habitat selection for mature and overmature forest stands, and against younger stands and logged areas, was documented for three-toed woodpeckers using 16 nests, 493 forage bouts, and 16 roosts, and for black-backed woodpeckers using 35 nests, 395 forage bouts, and 20 roosts. Home range sizes for 3 radio-tagged three-toed woodpeckers were 751, 351, and 131 acres (n=170, 352, and 131 locations, respectively). Home range sizes for 3 radio-tagged black-backed woodpeckers were 810, 303, and 178 acres (n= 124, 86 and 112 locations, respectively). Intra-specific home range overlap among both species appeared limited or nonexistent, except among paired individuals near the nest site. Inter-specific home range overlap was common between three-toed and black-backed woodpeckers and other Picidae.

Guidelines for management included establishing Management Areas which retain the characteristics of mature and overmature lodgepole pine or lodgepole pine-mixed conifer forest stands. Recommended sizes of Management Areas were 528 acres per pair of three-toed woodpeckers, at a minimum elevation of 4500 ft, and 956 acres per pair of black-backed woodpeckers, with some Areas at elevations less than 4500 ft. One Management Area could be designated for both species, if the respective habitat needs were met.

SUMMARY

INTRODUCTION

Three-toed and black-backed woodpeckers are two of the least known species of woodpeckers in North America. They are sympatric over most of their North American range and both are nonmigratory residents on the east slope of the Cascade Mountain Range. The woodpeckers are associated with trees characterized by scaly or flaky bark, but differ in the species of trees with which they are associated; the three-toed woodpecker is more closely associated with spruce (Picea spp.), and the roosts were on the lower study area where only lodgepole pine forest type was available. Mean canopy closure at roost sites was 40%. Mean dbh of trees in the roost stand was 6.0 in. Mean basal area of roost stands was 115 ft^2/acre. Lodgepole pine trees were used for 14 roosts. Mean dbh of roost trees was 11.0 in. Mean tree height was 65 ft.

MANAGEMENT IMPLICATIONS

Mature and overmature forest stands have a high incidence of disease, decay and mortality. Trees with disease and decay are undesirable components of a managed forest, but were used by three-toed and black-backed woodpeckers for home range, nesting, roosting, and foraging habitat. Nests were excavated in trees with heartrot, roosts were in diseased portions of trees or decayed snags, and forage sites were in mature and overmature stands, which have abundant disease and decay, and consequently abundant wood-boring insects. Conversion to and maintenance of lodgepole pine and lodgepole pine-dominated mixed conifer stands in a young, vigorous condition may eliminate or severely restrict incidence of wood-boring insects and heartrot, leading to declines in populations of three-toed and black-backed woodpecker.

Acreage of mature and overmature lodgepole pine forest stands are declining throughout the Oregon Cascades, because these stands are the prime target of the mountain pine beetle. Stands which experience high mortality nonetheless provide habitat for three-toed and black-backed woodpeckers. Individual trees within a stand may remain standing 10, 15 or 20 years, thus providing a continuum of habitat. Treating these stands, by logging, immediately converts them to a vigorous condition where incidence of death and decay is severely restricted, thus potential nesting and foraging substrate is drastically reduced. Although in time, stands without treatment may be structurally similar to treated stands, the time to reach that condition differs significantly. Because stands without treatment continue to provide habitat over a longer time than treated stands, thus there is a shorter period when old growth lodgepole pine is absent or scarce on the Deschutes or other National Forests. Consequently, a larger population of woodpeckers may survive, thereby increasing the potential for maintening viable populations of both species.

Designation of the three-toed woodpecker as an Indicator Species for mature and old growth lodgepole pine appeared appropriate, but only at elevations greater than 4500 ft. Much of the pure lodgepole pine on the east slope of the Cascade Mountain Range in Oregon occurs at elevations less than 4500 ft. We recommended the black-backed woodpecker as an Indicator Species for mature and old growth lodgepole pine, instead of the three-toed woodpecker. Unlike the three-toed woodpecker, the black-backed woodpecker used a range of elevations coincident with lodgepole pine. Futher, it responded to play-back recordings more frequently, over a longer time period, and with louder vocalizations than the three-toed woodpecker, thus may be more effectively monitored than the three-toed woodpecker.

Until more information is available, we believe the most effective method of insuring habitat for three-toed and black-backed woodpeckers is to exempt areas (i.e. Woodpecker Management Areas) from commercial or salvage timber management and place these areas under a special management strategy, which retains the characteristics of mature or overmature lodgepole pine habitat as long as possible, without treatment. Woodpecker Management Areas should be in lodgepole pine or lodgepole pine-dominated stands with the greatest probability of surviving the longest time, but if these stands no longer retain the characteristics of mature and overmature stands, or if the number of trees remaining is inadequate to support a pair of woodpeckers, then the designated Woodpecker Management Area should be relocated to a selected replacement. Replacement stands should be selected now, to provide the earliest possible replacement for declining Woodpecker Management Areas. Woodpecker Management Areas, and replacement areas, may be within areas previously designated as protected, such as old-growth areas, Spotted Owl Habitat Areas, winter recreation sites, Research Natural Areas, etc. Management Areas for each pair of three-toed woodpeckers should be 528 acres of lodgepole pine or mixed conifer forest in mature and overmature condition and at an elevation of 4500 ft or higher. Management Areas for each pair of black-backed woodpeckers should be 956 acres of lodgepole pine or lodgepole pine-dominated mixed conifer forest in mature and overmature condition. One Management Area of 956 acres, at an elevation of

4500 ft or higher, could be designated for 1 pair of both species. However, Management Areas for black-backed woodpeckers should not be restricted to elevations greater than 4500 ft because this species may be better adapted to conditions at lower elevations.

