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July 17, 2024

To: Objection Reviewing Officer

Upper Weber Watershed Restoration Project

USFS Intermountain Regional Office

324 25th Street

Ogden, Utah 84401

RE: OBJECTION AGAINST THE UPPER WEBER WATERSHED RESTORATION PROJECT

1. Name of 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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Katie Fite, Public Lands Director, Wildland Defe4nse, PO Box 125, Boise, ID 83701; phone 208-871-5738; katie@wildlandsdefense.org.

Signed for Objectors this 17th day of July, 2024.

Sara Johnson

Sara Johnson

2. **Name and Location of Project**

Upper Weber Watershed Restoration Project, Heber-Kamas Ranger District, Uinta-Wasatch-Cache National Forest.

3. **Responsible Official**

David Whittelkiend, Forest Supervisor, Uinta-Wasatch-Cache National Forest

4. **Attachments**

This Objection includes one attachment, Attachment #1.

5. Connection between Objection and Prior Public Involvement

On March 27, 2024, Objectors submitted 30-day comments on the draft Environmental Assessment (EA) for the Upper Weber Watershed Restoration Project. We listed the following failures of the agency as per required legal procedures for this project including a failure to provide the public with adequate description of the proposed project; we included 24 specific information failures that prevented the public from a clear understanding of this project. Most of these information deficiencies were not corrected in the final EA for the Upper Weber Watershed Restoration Project (hereafter "Upper Weber Project"); as such we would like to incorporate these comments into this objection instead of repeating them.

Another key issue we raised in these 30 day comments included a failure to provide the public a copy of the Biological Assessment (BA) for the project. The final NEPA documents for a proposed decision also did not include a BA. We are carrying this issue forward into this objection.

We were concerned why POD fuel breaks did not constitute new road construction within the Lakes Inventoried Roadless Area (IRA). This issue remains unresolved, as the agency did not address this question. We are carrying this issue forward into this objection.

We raised the issue of the agency's failure to demonstrate the Forest Plan amendment for management prescription 2.6 would still meet the intent of this direction in spite of the proposed exemption. This issue remains unresolved, as the agency did not demonstrate the amendment will still meet the actual intent of protecting these lands from disturbance, since the project will create significant disturbance impacts to this ecosystem.

We raised the issue of the agency's failure to provide a valid rationale for management intervention into the Lakes IRA. One issue was the false claim that only small dbh trees would be cut; another was that these forests are "fire dependent." Also, the claim that insects and diseases degrade the IRA for wildlife was unsupported. Another unsupported claim that that there are too many dead trees in the IRA, and as well, that forest stands are too homogenous and monotonous, which requires management intervention to create a diversity of age classes. All off these claims for a need for management intervention were never supported with any actual data. The project while claiming to improve wildlife habitat, has no actual habitat objectives for any wildlife species. Also, removing conifers from aspen is claimed to restore wildlife habitat without any actual information as to why this promotes wildlife. There was no map of old growth, or why treatment of old growth will maintain quality for associated wildlife. There is a great concern for the goshawk population on this forest, due to ongoing declines of nesting activity. There was no analysis as to why the project will maintain goshawk prey species. We raised the claim that uncharacteristic fire needs to be prevented in the Lakes IRA, since this term has never been actually defined; there was no information provided as to how the agency has determined the Lakes IRA is vulnerable to uncharacteristic fire. We also questioned why stand replacement fire would be detrimental to wildlife. We raised the concern about management of the sensitive Boreal Toad; planned disturbances of their habitat would seem to promote extinction of this toad in this landscape, including increased stream temperatures. We raised the concern about the failure of the agency to survey for sensitive forest raptors, including the Boreal and Flammulated Owls, and Three-toed Woodpecker. How can impacts of the project be determined if effects to these sensitive species will not be known? There were no habitat objectives listed for any of these 3 sensitive species, so there would be no habitat management for them, as well as potentially extensive destruction of nesting activities and nesting habitat. We noted the ongoing decline of western forest birds, and that this project will severely reduce habitat of 32 species likely present; these species were listed on page 12 of these comments. One impact would be the reduction of conifer seeds, forage many forest birds rely on. Removing dead trees will also remove habitat for up to 20 or more bird species that nest in cavities. A goal of the project is to reduce insects and diseases, which are also forage for wildlife, including the sensitive Three-toed Woodpecker. There is no management of old growth forests in the project area, even though up to 17 bird species depend upon old growth for productive breeding habitat. We provided examples of why treating old growth stands will reduce wildlife habitat quality. Removal of conifers from aspen will also degrade wildlife habitat. This project will have severe impacts on forest birds associated with riparian habitats by removal of forage (conifer seeds), hiding cover, thermal cover, and nesting sites. We noted that wetland riparian habitats provide habitat for up to 35 bird species, including 4 priority species. Riparian habitats in Utah have been identified as a priority habitat most in need of conservation, given that up to 104 bird species may use these various riparian habitats. We raised many concerns about the violation of the Roadless Area Conservation Rule (hereafter "Roadless Rule"). It is clear this is just a fuels reduction project, with severe impacts to wildlife. One of the issues as per violation of the Roadless Rule we raised is logging of commercial timber products, or firewood harvest. Also, the agency did not define management of roads and trails for this project. The public has not been provided with any transportation analysis, as it seems apparent that the prohibition of new road construction within IRA is being violated. Given the massive disturbances the agency plans for the Lakes IRA, it is implausible that the requirements for IRAs to be managed as undisturbed and as natural appearing can be met with this project. These massive disturbances will trigger massive mortality to wildlife, which is a direct violation of the intent of IRAs. This mortality occurs in standard land management activities, but would be unsuitable for wildlife-emphasis areas as IRAs. High levels of wildlife mortality was never identified as an exemption for IRAs. Finally, we noted that the agency's analysis of project impacts on climate change were nothing more than mere speculation. One concern is the expected increase in thermal stress to wildlife that will result from increased landscape temperatures in treated units. These impacts to wildlife were never addressed.

The issues addressed above have all been carried forward into this Objection. We have expanded on most issues, and as well, have provided references to support our concerns.

6. Remedy

It is extremely clear this proposal is a violation of the Roadless Rule for many reasons. It is just a fuels management project trying to be disguised as an ecosystem management project. Destruction of wildlife and their habitats hardly represents "ecosystem management." In addition, the major goal of this project, to prevent uncharacteristic fire, is disingenuous. First, the agency has never defined uncharacteristic fire in terms that can be measured. Second, the agency has not defined the data that demonstrates the Lakes IRA is vulnerable to uncharacteristic fire. Third, the agency did not define why stand replacement fire needs to be prevented for wildlife. Given the multiple legal violations, defined below in the body of this objection, in addition to violation of the Roadless Rule, that this project will trigger, our remedy is for the agency to withdraw the proposal and manage the Lakes IRA as is required by the Roadless Rule, without management intervention.

7. Legal Violations that the Upper Weber Project would trigger.

A. The agency will violate the National Environmental Policy Act (NEPA), the National Forest Management Act (NFMA), the Administrative Procedures Act (APA), the Endangered Species Act (ESA), the Migratory Bird Treaty Act (MBTA), the Bald and Golden Eagle Protection Act, and the Roadless Area Conservation Rule with implementation of the Upper Weber Watershed Restoration Project.

1. Violation of the Roadless Area Conservation Rule (Roadless Rule) and the NEPA.

The proposed project occurs in the Lakes Inventoried Roadless Area (IRA). Cutting of trees will not be limited to trees of "generally small dbh" in violation of the Roadless Rule. The Roadless Rule briefing paper indicates that the agency is defining "small dbh" as "not yet mature" so that very large trees will be cut. For Douglas-fir, small dbh would thus up to 36 inches dbh; for lodgepole pine, small dbh would be up to 24 inches; for spruce, small dbh would be up to 30 inches dbh; and for subalpine fir, small dbh would be up to 21 inches dbh. The final EA at 63 notes that subalpine fir and Douglas-fir trees 12 inches dbh or less will be thinned. The size of aspen trees to be cut is not identified. Hamilton (1993) defines old growth tree sizes for Doulgas-fir as 18-24 inches dbh or greater (page 29), Engelman spruce and subalpine old growth trees as 20 inches dbh or greater in Utah (pages 12, 21), Grand fir old growth as trees 24 inches dbh or greater (page 37), blue spruce trees as 16 inches dbh or greater (page 41), and aspen old growth trees as 12 inches dbh or greater (page 46). Lodgepole pine old growth trees are defined as 11 inches dbh or greater (page 51). Thus the Upper Weber project proposed to log old growth conifers, which conflicts with the Roadless Rule to cut "generally small diameter trees."

The roadless area briefing paper states that the goal of the project is to address uncharacteristic fire due to overly-dense trees. The definition of uncharacteristic fire (EA footnote 2 at page 18) is that the fire did not occur within the natural regime; uncharacteristically burned sites are expected to develop high soil erosion, insect outbreaks or invasive weeds. Apparently "normal fires" do not create soil erosion, insect outbreaks, or invasive weeds. The agency definition does not provide any actual measurable criteria for uncharacteristic fire based on identified measures of fire severity. These measures are the percentage of a burned area that is high severity fire, with a greater than 75% of the dominant overstory vegetation are killed; moderate severity where 35-75 % of the dominant overstory vegetation are killed; and low severity where less than 35% of the dominant overstory vegetation are killed; mixed severity fire refers to a fire event where a broad mix of low, moderate and high fire severity burn conditions occur (USDA 2018). A definition of uncharacteristic fire needs to define what specific percentages of burn severity levels occurred, including unburned areas, as well as where these occurred in the past. No such information was provided in the NEPA documents for the Upper Weber Project. Where have such fires occurred in the past in the general landscape of the Upper Weber landscape, which indicates such fires could also occur in this project area? What specific vegetation conditions were present in these documented uncharacteristic fires that triggered uncharacteristic fire effects? The claim that trees in the Upper Weber Project Area are so dense that they will burn "uncharacteristically" was not supported with any actual analysis, in violation of the NEPA. The agency did not provide any actual vegetation data, including basal area of conifer sites, to demonstrate these basal areas (stand density) are outside natural conditions, or are "uncharacteristic forests." Claiming these forests are outside natural density levels without any actual supporting documentation is also a NEPA violation. In addition, there was no analysis in the Upper Weber NEPA documents as to why existing forests are "too dense" for wildlife. In fact, the wildlife report and EA both suggest that impacts to wildlife will be short-term because forest densities will recover over time. The impacts are stated to result from forest thinning, which conflicts with agency claims that these forests are "too dense."

We asked the agency to define the change in songbird carrying capacity due to the reductions in hiding cover, thermal cover, and forage from the planned thinnings. In their response to comments, the agency stated that this information was not required to be provided. Why NEPA does not apply to a wildlife analysis for this project was not identified, however. A sensitive species on the forest is the Boreal Owl, a species that has been identified as sensitive to heat stress, and requires hiding cover for protection from predation for other forest owls (Hayward 1997; Herren et al 1996. Forest thinning will adversely impact this sensitive species by increasing forest temperatures and predation risks to this species. Cutting out snags will also reduce nesting habitat, as this species nests in cavities.

Forest thinning will also reduce nesting sites, hiding cover and thermal cover within the Lakes IRA to up to 67 species of western forest birds; 64% of these species are currently in decline (Rosenberg et al. 2019). Also, direct impacts of forage for this large suite of species will be massive. Removing conifers will reduce forage for up to over 25 forest birds (Smith and Balda 1979; Dobkin 1992; Smith and Aldous 1947), including:

> White-breasted Nuthatch, Red-breasted Nuthatch, Pygmy Nuthatch, Crossbills, Pine Siskin, Hairy Woodpecker, Pinyon Jay, Clark's Nutcracker, Gray Jay, Stellar's Jay, Mountain Chickadee, Cassin's Finch, Red-shafted Flicker, Crow, Robin, English Sparrow, Evening Grosbeak, Pine Grosbeak, Slate-colored Junco, Oregon Junco, Chipping Sparrow, Blue Grosbeak, Purple Finch, Black-billed Magpie, and Scrub Jay. Riparian species as the Goldfinch and Song Sparrow also consume conifer seeds.

Benkman (1996) (missed in literature cited, is Conservation Biology 7:473-479) discussed the management of conifer seed resources for forest birds, including the Red Crossbill. This report noted that large landscape areas of high conifer seed production are essential for this bird. These large landscape areas are also important as conifer seed production is sporadic in nature, so that birds have to move across landscapes to locate high production seed areas per year. Benkman (1996) noted that forest thinning will significantly reduce conifer seed production for a number of reasons, including the reduction of seed-producing trees, a reduction in conifer age, and a reduction in cross pollination due to forest thinning.

The project wildlife report identified several migratory birds that are Birds of Conservation Concern identified by the U.S. Fish and Wildlife Service. These include 3 bird species that feed on conifer seeds, including the Clark's Nutcracker, Evening Grosbeak, and Lewis's Woodpecker.