Black-backed woodpeckers are not currently assigned a special status (e.g. Indicator Species), thus designation of Woodpecker Management Areas may not be practical at this time. An alternative management strategy can be applied on a sale-by-sale basis. On each sale, habitat can be preserved for each pair of black-backed woodpeckers by removing 956 acres of inter-connected blocks of mature/overmature habitat from harvest. For example, if a sale area is 9500 acres of mature or overmature lodgepole pine-dominated habitat, management at 60% of potential would be for 6 pairs, or 6 areas of 950 acres each. The traditional approach for management of cavity-nesters at 60% of potential by retaining 60% of the snags and live replacement tree may be ineffective for black-backed and three-toed woodpeckers for two reasons. One - snags provide more than nesting habitat; snag retention at the 60% level is unlikely to occur in sufficient amounts to provide adequate feeding substrate for species dependent on wood-boring insects associated with trees with flaky/scaly bark. Two - this approach addresses a singular, albeit a key, component of the species' habitat. The interrelationships of an old growth, or mature/overmature ecosystem, and the species associated with it, are little known, but likely complex. Land managers do not, at this time, have the information necessary to manipulate habitat and insure these interrelationships will be maintained.

The figures for home range sizes and the amount of mature or overmature stands used by woodpeckers were estimated under conditions of abundant food supply. As the mountain pine beetle epidemic runs its course, and prey abundance declines, it is likely that the amount of area required to support a pair of three-toed or black-backed woodpecker will increase.

Three-toed and black-backed woodpeckers should be monitored to track changes in population levels as the mountain pine beetle epidemic runs its course and as the forest becomes increasingly managed, resulting in reduced levels of disease and decay. Survey routes to document number of woodpecker responses should be monitored annually. Population levels of three-toed and black-backed woodpeckers prior to the mountain pine beetle epidemic were undocumented, thus the effects of the mountain pine beetle epidemic on population levels is unknown. A review of population irruptions by three-toed and black-backed woodpeckers in eastern North America suggested that numbers of black-backed woodpeckers increase with increasing prey abundance, but that populations of three-toed woodpeckers are much less responsive to changes in prey abundance. It is possible that numbers of black-backed woodpeckers increased as the density of mountain pine beetles increased on the Deschutes National Forest. Similarly, populations may decline as the epidemic runs its course and prey for the woodpeckers becomes scarcer. It may be difficult to distinquish between the effects of the epidemic, and of timber management to control the epidemic, on populations of black-backed and three-toed woodpeckers. Documenting breeding success in Management Areas may be an effective method of combatting public outcry if woodpecker populations decline on the Forest.

This study provides a preliminary data base on habitat use by three-toed and black-backed woodpeckers; it is intended as a springboard for other studies. The pioneering nature of the study required a limited time frame, geographic scope, and sample size. Consequently, management recommendations represent the best available information at this point in time, but are intended to evolve as more information becomes available. Additional research should be a priority for land managers. Research needs include: (1) information on habitat use in areas without a bark beetle epidemic, (2) estimates for home range sizes of individuals of both species under a range of conditions, (3) estimates for breeding home range sizes of both species, (4) information on flexibility of the species to adjust to managed forest habitat, (5) information on the relationship of habitat quality and fragmentation to home range size, and (6) information on juvenile dispersal.

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FOREST MANAGEMENT AND CONSERVATION OF BOREAL OWLS IN NORTH AMERICA

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ABSTRACT.—Boreal Owls (*Aegolius funereus*) in North America occur throughout the boreal forests of Canada and Alaska and in subalpine forests of the Rocky Mountains north of central New Mexico. A recent assessment of Boreal Owl conservation status in the western mountains of North America suggested that Boreal Owls were not in immediate peril. However, in the long-term and in selected local areas, Boreal Owls likely face conservation problems. This conclusion reflects the hypothesized response of Boreal Owls to the type and pattern of forest harvest that occurred in the past and may occur in the future. Over the last 40 yr, a majority of timber harvest occurred as clear-cutting that removed the older, more diverse forest stands. Forest structure influences the availability of suitable cavities, the quality of roost sites, the foraging movements of individual owls and prey availability. Components of mature and older forests are especially important to Boreal Owl habitat quality; the owls nest in large tree cavities and prey populations are most abundant in older forest stands. Clear-cut sites will remain unsuitable for roosting or foraging for a century or more and new nest trees will not develop in some situations for two centuries or longer. Timber harvest which maintains components of mature forest well dispersed across the landscape may be compatible with conservation of Boreal Owls. In particular, forest management must consider the consequences of management decisions across broad spatial scales and over a long-term horizon. Metapopulation modeling and experimentation through adaptive management will be necessary to develop timber harvest practices compatible with conservation of Boreal Owls.

KEY WORDS: *forest management; Boreal Owl;* Aegolius funereus; *woodpeckers; small mammals; adaptive management.*

Administración Forestales y Conservación de Búhos Boreal en Norte América

RESUMEN.—El Búho Boreal *Aegolius funereus* en norte américa ocurre en todas partes de bosques boreal en Canadá y Alaska y en bosques sub-alpino en las montañas Rocosas norte de centro Nuevo México. Una evaluación reciente del estado de conservación del Búho Boreal en las montañas del oeste en norte américa sugiera que el Búho Boreal no está en peligro inmediato. Sin embargo, en la larga duración, y en áreas seleccionadas en el local, Búho Boreal pueden encontrarse con problemas de conservación. Esta conclusión reflecta la respuesta hipotesisada del búho boreal para el tipo y ejemplo de cosechas de bosque que ocurrió en el pasado y puede ocurrir en el futuro. En los últimos 40 años una mayoría de cosechas de madera ocurrió en el corte-completo que quito áreas de bosque maduros y de mas diversidad. La estructura de bosque influencia la disponibilidad de parcelas suficiente, la calidad de perchas, los movimientos de forraje de búhos solitarios, y la disponibilidad de cazar. Componente de bosques maduros y viejos son especialmente importante al hábitat del Búho Boreal: Los búhos hacen nidos en cavidades grandes de los árboles y poblaciones de cazar son mas abundante en parcelas de bosque viejas. Sitios cortados-completo se quedaran inconveniente para perchas o forraje para un siglo o más y árboles con nidos nuevos no se desarrollan en unos situaciones por dos siglos o mas. Cosecha de maderas que mantienen componente de bosques maduros bien dispersos a través el paisaje puede estar compatible con la conservación de Búhos Boreal. En particular la administración de bosques necesita considerar las consecuencias de las decisiones que hace a través de la escala de espacio amplio y sobre suficiente tiempo con perspectiva. Modelos de meta-población y experimentación a través de administración adoptivo va ser necesario para desarrollar costumbres compatible de cosechas de madera con conservación de Búhos Boreal.

[Traducción de Raúl De La Garza, Jr.]

The North American distribution for Boreal Owls (*Aegolius funereus*) forms a relatively continuous band extending from the Pacific to Atlantic coasts in the boreal forests of Alaska and Canac (Godfrey 1986). South of the continuous transcontinental band, disjunct populations occur in the

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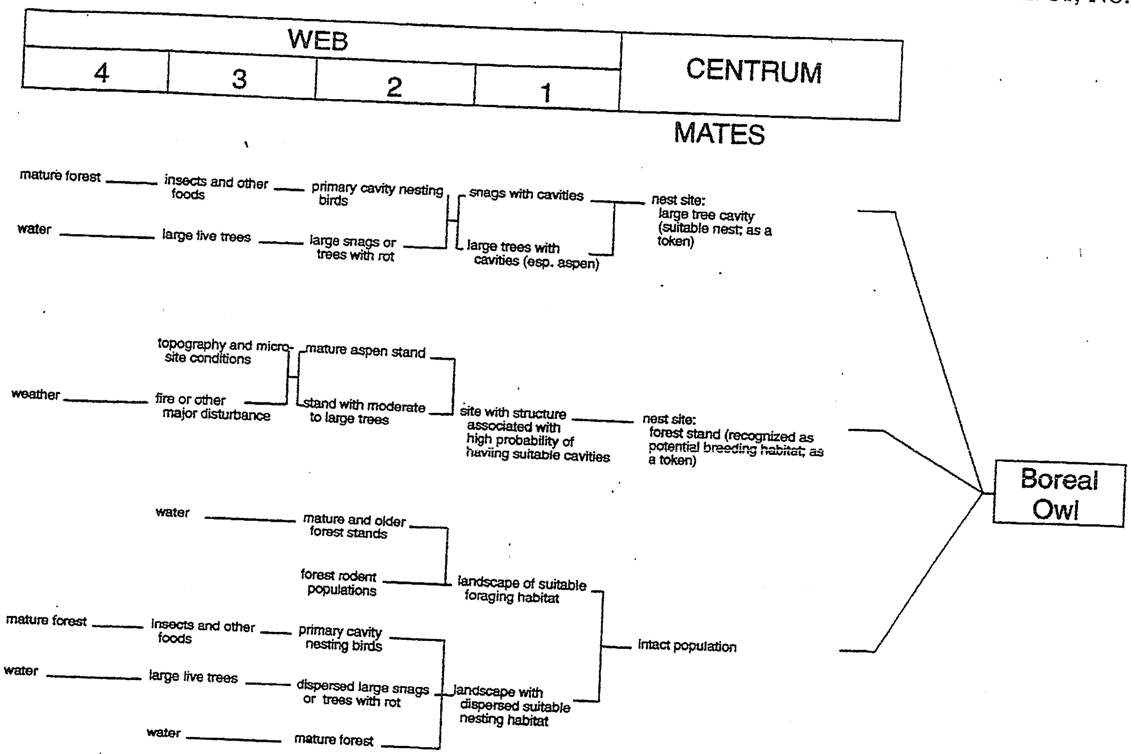

Figure 1. Envirogram (Andrewartha and Birch 1984) illustrating the relationship between Boreal Owls and specific components of the forest system. This portion of a larger envirogram (Hayward 1994b) focuses on Boreal Owl nesting ecology.

based more on efficiency in finding a cavity than increased survival after locating a nest.

The same studies in Idaho suggest that patch size may not be an important characteristic of nest stands. Nest stands ranged in size from 0.8 to 14.6 ha and averaged 7.6 ha.

Roosting habitat. Patterns of roosting habitat use also suggest these owls choose forests with particular structural features during certain times of the year. In Idaho, forest structure at summer roost sites differed substantially from paired random sites. Roost sites had higher canopy cover, basal area, and maybe most important, were significantly cooler microsites (Hotelling's T^2, $P < 0.001$) (Hayward et al. 1993). In summer, and particularly in the southern portion of their range, Boreal Owls find roost sites to minimize heat stress. We witnessed owls gular fluttering and other behaviors associated with heat stress when the temperature was as mild as 18°C. I hypothesize that the elevational distribution of Boreal Owls in the Rockies may be determined, in part, by summer temperatures and the availability of cool microsites for roosting. Forest structure, then, may influence the distribution of Boreal Owls through an interaction with limitation by heat stress.