A goal of the treatments within the Lakes IRA is also to remove snags, either via contracts or due to firewood cutters. For example, the EA at 4 states that forests are "full of dead trees." It is not clear why high levels of snags are detrimental to wildlife within the Lakes IRA. There are an estimated 20 or more birds that could occur in the Upper Weber Project Area that use snags for nesting (USDA 2018; USDA 1990), including:

> American Kestrel, Three-toed Woodpecker, Black-capped Chickadee, Boreal Owl, Downy Woodpecker, Brown Creeper, Flammulated Owl, Hairy Woodpecker, House Wren, House Finch, Lewis's Woodpecker, Mountain Bluebird, Mountain Chickadee, Northern Flicker, Pygmy Nuthatch, Red-breasted Nuthatch, Tree Swallow, Violet-green Swallow, Western Bluebird, White-breasted Nuthatch, Williamson's Sapsucker, Northern Pygmy Owl, and Northern Saw-whet Owl.

Three of these species that require snags for nesting are Sensitive Species on the Unita-Wasatch-Cache National Forest: Boreal Owl, Flammulated Owl, and Three-toed Woodpecker. Research has shown that the sensitive Three-toed Woodpecker nests in stands with high densities of dead trees, up to 70 larger snags per acre (Saab et al. 2012). Removing snags is habitat destruction for wildlife within this Lakes IRA, and is inconsistent with the function of undisturbed landscapes to provide natural wildlife habitat. There was no analysis of how the project will impact snag-associated wildlife, in violation of the NEPA. Claims that leaving a few snags per acre as per Forest Plan direction are invalid as per the NEPA as the Forest Service has never demonstrated snag retention numbers left in vegetation treatments are a valid proxy for populations of snag-associated wildlife. This proxy has been identified as invalid almost 30 years ago by a Forest Service research document (Bull et al. 1997).

The project EA repeatedly also notes that forest thinning is needed to reduce insects and disease. There was no discussion as to why insects and disease processes are bad for wildlife. Not only do these processes create essential snag habitat, but also provide essential forage for wildlife, including the sensitive Three-toed Woodpecker (Goggans et al 1987). The agency did not address why reduction of foraging resources for this sensitive species is needed in this IRA. Also, as noted in the roadless briefing report, aspen trees up to 10.5 inches dbh when mature, and thus provide only small snag sizes to wildlife. It is the conifers within aspen stands that provide large snags for wildlife. For example, the NEPA analysis for the Upper Weber Project notes that mature Douglas-fir trees can be 36 inches dbh, mature lodgepole pine mature trees can be 24 inches dbh, mature spruce trees can be 30 inches dbh, and subalpine fir trees can be 21 inches dbh. The Boreal Owl, a sensitive species on this forest, uses an average snag size of 25 inches dbh (Hayward 1997), while the Flammulated Owl, another sensitive species on this forest, uses an average snag size of 28 inches dbh (Bull et al. 1990). There is a severe adverse impact to forest birds that nest in cavities when conifers are cut out of aspen stands. In addition, removal of conifers will result in increased temperatures and wind speeds in these stands, impacts that will be adverse to all wildlife species, not just birds.

Also, this removal of conifers out from aspen stands on 1,105 acres within the Upper Weber Project Area will reduce forage resources for up to 20 or more bird species, but also remove forage for the red squirrel, an important prey species for the Northern Goshawk (Salafsky et al. 2005; Salafsky et al 2007). Forest thinning has been demonstrated to reduce red squirrel populations (Holloway and Malcolm 2006).

A stated objective of treating aspen stands is to increase aspen regeneration, which is lacking (EA 20). There is no discussion as to the ongoing livestock problem with aspen regeneration, and why this issue isn't being addressed with livestock management. Removing conifers will not stop cows from browsing aspen.

There is no inventory for old growth in the Upper Weber project area. The agency says this "inventory" will be done as the project is implemented. Thus the public has no information on old growth in this landscape. There may be considerable old growth that would be degraded/destroyed with this project. As per the old growth analysis in the project EA at 63, there are 3,184 acres of potential subalpine fir old growth in treatment units; there are a potential 870 acres of old growth Douglas-fir; there are 175 acres of potential aspen old growth in treatment units, and 679 acres of potential old growth in aspen/conifer treatment units. Thus a total of 4,980 acres of potential old growth that will be degraded by the Upper Weber Project. This is 64% of all planned treatment units in the Lakes IRA 7,726 acres).

The agency claims that treatment of any old growth stands in the Upper Weber Project will maintain old growth characteristics (project wildlife report), and will actually improve old growth habitats (EA 32). However, there is no analysis in the Forest Plan documents (environmental impact statements) that address old growth treatment impacts on wildlife. This would include 16 or more bird species present on the UWC National Forest (USDA 2018; USDA 1990):

> Boreal Owl, Flammulated Owl, Three-toed Woodpecker, Brown Creeper, Golden-crowned Kinglet, Hairy Woodpecker, Hammond's Flycatcher, Hermit Thrush, Lewis's Woodpecker, Pine Grosbeak, Pygmy Nuthatch, Red-breasted Nuthatch, White-breasted Nuthatch, Northern Pygmy Owl, Northern Goshawk, and Williamson's Sapsucker.

Four of these old-growth associated wildlife species are sensitive species on the UWC National Forest: Northern Goshawk, Boreal Owl, Flammulated Owl, and Three-toed Woodpecker. No monitoring data was cited, including with past projects, to measure old growth treatment impacts on these 4 sensitive species. As was noted in the project EA (Figure 3), goshawks on the UWC National Forest have been in significant decline for quite a few years; occupancy was about 50% of known territories in 2003, but only 10% in 2020. In spite of this ongoing significant impact, the agency did not define the current or planned habitat conditions for the goshawk in the Upper Weber Project Area. There are no descriptions of the structural stages (SS 1-6) for the project area, either currently or post-project. There is no discussion, as well, as to why current structural stages need to be modified to improve goshawk habitat. As previously noted, forest thinning will be highly detrimental to goshawks due to reductions of red squirrel populations. Given that the actions are being proposed within an IRA, benefits to this sensitive species could easily by measured, but instead, were avoided. This brings into question the actual impact of this project on goshawks. The impacts of this project on goshawks is an essential analysis requirement for this project, given the severe declines of goshawks on this forest, as well as their dependence upon older, more dense forest stands for prey (Reynolds et al. 1992).

There were no habitat measures identified for the Upper Weber Project for any of these 4 sensitive species. It will be impossible to manage for any of them without habitat conservation measures. For example, the Targhee National Forest Revised Forest Plan (1997) included habitat measures for the 3 raptors. These includes providing at least 20% old growth for goshawk territories, and 40% old growth for Boreal Owl territories, with no activity allowed within a 30-acre nesting area for both the Boreal and Flammulated Owls. Given the extensive landscape fragmentation of natural habitat that will occur in the Upper Weber landscape, impacts to all 3 sensitive raptor species is highly likely. Also, there are no surveys planned or required for the Boreal and Flammulated Owls. The number of nesting areas that will be disturbed during the nesting season, and the acres of nesting habitat that will be destroyed with treatments, is unknown. Yet the agency has determined that this project will not have significant adverse impacts on these species.

In our previous comments, we requested that densities of sensitive species, such as the Boreal and Flammulated Owls, be estimated for the Upper Weber Project Area based on surveys done for other projects in this project area, including past and ongoing (Upper Provo and Bourbon). If no wildlife surveys have been done, or will be done for these projects as well, the agency needs to identify this potentially severe cumulative impact to these species due to destruction and/or disturbance of nesting habitats.

The agency noted that goshawk surveys will be done at some time in the future. The results of these "potential surveys" cannot be provided to the public, in violation of the NEPA. Also, analysis of project impacts on goshawks cannot be based on surveys that have not yet been done. It is critical that the agency demonstrate to the public specifically how goshawks are being addressed as per treatment units. Also specific mitigation measures, including buffers, need to be mapped and time periods for protection also noted.

The agency also needs to identify any Golden Eagle nest sites in the Upper Weber Project area in order to comply with the Bald and Golden Eagle Protection Act. The agency noted this eagle has been identified as a BCC by the USFWS. Nest buffers of 0.5 miles are recommended to prevent disturbances during the nesting season (Suter and Joness 1981). Protection of eagle nesting sites cannot be possible without surveys. There have been no surveys for Golden Eagles in the Upper Weber Project Area.

The current best science recommends from 20-25% old growth forest neotropical migratory birds (Montana Partners in Flight 2000). Old growth recommendations for the Northern Goshawk, a sensitive species on the UWC National Forest, is 20% (Reynolds et al. 1992). Given that historical levels of old growth in the Northern Rockies was estimated from 20-50% (Lesica 1996), optimum levels of old growth for birds would likely be 50%, or consistent with historical levels over time. The Upper Weber Project is supposed to address the historical levels of habitat, including old growth. There is no mention of historical levels of old growth. Within planned treatment areas, it was noted in the old growth analysis of the EA that 4,908 acres out of 7,726 treatment acres could be old growth, which would be 64% of these units. It is possible that this landscape has a high quality for forest birds due to old growth habitat, and associated forested snag habitat. This would demonstrate the high value of roadless lands to wildlife, due to a lack of timber harvest and other treatments that destroy/reduce old growth values.

It is not clear how many acres of forest will actually be treated in the Upper Weber Project, in violation of the NEPA. In the response to comments at 5, it is noted that treatments will include 7,726 acres for fuels reduction. But the draft

DN notes that there will be 8,079 acres of thinning, piling, and pile burning, and 612 acres of stand improvement. This would be 8,691 acres of treatment.

Crown fires are known to be essential for many birds. Hutto (1995) noted that at least 15 bird species were more abundant in recently-burned forests than unburn ed forests; standing fire killed trees provided nest sites for at least 31 bird species. Hutto and Patterson (2016) studied bird nesting activity in various levels of burned forests over an 11 year period, and of 50 bird species, 60% of them were detected more frequently within rather than outside burned habitat. The Olive-sided Flycatcher, noted in the project wildlife report to be a USFWS BCC, is noted to select the edges of burned forest for nesting *(Id.)*. The agency in the Upper Weber project NEPA analysis did not define why controlling crown fires within the Lakes IRA is needed for wildlife.

There is no transportation analysis for this project. It is impossible to determine how access to the treatment areas will be obtained. It is clear that heavy equipment, including tract or rubber-tire excavators and hydraulic masticators will be used; in mechanical vegetation treatment units, there will be designated skid trails; when using tracked machines, there will be requirements to avoid unnecessary pivots/turning to reduce soil disturbance; there will be a limit to the number of off-trail passes with boom-mounted implements; after logging, landings will be recontoured; use of unmapped routes to cross waterbodies with machinery will require consultation with the Forest Service; there will be "overland travel routes." These overland travel routes, including agency ATV use for access, may be a very significant impact on the natural appearance of this IRA, which requires a valid analysis. The draft EA noted many times that temporary roads would be used for this project (e.g., EA at 20. 22. 23. 30, 31, 50, 51. The final DN states that no temporary roads will be constructed. AS such, how is this project to be implemented now without temporary roads? Because of a lack of a transportation analysis, the public has no actual information on motorized access for this project. For example, are trails going to be converted into roads for access? Is this consistent with the Roadless Rule? It also appears that there will be over several miles of new roads as shown in EA figure 2, along Slader Ridge as well as extensions to the south down to Rhoades Lake. It appears that the agency is concealing what may be significant temporary road construction within the Lakes IRA, which is prohibited by the Roadless Rule. This is also a NEPA violation for concealing transportation planning from the public.

Given the acknowledged severe disturbance impacts of this project, especially to scenery, it is not clear how this management proposal will maintain the undisturbed character of the Lakes IRA, including violation of Forest Plan direction for visuals. Is there some exception to visuals management for IRAs that exist but was not identified in the NEPA documents or Roadless Rule requirements?

There is no map of the Lakes IRA, even though this was requested by objectors in our 30-day comments. This map should include other past/ongoing projects within this IRA, as is required by the NEPA.

Although the descriptions regarding treatment types for the project do not includes broadcast burning the draft DN notes that prescribed burning will occur (page 14, Table 4). The agency has clearly concealed how many acres are planned for prescribed burning the Upper Weber Project, in violation of the NEPA. Along with prescribed burning impacts, the Upper Weber NEPA documents also do not assess any of the project impacts on forest birds due to smoke. It has been documented that birds are highly sensitive to smoke (Defiance Canyon Raptor Rescue 2022), and the impact of direct mortality as well as reduced longer-term fitness from smoke toxicity on birds was never evaluated for this project, even though it entails massive burning activities.

Along with no analysis of smoke toxicity impacts to wildlife, the agency also failed to estimate the direct mortality to nestlings and newly-fledged birds from treatment activities, which may include prescribed burning along with cutting of multiple-sizes of conifers, and heavy removal of conifers in aspen stands. There was no information provided as to the average number of bird nests per acre where treatments will occur, and what percentage of these nests will be impacted if treatments occur during the nesting and early fledging season. Also, many bird species will attempt a second nesting period, especially if their first nesting attempt failed. Breeding seasons for birds identified in the Upper Weber NEPA documents include: 5/1-7/31 for the Three-toed Woodpecker; 1/15-7/15 for the Clark's Nutcracker; 5/15-7/15 for the Cassin's Finch;' 5/115-8/10 for the Evening Grosbeak; 5/20-8/31 for the Olive-sided Flycatcher; 5/1-7/31 for the Virginia's Warbler. All these are BCC or UWC sensitive species. The agency needs to estimate the mortality that will be triggered on these species, as per both the NEPA and the MBTA. Without this "hard look" the agency is failing to consider an important impact of the proposed project.