Foraging habitat. A variety of evidence suggests that Boreal Owls in the Rockies forage principally in mature and older forest, especially spruce-fir forests (Hayward 1987). These observations are corroborated by evidence that red-backed voles (*Clethrionomys gapperi*) represent a dominant prey for Boreal Owls throughout their range in North America (Bondrup-Nielsen 1978, Palmer 1986, Hayward and Garton 1988, Hayward et al. 1993). Red-backed voles are principally forest voles (Hayward and Hayward 1995). Our studies of small mammals in Idaho found redbacks were up to nine times more abundant in mature spruce-fir forest than other forest habitats (Hayward et al. 1993). The argument for the importance of mature forest for foraging stems also from observations of snow characteristics in openings, young forest and mature forests. Snow crusting is significantly reduced

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Composition of Bird Communities Following Stand-Replacement Fires in Northern Rocky Mountain (U.S.A.) Conifer Forests

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Abstract: *During the two breeding seasons immediately following the numerous and widespread fires of 1988, I estimated bird community composition in each of 34 burned-forest sites in western Montana and northern Wyoming. I detected an average of 45 species per site and a total of 87 species in the sites combined. A compilation of these data with bird-count data from more than 200 additional studies conducted across 15 major vegetation cover types in the northern Rocky Mountain region showed that 15 bird species are generally more abundant in early post-fire communities than in any other major cover type occurring in the northern Rockies. One bird species (Black-backed Woodpecker,* Picoides arcticus*) seems to be nearly restricted in its habitat distribution to standing dead forests created by stand-replacement fires. Bird communities in recently burned forests are different in composition from those that characterize other Rocky Mountain cover types (including early-successional clearcuts) primarily because members of three feeding guilds are especially abundant therein: woodpeckers, flycatchers, and seedeaters. Standing, fire-killed trees provided nest sites for nearly two-thirds of 31 species that were found nesting in the burned sites. Broken-top snags and standing dead aspens were used as nest sites for cavity-nesting species significantly more often than expected on the basis of their relative abundance. Moreover, because nearly all of the broken-top snags that were used were present before the fire, forest conditions prior to a fire (especially the presence of snags) may be important in determining the suitability of a site to cavity-nesting birds after a fire. For bird species that were relatively abundant in or relatively restricted to burned forests, stand-replacement fires may be necessary for long-term maintenance of their populations. Unfortunately, the current fire policy of public land-management agencies does not encourage maintenance of stand-replacement fire regimes, which may be necessary for the creation of conditions needed by the most fire-dependent bird species. In addition, salvage cutting may reduce the suitability of burned-forest habitat for birds by removing the most important element—standing, fire-killed trees—needed for feeding, nesting, or both by the majority of bird species that used burned forests.*

Composición de las comunidades de aves luego del reemplazo de rodales a causa de incendios forestales en bosques de coníferas de las montañas Rocosas del norte

Resumen: *Durante las dos últimas temporadas de cría immediatamente después de los numerosos y extensos incendios de 1988, estimé la composición de la comunidad de aves en cada uno de los sitios de bosques incendiados, en el oeste de Montana y el norte de Wyoming. Detecté un promedio de 45 especies por sitio y un total de 87 especies en todos los sitios combinados. Una recopilación de estos datos con otros de conteo de aves a partir de más de 200 sitios adicionales, conducido a lo largo de 15 tipos principales de cobertura de vegetación en las montañas Rocosas del norte mostró que 15 especies de aves eran en general más abundantes en las comunidades tempranas posteriores al incendio, que en cualquier otro tipo principal de cobertura presente en las Rocosas del norte. Una especie de ave (el pájaro carpintero de espalda negra,* Picoides arcticus*) parece estar restringida en su distribución a los árboles muertos en pie, que quedan a causa del reemplazo de rodales a partir de los incendios. Las comunidades de aves en los bosques recientemente incendiados, son diferentes en composición de aquellos que caracterizan otros tipos de cobertura de las montañas Rocosas (in-*

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Table 4. The numbers of seven species of conifers (>10 cm diameter at breast height) encountered along a series of transects in the Grant Village, North Fork, Canyon Creek, and Blackfoot-Clearwater sites, and the percentages of those used by woodpeckers for feeding purposes.

Tree Species	*(n)*	*Woodpecker use (%)**
Ponderosa pine, *Pinus ponderosa*	297	80.5
Western larch, *Larix occidentalis*	100	64.0
Douglas-fir, *Pseudotsuga menziesii*	593	47.9
Engelmann spruce, *Picea engelmanni*	109	2.3
Lodgepole pine, *Pinus contorta*	647	0.2
Subalpine fir, *Abies lasiocarpa*	172	0.0

**Percentages differ significantly among tree species ($G = 1081$, $p = 0.000$).*

species are not the same as those that best predict the presence of another. Accordingly, the single variable that shows the best partial correlation with bird abundance varies widely among species (Table 7).

Discussion

Contrary to what one might expect to find immediately after a major disturbance event, I detected a large number of species in forests that had undergone stand-replacement fires. Huff et al. (1985) also noted that the density and diversity of bird species in one- to two-year-old burned forests in the Olympic Mountains, Washington, were as great as in adjacent old-growth forests. These numbers are not an artifact of birds simply passing through on their way from one adjacent unburned area to another. Most species we detected were feeding in the burned forests, and at least a third (36%) of those detected were nesting therein as well. If the birds were merely feeding while passing through, I should have detected more species and individuals in small burns and fewer in large burns because the probability of passage should decrease with increased isolation from unburned source areas. In fact, the presence of a species was

Table 6. Number (%) of cavity and open-cup nests in each of six classes of potential nest sites.

Nest Site	*Cavity Nests*	*Open-Cup Nests*	*Available (%)**
Broken-Top Conifer	15 (31)	3 (14)	6
Intact-Top Conifer	12 (25)	9 (44)	92
Broken-Top Aspen	2 (4)	0 (0)	0
Intact-Top Aspen	18 (38)	0 (0)	2
In Bank, On Ground	1 (2)	8 (38)	n/a
In Shrub	0 (0)	1 (5)	n/a

**Based on a sample of 200 trees along a single, 10-m-wide transect in the Canyon Creek site.*

largely independent of burn size; in only two cases (Townsend's Solitaire [*Myadestes townsendi*] and Solitary Vireo [*Vireo solitarius*]) was bird abundance significantly negatively correlated with burn size, and those species may indeed have been present in the smaller burns because of the proximity of unburned forest to some of the census points.

Several bird species seem to be relatively restricted in distribution to early post-fire conditions. These include Olive-sided Flycatcher, Three-toed Woodpecker, Black-backed Woodpecker, Clark's Nutcracker [*Nucifraga columbiana*], and Mountain Bluebird [*Sialia currucoides*]. Although none of these species may be considered an early post-fire obligate in the strictest sense, few strict obligates are associated with any habitat (Niemi & Probst 1990). I believe it would be difficult to find a forest-bird species more restricted to a single vegetation cover type in the northern Rockies than the Black-backed Woodpecker is to early post-fire conditions. Although it is possible that Black-backed Woodpecker populations are maintained by source refuges of low numbers in unburned forests, it is equally likely that their populations are maintained by a patchwork of recently burned forests. The relatively low numbers in unburned forests may be sink populations that are maintained by birds that emigrate from burns when conditions become less suitable 5–6 years after a fire.

Table 5. The sizes of each of three species of trees used by woodpeckers for feeding purposes in the Blackfoot-Clearwater site.

Tree Status	*Tree Diameter at Breast Height (cm)*					
	0–10	*10–20*	*20–30*	*30–40*	*>40*	*Significance**
Douglas-fir, *Pseudotsuga menziesii*						
not fed upon	269	180	77	9	0	
fed upon	10	70	123	24	10	0.0000
Ponderosa pine, *Pinus ponderosa*						
not fed upon	261	39	17	1	1	
fed upon	72	175	48	7	9	0.0000
Western Larch, *Larix occidentalis*						
not fed upon	16	4	0	0	0	
fed upon	11	30	3	0	0	0.0001

**Based on G-test of independence between tree size and signs of feeding activity.*

Table 7. Statistical results (p values) from analyses of bird communities following fire in conifer forests.

Species[a]	Variation among Sites: Abundance[b]	Variation among Sites: Occurrence[c]	Burn Size[d]	Best Correlate[e]
Red-tailed Hawk	0.164	0.293	−0.015	fire intensity (−)
American Kestrel	0.865	0.958	0.334	fire intensity (−)
Calliope Hummingbird	0.037	0.830	−0.134	larch cover
Williamson's Sapsucker	0.000	0.646	0.110	fir-cedar cover
Hairy Woodpecker	0.003	0.003	−0.362	fire intensity (−)
Three-toed Woodpecker	0.000	0.000	0.193	larch cover
Black-backed Woodpecker	0.003	0.006	0.237	number of small trees
Northern Flicker	0.000	0.000	−0.053	number of small trees
Olive-sided Flycatcher	0.000	0.000	0.276	ground cover
Western Wood-Pewee	0.000	0.000	−0.210	deciduous tree cover
Hammond's Flycatcher	0.043	0.080	−0.106	shrub cover
Dusky Flycatcher	0.000	0.000	−0.122	deciduous tree cover
Tree Swallow	0.000	0.000	0.499*	number of small trees
Gray Jay	0.000	0.151	−0.105	Douglas-fir cover
Steller's Jay	0.000	0.000	−0.089	subalpine fir cover
Clark's Nutcracker	0.000	0.000	0.088	ground cover (−)
Common Raven	0.000	0.000	0.198	fir-cedar cover (−)
Black-capped Chickadee	0.017	0.175	−0.008	spruce cover (−)
Mountain Chickadee	0.000	0.000	0.196	shrub cover (−)
Red-breasted Nuthatch	0.000	0.000	−0.337	lodgepole cover (−)
House Wren	0.000	0.000	−0.219	fir-cedar cover
Ruby-crowned Kinglet	0.000	0.000	0.004	subalpine fir cover (−)
Mountain Bluebird	0.000	0.000	0.032	fire intensity
Townsend's Solitaire	0.000	0.000	−0.430*	spruce cover (−)
Swainson's Thrush	0.000	0.000	−0.140	larch cover
Hermit Thrush	0.000	0.000	−0.079	ponderosa pine (−)
American Robin	0.000	0.000	0.160	number of small trees (−)
Varied Thrush	0.000	0.000	−0.078	subalpine fir cover
Solitary Vireo	0.000	0.023	−0.552**	larch cover
Warbling Vireo	0.000	0.000	0.218	deciduous tree cover
Orange-crowned Warbler	0.001	0.029	−0.284	larch cover
Yellow-rumped Warbler	0.000	0.000	0.339	number of big trees
Townsend's Warbler	0.000	0.014	−0.038	fire intensity (−)
MacGillivray's Warbler	0.000	0.000	−0.132	larch cover
Wilson's Warbler	0.141	0.342	0.240	number of small trees
Western Tanager	0.000	0.000	−0.310	subalpine fir cover (−)
Black-headed Grosbeak	0.000	0.333	0.062	deciduous tree cover
Lazuli Bunting	0.000	0.000	−0.275	ground cover
Chipping Sparrow	0.000	0.000	−0.307	ponderosa pine
Fox Sparrow	0.001	0.045	−0.014	spruce cover
Lincoln's Sparrow	0.001	0.000	0.361	number of big trees
White-crowned Sparrow	0.000	0.000	0.507*	deciduous tree cover
Dark-eyed Junco	0.000	0.000	0.358	number of big trees
Brown-headed Cowbird	0.000	0.000	0.228	subalpine fir cover (−)
Cassin's Finch	0.000	0.000	−0.144	fire intensity
Red Crossbill	0.000	0.000	0.209	deciduous tree cover (−)
Pine Siskin	0.000	0.000	0.114	intensity[a]

[a] Scientific names given in Table 2.
[b] p value associated with Kruskal-Wallis ANOVA, which was used to test for among-site differences in mean number of individuals per point.
[c] p value associated with G-test, which was used to test for among-site differences in the probability of occurrence.
[d] Pearson rank correlation between mean number of individuals per point and burn size. Asterisk indicates significance at $p < 0.05$, and double asterisk indicates significance at $p < 0.01$. Analyses included only sites with at least five sample points.
[e] Independent variable with highest partial correlation from multiple regression that included 13 independent variables.

Detailed studies of movement patterns and demography, needed to resolve this issue, are presently lacking.