There was no actual analysis of project impacts on the spread of cheatgrass across this treatment landscape. Increases invasive annuals within an IRA would not represent restoring an ecosystem function. This severer impact was never identified as a violation of the Roadless Rule, however, in the agency's analysis of the Upper Weber Project. There are currently over 31.3 million acres of cheatgrass infestations across the western US (High Country News 2024). Cheatgrass is promoted by ground disturbances, including logging and prescribed burning (Forest News 2024). The Upper Weber Project will clearly increase the acreage of cheatgrass within this IRA. In addition, there are no current successful remedies for removing cheatgrass across larger areas of the landscape. Thus the increase of cheatgrass within the IRA is essentially an irretrievable impact, one that requires completion of an EIS.

The unavoidable but undefined increase of cheatgrass within the Lakes IRA from the Upper Weber Project is also a direct contradiction of the stated goal of the project, to reduce fuels and prevent uncharacteristic fire. It has been well documented that cheatgrass increases fire frequencies due to its very high flammability (Forest News 2024). It is a violation of both the Roadless Rule and the NEPA for the Forest Service to claim that massive landscape disturbances, and the resulting and unavoidable increases in cheatgrass, are needed to protect the Lakes IRA from fire, when in fact this project will increase fire potentials for this

IRA. As noted in Forest News (2024), in addition to ground disturbances, such as logging and fire, opening forest landscape with thinning benefits cheatgrass, which is a sun-loving grass.

2. Violations of the NEPA, APA, NFMA, MBTA, and ESA

The agency did not provide any assessment of how the project will impact birds and the wolverine due to exacerbation of ongoing climate impacts from global warming. The wolverine has been noted to sensitive to heat stress (Parks 2009). The massive reduction in forest density from the Upper Weber Project will increase current levels of forest temperatures, but the amount of increase was not addressed by the agency. This failure to address project impacts on this threatened species is a violation of the ESA, where the current best science was not used in the agency's assessment of project impacts to this species.

The agency violated the ESA by failing to prepare a Biological Assessment (BA) on the wolverine. There is no BA in the project record.

A sensitive species on the forest, the Boreal Owl, is also noted to be sensitive to heat stress (Hayward 1993, Hayward 1997). The impact of increased forest temperatures on this sensitive species from project treatments was not addressed by the agency. Nor was there any analysis of how increased temperatures will impact forest songbirds. Heat domes that create extreme heat are occurring across the US which have been identified as federal disasters for humans (4 recent articles in The Week 2024). However, these are likely having severe impacts on wildlife as well. A Forest News article (2021) stated that temperatures of 4-15 degrees above normal are occurring across much of the western U.S. due to climate change. A report in Montana Outdoors (2023) noted that over the past 65 years, the state's temperatures have increased 0.42 degrees per decade, which is an average increase of 2.7 degrees over that time; the greater rate of change occurs at higher altitudes; by mid-century, computer models predict a 5-degree temperature increase in eastern and north-central Montana and a 4-degree increase in central and western Montana; by mid-century, eastern Montana is expected to have 39 more days above 90 degrees each year, and western Montana will see 10-15 additional days of 90-degree plus temperatures. The impact on wildlife needs to be addressed when agency actions promote increased heat. This is required for the agency to take a "hard look" at proposed projects.

The agency also needs to assess how the Upper Weber Project will affect thermal biology of this landscape. Although the more dramatic increases in temperature in forested landscape results from clearcutting (Knoss 2016), forest thinning will also increase local forest temperature; greater sunlight exposure within thinned forest stands will increase temperatures, as well as evaporation and drying of vegetation. Lawrence et al. (2022) noted that forest cover, structure and composition change shifts the biophysical processes (the water and energy balance) that may enhance or diminish climate effects; forest cover promotes climate stability by reducing extreme temperatures during all seasons and times of day; changes to maximum temperatures are driving extinction, not changes in average temperatures; deforestation is associated with an increase in the maximum daily temperatures during the year at higher latitudes; extremely hot days are significantly more common following deforestation; deforestation has increased the frequency and intensity of hot dry summer two to four fold; forests provide local cooling during the hottest times of the year anywhere on the planet; forests are critical to adapting to a warmer world; forests also minimize the risk due to drought associated with heat extremes; continued deforestation could severely stress remaining forests by warming and drying local and regional climates; forests provide essential local climate stabilization benefits by reducing surface temperatures during the warm season, and also reduce extreme cold; the role of forests in climate mitigation must be considered in addition to its effects on atmospheric C02.

There is a recent example of how climate change, and associated extreme weather events, can impact wildlife. As reported by D'Ammassa (2020), hundreds of thousands, if not millions of migratory bird deaths occurred in New Mexico due to extreme weather events. Deaths were attributed to both hypothermia, disorientation, and starvation (USGS 2020) due to inclement weather.

The Upper Weber NEPA documents lack any discussion on the role of forests, including specifically within IRAs to protect wildlife from the ongoing impacts on climate change. Without this consideration, the agency has failed to take a "hard look" at how the proposed vast treatments within the Lakes IRA will change climate conditions for wildlife. These adverse impacts are tied directly to the agency's claim that management intervention is needed within the Lakes IRA in order to maintain ecosystem function, including mitigating fire extremes. Ecosystem function needs to include suitable climatic conditions for wildlife to persevere and reproduce. Without identifying and evaluating how the project will exacerbate ongoing climate impacts, the agency's claim of necessary management intervention is arbitrary (a violation of the Administrative Procedures Act) as well as a violation of the NEPA.

AS noted in our 30-day comments on the Upper Weber Project, the agency did not demonstrate that the Forest Plan amendment for Prescription Area 2.6 will meet the intent of protecting wildlife habitat. The amendment will instead result in expansive habitat disturbances and habitat losses to wildlife, impacts that would not occur provided prescription area 2.6 was maintained.

Attachment #1 for the Objection filed by NEC et al. for the Upper Weber Watershed Restoration Project on the Uinta-Wasatch-Cache National Forest.

Attachment 1 includes relevant portions of the following reports and/or publications cited in the objection:

Bull, E., A. Wright, and M. Henjum. 1990. Nesting habitat of Flammulated Owls in Oregon. Journal of Raptor Research 24:52-55.

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NESTING HABITAT OF FLAMMULATED OWLS IN OREGON

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nest = 28"

Abstract.—Thirty-three Flammulated Owl (*Otus flammeolus*) nests were located in northeastern Oregon during 1987–1988. The average nest tree dbh and height of the cavity were 72 cm and 12 m, respectively. Important characteristics of nest habitat included: large-diameter dead trees with cavities at least as large as those made by Northern Flickers (*Colaptes auratus*); located on ridges and upper slopes with east or south aspects; in stands of large diameter (>50 cm dbh) ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) or grand fir (*Abies grandis*) with ponderosa pine in the overstory.