In addition to the relative restriction of a few species to early post-fire conditions, many more were simply relatively abundant therein. In the results I note 15 species (including the five listed above) that were more frequently detected in recently burned forest than in any other cover type available in the northern Rockies. An additional six species (Common Nighthawk [*Chordeiles minor*], Calliope Hummingbird [*Stellula calliope*], Northtoco mi

ern Flicker [*Colaptes auratus*], Steller's Jay [*Cyanocitta stelleri*], Orange-crowned Warbler [*Vermivora celata*], and Chipping Sparrow [*Spizella passerina*]) were most abundant in the slightly older burned forests (10–40 years after fire) (Table 3). Three species (American Robin, Yellow-rumped Warbler [*Dendroica coronata*], and Dark-eyed Junco) were detected in both early- and mid-successional burned forest studies 100% of the time. Thus, burned forests may be of critical importance to a large number of Rocky Mountain bird species that are either relatively restricted to or relatively abundant in such forests.

The picture I paint of bird communities in burned forests contrasts sharply with that painted by other authors (Emlen 1970; Bendell 1974; Lyon et al. 1978; Niemi 1978; Lyon & Marzluff 1985), who have stated that bird communities change little after fire. After a careful review of those papers and the papers that those authors summarized, however, it is clear that the no-effect conclusions have emerged, in part, from studies of low-intensity fires or nonforested habitats and almost always from comparisons of one or two study sites and one or two controls—far too little replication to draw general conclusions about fire effects. Most important, however, the no-effect conclusions are based on composite statistics such as total bird density, species richness, and within-guild abundances, which hide more than they reveal in terms of biological effects of fire on specific species.

Bird species that use burned forests occupy a variety of feeding guilds and most rely heavily on the standing dead trees for food acquisition. For example, several bird species detected in recently burned forests may be taking advantage of the increased availability of conifer seeds after cones open in response to fire. Seed eaters that feed on conifer seeds (especially Clark's Nutcracker, Cassin's Finch [*Carpodacus cassinii*], Red Crossbill [*Loxia curvirostra*], and Pine Siskin [*Carduelis pinus*]) were more abundant in early post-fire habitat than in any other cover type, and they were significantly more abundant (Mann-Whitney $U = 29,568$, $p < 0.001$) in the first year than in the second year following a fire, when conifer-seed resources would have been more depleted. Another feeding group that seems to depend on food provided by the burned trees includes the bark-probing woodpeckers, which eat primarily wood-boring beetles (Beal 1911). Woodpeckers are clearly responding to the increase in availability of cerambicid and buprestid beetle larvae (Evans 1964; Komarek 1969; Bock & Bock 1974; Fellin 1980; Harris 1982; Amman & Ryan 1991), which in some cases are themselves responding to the increase in availability of unburned wood that lies beneath the bark of fire-killed trees (Amman & Ryan 1991). Adult beetles in the genus *Melanophila* are, in fact, specialized to feed on fire-killed trees and are capable of using infrared sensors to detect and colonize burned forests more than 161 km distant (Evans 1964, 1966). Finally, aerial insectivores (flycatchers, swallows) relied on standing dead trees as perch sites from which they sallied into the open air space for their prey.

Because the pattern of relative bird abundances differed among sites, the relative suitabilities of sites probably also differed among bird species. The same conclusion is suggested by results of the partial correlation analysis, in which the specific elements associated with bird abundance differ among species.

Most (77%) of the bird species I detected in burned forests were migrants. With concern about declining populations of migrants (Askins et al. 1990), perhaps conservation biologists should be devoting more attention to the loss of early successional habitats born of "natural" disturbance by investigating the extent to which such habitats are necessary for the maintenance of viable populations.

Conservation and Management Implications

The Importance of Stand-Replacement Fires

Fires are clearly beneficial to numerous bird species and are apparently necessary for some. The same case has been made for plants, in which some species germinate and flower only within 1–3 years after a fire and then bank their seeds for storage until the next fire (Heinselman 1981). Fire is such an important creator of the ecological variety in Rocky Mountain landscapes (Arno 1980; Gruell 1983) that the conservation of biological diversity is likely to be accomplished only through the conservation of fire as a process. Fire is in fact ". . .the only natural agent that is sufficiently widespread, abundant, fast, and regular to hold plant successions in seral stages on a vast scale and, therefore, to maintain the diversity of animal life that is so dependent upon such successional vegetation" (Komarek 1966). Efforts to meet legal mandates to maintain biodiversity should, therefore, be directed toward maintaining processes like fire, which create the variety of vegetative cover types upon which the great variety of wildlife species depend (Hansen et al. 1991).

Unfortunately, we are not currently managing the land to maintain the kind of early successional seral stages that follow stand-replacement fires and, hence, many fire-dependent plant and animal species. Why not? First, prescribed fires in conifer forests are most often low-intensity, understory burns that are justified by the argument that, with past fire prevention, forest composition is now "unnatural" and that we need to reintroduce a native fire regime of frequent, mild, understory burns to restore forests and to prevent catastrophic crown fires, which are "destructive" and "unnatural" (Biswell 1968; Alexander & Dube 1982). This justification holds only for a very limited number of habitat types, however (for example, low-elevation ponderosa pine forests). Most of the forested landscape in the northern Rockies evolved under a regime of high-intensity, large fires every 50–100 years (Fischer & Bradley 1987), not under a regime of low-intensity, frequent understory burns. A study of fire history in the Selway-Bitterroot Wilderness showed that less than 10% of the forested land experienced non-lethal fire; most of the forest types experienced partly to completely lethal fires every 100–200 years (Brown et al. 1994). Although some might argue that all forest types have been subjected to fire suppression for too long and that unnaturally dense understory buildups are leading to unnaturally severe fires, the stand-replacement fires that currently consume forests that evolved under that regime (for example, the 1988 Yellowstone fires) are not at all unusual in intensity or extent (Romme & Despain 1989).