Habitat para anidar de los Otus flammeolus en Oregón

Extracto.—Treintitres nidos de buhos (*Otus flammeolus*) han sido localizados en el noreste de Oregón durante 1987–1988. Los promedios de profundidad y altura de la cavidad en el árbol fueron de 72 cm y 12 m respectivamente. Las características más notables del habitat para los nidos incluían: árboles secos de gran diámetro con cavidades por lo menos tan grandes como las que hacen los *Colaptes auratus*; ubicados en cumbres y altas pendientes con frentes al este o sur; en Pinos Ponderosa (*Pinus ponderosa*) de gran diámetro (>50 cm de profundidad) Abetos Douglas (*Pseudotsuga menziesii*) o Abetos Grandes (*Abies grandis*), con Pinos Ponderosa en la parte alta.

[Traducción de Eudoxio Paredes-Ruiz]

The Flammulated Owl (*Otus flammeolus*) is a small, migratory, insectivorous cavity-nester of coniferous forests in western North America (Bent 1938). This species was once considered rare (Bent 1938), but recent studies have shown it to be common in some areas of Colorado (Reynolds and Linkhart 1987a), New Mexico (McCallum and Gehlbach 1988), California (Winter 1974, Marcot and Hill 1980), and Oregon (Goggans 1986).

Detailed information on nesting habitat is essential for effective management of habitat for this owl. Land management agencies are maintaining dead trees for cavity-nesting birds, but need more detailed information on the species and size of dead trees and sites best suited to Flammulated Owls. Our objective was to describe the nesting habitat of Flammulated Owls in northeastern Oregon.

Study Area

The study was conducted on a 5270-ha area on the Starkey Experimental Forest (Starkey) located 35 km southwest of La Grande in northeastern Oregon. Starkey is characterized by undulating uplands dissected by moderately- to steeply-walled drainages with elevations of 1070 to 1525 m. The study area consisted of a mosaic of forests (84% of area) interspersed with shallow-soil grasslands (16%). Forest types (classified by Burr 1960) were 14% open ponderosa pine (*Pinus ponderosa*), 41% ponderosa pine/Douglas-fir (*Pseudotsuga menziesii*), and 45% grand fir (*Abies grandis*) with Douglas-fir/ponderosa pine/western larch (*Larix occidentalis*).

Fire suppression and selective timber harvesting in the 1930s resulted in uneven-aged stand structure. Multilayered canopies with some much larger trees characterized most stands. As these large trees died or were cut, favorable conditions allowed new tree establishment. Over time, this created multilayered stands with numerous patches of young, even-aged trees and a few large, overmature trees. We assigned stands into 1 of 3 successional stages. Class A were stands with all trees <30 cm dbh; class B were stands with >12 trees 30–50 cm dbh/ha; class C were stands with >12 trees >50 cm dbh/ha. Ninety percent of the area had not been logged in 40 years; the remainder had a partial removal of the overstory within the last 15

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years. Large-diameter dead trees containing nest cavities were abundant (98/40 ha) and distributed throughout the study area.

METHODS

We searched for Flammulated Owls during April–July in 1987 and 1988. In April and May of each year we walked 26 routes totaling 220 km through the study area after sunset listening for Flammulated Owl vocalizations. Routes were 0.3–0.5 km apart and followed roads when available; the entire study area was covered in 2 months. We stopped every 0.3 km for 5 min. We first listened for vocalizations; if none were heard, we imitated the owl's vocalization. If an owl was heard, we recorded date, time, location, and forest type.

In June and July we searched for nests during the day in areas within 0.5 km of where individual owls were heard at night. We scratched the bark of all trees with a cavity large enough to accommodate a Flammulated Owl in order to get the owl to reveal itself. A Flammulated Owl in a cavity in June or July during the day was classified as a nest. We believe this was a valid assumption because radio-tagged male Flammulated Owls roosted on branches of live trees during nesting, not in cavities (Goggans 1986, Reynolds and Linkhart 1987b). Reynolds (pers. comm.) confirmed that cavities containing an owl during the day were always nests.

At each nest we recorded: tree species, condition (live or dead), dbh, height, cavity type (Pileated Woodpecker *Dryocopus pileatus*, Northern Flicker *Colaptes auratus*, or natural), and cavity height from the ground. Pileated Woodpecker cavities were dome-shaped and approximately 12 cm high and 9 cm wide; Northern Flicker cavities were round and approximately 6–8 cm in diameter. Habitat characteristics were measured in a 0.1-ha circular plot centered on the nest tree: location (ridge, slope, draw), slope aspect (measured with compass) and gradient (measured with clinometer), forest type and successional stage, tree density (number stems/0.1 ha), distance to opening >1 ha in size, canopy closure (measured with spherical densiometer), and number of canopy layers.

To obtain a sample of available dead trees, we searched 1534 ha of the study area and measured dead trees >50 cm dbh with potential nest cavities for Flammulated Owls. We located 3706 dead trees, 342 of which contained cavities that had been excavated by Pileated Woodpeckers or Northern Flickers, as determined by size and shape of the cavity entrance. Cavities in live trees were not recorded due to the difficulty in finding them. We did not climb trees with potential cavities to verify that they were cavities, because the majority of the trees were unsafe to climb. Only dead trees >50 cm dbh were characterized because 88% of the Flammulated Owl nests occurred in dead trees this size. These data were considered representative of the entire study area because of the homogeneity in habitat type, successional stage, and snag density throughout the study area. Cost and time constraints prohibited a complete survey of all snags on the study area.

At each dead tree with a potential cavity we recorded tree species, dbh, height, size of cavity, forest type, successional stage class, logging activity, slope aspect, and slope position. Chi-square analyses were used to compare the number of nests observed with the number expected based on data from available dead trees with cavities: 1) by forest type, 2) by tree species, 3) by type of cavity (Pileated Woodpecker versus Northern Flicker cavities), 4) by successional stages, 5) by logging activity, 6) by slope position, and 7) by slope aspect. An unpaired t-test was used to compare dbh and height of nest trees with those of available dead trees. Significance was established when $P \leq 0.05$.

RESULTS

In 1987 the first Flammulated Owl was heard on 3 May, and 24 calling sites were located in May during 19.5 hours of walking routes. In 1988 the first Flammulated Owl was heard on 10 May, and 62 calling sites were located in May during 108.5 hours of walking routes. No Flammulated Owls were heard in April either year.

Calling activity was greatest within 2 hr after sunset when 77% of the owls were first heard. Only 26% of the time spent listening was within this 2-hr period. The remainder of the time was spent listening 2–7 hr after sunset. The location of singing owls detected was independent of forest type ($\chi^2 = 0.64$, 2 df, $P = 0.73$).

We located 13 nests in 1987 and 21 nests in 1988. All nests were located in June and July, and only 1 tree was used both years by nesting Flammulated Owls. Of these 33 different nest cavities, 67% had been excavated by Pileated Woodpeckers, 27% had been excavated by Northern Flickers, and 6% had been created by decay. By comparison, the available cavities large enough to accommodate these owls included 45% Pileated Woodpecker and 55% Northern Flicker cavities. Relative to availability, Flammulated Owls used a higher percentage of Pileated Woodpecker cavities than expected ($\chi^2 = 8.15$, $P < 0.01$).

Ninety-one percent of nests were in dead trees and 9% in live trees. Seventy percent of the nests were in ponderosa pine, 27% in western larch and 3% in grand fir trees. There was no difference between species ($\chi^2 = 1.47$, 2 df, $P = 0.49$) or dbh ($t = 0.37$, 368 df, $P = 0.71$) of dead trees used as nests and those available with cavities large enough to accommodate the owls (Table 1). Height of nest trees was significantly greater than of available trees ($t = 3.49$, 368 df, $P < 0.01$).

Fifty-eight percent of the nests occurred in ponderosa pine/Douglas-fir forest types, while the remainder occurred in grand fir forest. Ponderosa pine was an overstory species at 73% of the nest sites. Although there was no difference between used and available dead trees by forest type ($\chi^2 = 3.20$, 2 df, $P = 0.13$) or logging activity ($\chi^2 = 1.6$, 1 df, $P = 0.22$), there was a difference among successional stage ($\chi^2 = 6.35$, 1 df, $P = 0.04$), slope aspect ($\chi^2 = 8.87$, 3 df, $P < 0.05$), and slope position ($\chi^2 = 9.86$, 3 df, $P < 0.05$). Ridges and the upper third of slopes were used more and lower slopes and draws were used less often than expected if selection was random. East and south slopes were used in greater proportion and north and west slopes used in lesser proportion than if used at random based on available dead trees with cavities large enough to accommodate the owls. Stands with trees >50 cm dbh were used as nest sites in greater proportion than if selected at random; 42% of the nests occurred here, yet only 24% of available cavities were in these stands.

Table 1. Measurements taken at 33 Flammulated Owl nest trees in Oregon, 1987–1988.

VARIABLE	MEAN	SD
Nest tree		
DBH (cm)	72	14.4
Height (m)	24	9.1
Cavity height (m)	12	4.7
Nest habitat		
Trees >10 cm/0.1 ha	33	14.6
Trees 2–10 cm/0.1 ha	48	29.6
Canopy closure (%)	55	20.1
Number of canopy layers	2.5	0.5
Slope gradient (%)	18	11.8
Distance to opening (m)	50	51.3

DISCUSSION

The detection of 62 singing owls during 1 nesting season suggests that Starkey had a high density of Flammulated Owls. Only a portion of the owls were detected because the entire study area could not be covered in the 2–3 week period that the birds vocalized intensively. Densities of singing owls have been reported as 0.72/40 ha in Oregon (Goggans 1986), and 2.1/40 ha (Winter 1974) and 0.03–1.09/40 ha (Marcot and Hill 1980) in California. Density of pairs has been reported as 0.47/40 ha in Oregon (Goggans 1986) and 0.03–0.5/40 ha in Colorado (Reynolds and Linkhart 1987b).

Apparent preference for Pileated Woodpecker cavities as nest sites was perhaps due to the larger cavities Pileated Woodpeckers excavate or the higher placement above the ground of these nests compared to those of flickers ($\bar{X} = 15$ m, SD = 5.6; $\bar{X} = 8$ m, SD = 6.2, respectively; Bull et al. 1986). Nests in live trees may have been underrepresented as such cavities are more difficult to detect. Nonetheless, large snags with Pileated Woodpecker cavities are clearly an important part of Flammulated Owl nesting habitat.

Ridges and upper slopes were perhaps preferred because they provided the gentlest slopes, which would minimize the energy expenditure of birds carrying prey to nests or because of prey availability. Goggans (1986) suggested such preference may be related to the diversity and density of prey. Prey may also be more abundant or at least more active on higher slopes because these slopes are warmer than lower ones (Reynolds, pers. comm.). The preference for east and south aspects may also be related to temperature and availability or abundance of prey.

Reynolds and Linkhart (1987b) suggested that stands with trees >50 cm dbh were preferred because they provided better habitat for foraging due to the open nature of the stands, allowing the birds access to the ground and tree crowns; stands of dense, young trees were avoided. Some stands of larger trees also allow more light to the ground which produces ground vegetation, serving as food for some insects preyed upon by owls.

Our findings suggest that the best way to manage habitat for the Flammulated Owl is to leave dead trees (>50 cm dbh and >6 m tall) with cavities at least as large as a Northern Flicker cavity. These trees are best left on ridges or upper slopes with east or south aspects in stands of large-diameter (>50 cm dbh) ponderosa pine/Douglas-fir or grand fir forest types, with ponderosa pine in the overstory. Retaining large diameter live trees in addition to snags, will provide for future snags. Another approach is to manage habitat for Pileated Woodpeckers and Northern Flickers in these same situations, as they will provide nest sites for these small owls over time.

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Habitat measure - Proportion of time a drainage produces cones.

80 yr Rot = 62%
300 yr = 90%

Exotic - Red sq. caused decline of crossbill in newfoundland.

7 sps Red Cross

Logging, Conifers, and the Conservation of Crossbills

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Abstract: *A survey of the forestry literature shows that as the age and area of coniferous forests decline, decreased conifer seed production and increased frequencies of cone failures can be expected. This would, in turn, cause declines in crossbills (*Loxia*), which specialize on the seeds held in conifer cones. In western North America, at least five different species of Red Crossbills (*L. curvirostra*) have recently been distinguished (Groth 1990); each specializes on a different species or even a single variety of conifer (Benkman 1993). Measures for conserving this diversity of crossbills include protecting mature and old-growth stands, and increasing rotation ages throughout the range of each of the required conifers. These recommendations are not unique to crossbills, but rather the loss of crossbill diversity is another reason to employ such measures.*

Explotación forestal, coníferas y conservación de piquituertos ("crossbills")

Resumen: *Un estudio de la literatura forestal demostró que cuando la edad y el área de los bosques de coníferas declinan, se puede esperar un decrecimiento en la producción de semillas y un incremento en la frecuencia de fracasos de los conos (piñas). Esto, a su vez, puede causar declinación en piquituertos (*Loxia*), los cuales se especializan en las semillas que se encuentran en los conos de coníferas. En el oeste de Norte América, por los menos cinco especies diferentes de Piquituertos Rojos (*L. curvirostra*) han sido reciéntemente identificadas (Groth 1990); cada una está especializada en una especie diferente o incluso en una variedad particular de coníferas (Benkman 1993). Medidas para conservar esta diversidad en piquituertos incluyen protejer rodales maduros y de crecimiento antiguo e incrementar la edad de rotación a lo largo del rango de cada una de las coníferas requeridas. Estas recomendaciones no son únicas para los piquituertos, pero la pérdida de la diversidad de los piquituertos es otra razón para emplear tales medidas.*