Second, current human population and human settlement trends allow for the retention of very few areas large enough to allow free-ranging fire, and almost none of those areas have prescriptions allowing stand-replacement fires to occur (Agee 1991). Even when there is plenty of space to let fires burn, the general response is to expend enormous resources to eradicate fire because of the damage it does to timber resources, the danger it poses to humans and their buildings, and—despite ample evidence to the contrary—the damage it may do to tourism because of the visual impact. Brown and Arno (1991) have addressed this growing predicament of putting fire back into the landscape while still operating within the economic, social, and political constraints that society continues to impose: It will not be easy.

Third, there is a lack of public education about the benefits of stand-replacement fires. The biological naivete surrounding the 1988 fires was astounding and did more to muster opposition than support for "let it burn" wilderness policies. The lack of understanding demonstrated by the public, especially prominent politicians, generated a good bit of the conflict over policy (Cutler 1988). Simple facts—for example, there exists a strong distributional association between some bird species and burned forests—should be used to garner support from the public for liberal prescribed-fire policies.

Fourth, forests are not being managed in ways that mimic natural processes. One could argue that the loss of burned forest acreage due to fire control has been compensated for, at least in part, by timber harvesting. Many people believe that the conditions present after a clearcut are basically the same as those present after a severe fire (Kohrt 1988; Maschera 1988; Eggleston 1989; Swift 1993). But conditions created by a stand-replacement fire are biologically unique, at least in terms of the biomass of standing, dead trees that remain and, to a much greater extent, in terms of ecosystem structure and function. Clearcutting is, in general, a poor substitute for fire because such timber harvesting does not retain some of the most important elements, such as standing, dead trees, that are integral components of the post-fire ecosystem and that probably contribute to unique successional pathways (Agee 1991; Hansen et al. 1991) and wildlife communities.

Stand-replacement fires should not be viewed as unnatural disasters that can (and should) be prevented (Kipp 1931). As Heinselman (1985) has argued, plans to maintain stand-replacement fire regimes are justified in at least the more remote of our public lands, and prescribed-fire regimes should not be limited to periodic, mild, understory burning in lower-elevation ponderosa pine forests. Managers must also be careful to mimic all aspects of natural disturbance (such as timing, frequency, and intensity) and not just introduce disturbance as such (Hobbs & Huenneke 1992). Finally, because the pattern of relative bird abundances differs among burns, managers probably need to provide a diversity of burned cover types, intensities, and maybe even a variety in landscape contexts of burns to provide for the variety of species that may depend on fire.

Post-fire Timber Harvesting

On public lands, managers should leave an adequate amount of standing, dead trees after a fire because of the species that depend on that forest element. The current tendency to expedite timber "salvage" sales on burned forest lands needs to be re-examined. Already, as much as 60% of all timber sales on some forests in the Northern Region of the U.S. Forest Service come from salvaged timber (Schwennesen 1992). These sales, which are often exempt from public notice or comment, are generally supported by a well-meaning but misguided public that believes "dead and dying timber ought to be harvested and put to use" (Schwennesen 1992).

If some bird species require burned forests for the maintenance of viable populations (which is strongly suggested by this study), then post-fire salvage cutting may be conducted too frequently to be justified on the basis of sound ecosystem management. In instances where a salvage cut is deemed necessary, managers who wish to mitigate such effects by leaving some of the standing dead trees should be aware that bird species differ in the microhabitats they occupy within a burn. Therefore, methods that tend to "homogenize" the stand structure (such as selective removal of all trees of a certain size and/or species) will probably not maintain the variety of microhabitats and, therefore, bird species that would otherwise use the site. Selective tree removal also generally results in removal of the very tree species (Table 4) and sizes (Table 5) preferred by the more fire-dependent birds. It may be best, instead, to take trees from one part of the burn and leave another part of the burned area untouched. That way, some of the guesswork associated with choosing what to leave is avoided. This is clearly an area that deserves additional research attention.

Implications for Live-Tree Harvesting Methods

It is unfortunate that the effect of a timber harvesting method on birds (and other vertebrates) is nearly always evaluated in terms of how much the bird community composition changes from before to immediately after harvest (Hutto et al. 1993; Hejl et al. 1995). The method that best mitigates immediate harvest effects (that produces the least change) is generally viewed as the best alternative. Instead, maybe managers should favor methods that minimize deviation not from the bird communities typically associated with the pre-cut forest, but from those associated with the series of post-fire successional communities anticipated to have eventually occurred on that particular plot of land. In this light, many of the "new forestry" thinning practices, which appear favorable in terms of mitigating the immediate effects of cutting, may not represent the best strategy in terms of minimizing the impact of timber harvesting on natural patterns and processes. This is because many of the newer harvesting practices in mid- to high-elevation conifer forests create structurally artificial stands of thinned trees, which may bring "unnatural" combinations of bird species together, eliminate the full range of seral stages, and, perhaps worst of all, reduce the prospect of fire in the future (Gruell 1980). Recent full-page ads by the timber industry in the northern Rocky Mountains (for example, Missoulian, 24 August 1994, p. A-10), have, in fact, emphasized the fire-prevention "benefit" of forest thinning. Such a consequence may be fine at the urban-forest interface. It may be a well-intentioned but misplaced goal, however, for forested wildlands.

Most selective harvesting and thinning methods also result in the loss of large trees, many of which are otherwise destined to become the kind of snags that many primary and secondary cavity nesters depend on for nesting purposes should a stand-replacement fire occur. The predominant use of already existing snags by cavity nesters in burned forests (Table 6) implies that excavation is much easier in those than in the plentiful but otherwise less suitable (sometimes case-hardened) standing, dead trees. Because the most suitable nest trees for cavity excavation are snags that are themselves old-growth elements, one might even suggest that many of the fire-dependent, cavity-nesting birds depend not only on forests that burn, but on older forests that burn. Clearly, the relationship between pre-fire forest structure and post-fire bird communities deserves more attention.

A comparison of the bird communities in recent clearcuts and recent burns (Fig. 1) reveals a fair amount of similarity in the face of some important differences between the two cover types (Table 3), due primarily to the presence of standing dead trees in the burned sites, which are used for feeding and/or nesting purposes by a large number of bird species (see also Davis 1976). I found an even greater overall similarity between clearcuts and burns that are in mid-successional stages, suggesting that, when considered over all post-harvest successional stages, clearcutting may come closer to matching the natural patterns of bird occupancy on a patch of land than do many (or most) other cutting practices. I must reiterate, however, that the relative abundances of many species differ quite markedly between recently burned and recently cut forests. Even in mid-successional burns and clearcuts, which showed a greater relative similarity in bird-community composition than the earlier stages did, there were still significant differences in the absolute abundances of a large number of individual species (for example, compare the two abundance estimates for Calliope Hummingbird, Red-naped Sapsucker [*Sphyrapicus nuchalis*), Clark's Nutcracker, and Cedar Waxwing [*Bombycilla cedrorum*]). Therefore, even though the bird communities in clearcuts begin to look similar to those in fire-disturbed forests after a decade or two (Fig. 1), the bird communities are still quite different (in an absolute sense) from those that occur after a natural fire. Perhaps the best alternative to traditional harvesting methods in forests that evolved under standard-replacement fire regime may be to conduct some sort of partial harvest, after which the remaining forest would be burned lethally.

Fire (and its aftermath) should be seen for what it is: a natural process that creates and maintains much of the variety and biological diversity of the Northern Rockies. Most current cutting practices neither create large amounts of standing dead timber nor allow forests to cycle through stages of early succession that are physiognomically similar to those that follow stand-replacement fires. Unless managers begin to couple lethal burning with their cutting practices in those forests that evolved under stand-replacement fire regimes, traditional land-management practices will not achieve the goals of ecosystem management.

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Positive effects of fire on birds may appear only under narrow combinations of fire severity and time-since-fire

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Abstract. We conducted bird surveys in 10 of the first 11 years following a mixed-severity fire in a dry, low-elevation mixed-conifer forest in western Montana, United States. By defining fire in terms of fire severity and time-since-fire, and then comparing detection rates for species inside 15 combinations of fire severity and time-since-fire, with their rates of detection in unburned (but otherwise similar) forest outside the burn perimeter, we were able to assess more nuanced effects of fire on 50 bird species. A majority of species (60%) was detected significantly more frequently inside than outside the burn. It is likely that the beneficial effects of fire for some species can be detected only under relatively narrow combinations of fire severity and time-since-fire. Because most species responded positively and uniquely to some combination of fire severity and time-since-fire, these results carry important management implications. Specifically, the variety of burned-forest conditions required by fire-dependent bird species cannot be created through the application of relatively uniform low-severity prescribed fires, through land management practices that serve to reduce fire severity or through post-fire salvage logging, which removes the dead trees required by most disturbance-dependent bird species.

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Additional keywords: Black-backed Woodpecker, conifer forest, ecological integrity, fire severity, mixed-severity fire, restoration, salvage logging, wildfire.

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Introduction

The earliest synthesis of fire effects on birds (Kotliar *et al.* 2002) revealed that many species respond positively, others negatively and still others in a mixed fashion to burned forest conditions. Perhaps the most important pattern that emerged from this synthesis is that some species (the more extreme including the American Three-toed Woodpecker (*Picoides dorsalis*), Black-backed Woodpecker (*Picoides arcticus*), Mountain Bluebird (*Sialia currucoides*) and Tree Swallow (*Tachycineta bicolor*)) are relatively abundant in burned forest conditions. One (the Black-backed woodpecker) is even relatively restricted in its distribution to such conditions. For example, Hutto (1995) reported that 15 species were more abundant in burned forests than they were in any of the other 14 vegetation types included in his meta-analysis. This carries important management implications because those species may depend to a large extent on fire to create the habitat conditions they need for persistence – habitat conditions that are severely compromised by fire prevention, fire suppression, and post-fire salvage logging, seeding, tree planting and removal of native shrubs (Saab and Dudley 1998; Kotliar *et al.* 2002; DellaSala *et al.* 2006; Hutto and Gallo 2006; Hutto 2008; Saab *et al.* 2009; Swanson *et al.* 2011; DellaSala *et al.* 2014; Tingley *et al.* 2014).

Until very recently, studies of fire effects did not distinguish the effects of low-severity, mixed-severity and high-severity fires. Therefore, reported responses of species were oftentimes different from one study to the next, and terms like 'mixed responder' were included in tables generated from synthetic work on fire effects (Kotliar *et al.* 2002). Kotliar *et al.* (2005) noted that fire severity, time-since-fire, vegetation type and other considerations could probably explain some of the variation among studies, but it was not until Smucker *et al.* (2005) characterised the severity of the fire surrounding each of a series of survey points that bird responses to fire became much less ambiguous and remarkably consistent. Smucker *et al.* (2005) proposed that most bird species respond predictably to fire, but that the type of response (positive or negative) depends strongly on fire severity. Subsequently, numerous studies (e.g. Covert-Bratland *et al.* 2006; Kirkpatrick *et al.* 2006; Conway and Kirkpatrick 2007; Koivula and Schmiegelow 2007; Kotliar *et al.* 2007; Hanson and North 2008; Kotliar *et al.* 2008; Vierling and Lentile 2008; Nappi *et al.* 2010; Nappi and Drapeau 2011; Dudley *et al.* 2012; Fontaine and Kennedy 2012; Lee *et al.* 2012; Lindenmayer *et al.* 2014; Rush *et al.* 2012; Hutto *et al.* 2015; Stephens *et al.* 2015) have demonstrated a marked effect of fire severity on either the occurrence or breeding success of selected