Introduction

Specialist species are particularly vulnerable to extinction. Among the most specialized groups of birds are the crossbills (*Loxia*), which are adapted for foraging on seeds in conifer cones (Newton 1972; Benkman 1987*b*, 1988*a*, 1988*b*, 1993; Benkman & Lindholm 1991) and whose survival and reproduction depend on the availability of conifer seeds (Newton 1972; Benkman 1987*a*, 1988*a*, 1990, 1992*a*). Crossbills also are a speciose lineage. What had previously been recognized as a single species, the Red Crossbill (*L. curvirostra*) in North America, consists of at least seven distinct species (Groth 1990). Although the taxonomic basis for recognizing these different crossbills at the species level is still largely unpublished (but see Groth 1988), each species differs in morphology, vocalizations, and ecology, and several species commonly nest syntopically without much interbreeding (Groth 1988, 1980; Benkman 1993). Moreover, most if not all of these species of crossbill are adapted specifically for foraging on single species of conifers, and in some cases just one variety of conifer (Benkman 1989, 1993). For instance, in western North America different species of Red Crossbill are

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adapted specifically for foraging on each of western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*), lodgepole pine (*Pinus contorta* var. *latifolia*), ponderosa pine (*P. ponderosa* var. *scopulorum*), and possibly Sitka spruce (*Picea sitchensis*) (Benkman 1993).

Undisturbed conifer forests were recently widespread but now are among the most intensively exploited habitats (Caufield 1990; McLaren 1990; Norse 1990; also see Rosencranz & Scott 1992). For example, only around 10% of the original old-growth forest in Washington and Oregon may remain (Norse 1990), and most second-growth on federal lands typically has rotation ages of about 80 years (Brown 1985). Many industrial forest lands are managed on a 40 to 60 year rotation (J. F. Franklin, personal communication). Consequently, species dependent on mature conifer forests, such as crossbills, will inevitably decline, with local and possibly global extinctions.

Dependency on a single resource makes crossbills extremely vulnerable to habitat loss and alteration. Because the link between crossbills and their food resources, conifer seeds, is so strong and clear (Newton 1972; Benkman 1987*a*, 1990, 1992*a*), the conservation strategy for crossbills is simplified (see Terborgh 1986). Moreover, because the basic mechanisms influencing crossbill ecology are well understood (Newton 1972; Benkman 1987*a*, 1987*b*, 1988*a*, 1989, 1990, 1992*a*), the general impact on crossbills of habitat alteration, such as that from logging, can be anticipated; this is the subject of this paper. Such predictive analyses are important, in part, because the actual status of nomadic crossbill populations is difficult to assess and large declines may go undetected. For example, during the early 1970s in Newfoundland there appears to have been a severe decline of the formerly abundant and endemic Newfoundland Crossbill (Benkman 1989, 1992*b*; Pimm 1990). This decline was not noted until nearly twenty years later, however, and only after I had predicted such a result from the introduction (in 1963–1964) of red squirrels (*Tamiasciurus hudsonicus*) onto Newfoundland (Benkman 1989, 1992*b*). Although I concentrate on crossbills, the impact may be applicable to populations of numerous other conifer seed-eating animals (see Smith & Balda 1979).

Temporal Variation in Habitat Quality

Logging at short rotation ages increases the domination of forests by younger trees (Harris 1984), which, for at least three reasons, greatly reduces cone and seed production relative to mature or old-growth forests. First, shorter rotation ages reduce the proportion of time a given stand is capable of producing seeds (Fig. 1). Most conifers begin producing cones (seeds) only after they

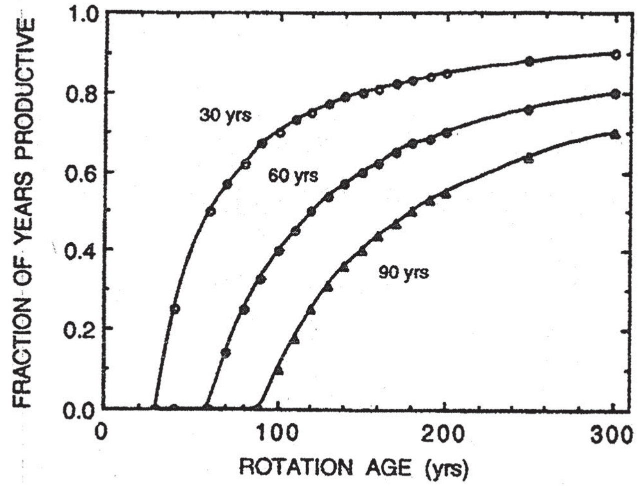

Figure 1. The fraction of years a forest might produce cones in relation to rotation age. The different curves represent different ages (30, 60, and 90 years) after which cones are produced. Although many conifers begin producing cones after 30 years of age (Fowells 1965), the smaller cone crop sizes and the higher frequency of cone failures for younger trees effectively shift the curve down and to the right (toward or below the 90 year curve).

are about 20 to 30 years old (Fowells 1965). Thus, when rotation ages are 80 years and trees produce cones only after 30 years of age, the proportion of time that an area may produce any seed is about five (62%) out of eight years. The intervals between stand-replacing disturbances such as fire or wind in unmanaged stands in western North America are variable, although they usually are greater than 80 years (see Habeck 1988; Peet 1988; Spies & Franklin 1988; Alaback & Juday 1989). For example, west of the Cascade Range many old-growth forests were older than 300 years prior to logging, and many may have been older than 750 years (Spies & Franklin 1988). Assuming that stand-replacing disturbances occur at intervals of 300 years, and that 30 years is the minimum age for cone production, unmanaged stands potentially produce seed at least nine (90%) out of every 10 years. Thus, short rotations increase the proportion of the landscape that is of pre-seed-production age.

Second, younger conifers produce smaller cone crops than do older conifers. For example, an old-growth stand of Douglas fir produces 20 to 30 times more cones than a 50- to 100-year-old second-growth stand (Burns & Honkala 1990). Maximum cone production for some conifers is not reached until trees are 200 years of age (such as the sugar pine [*Pinus lambertiana*; Fowells & Schubert 1956] and Douglas fir [Fowells 1965]). In addition, smaller cone-producing trees in a stand fail to produce cones more often than larger and presumably older trees (such as ponderosa pine and Douglas fir; Fowells & Schubert 1956, Shearer 1986, Burns & Honkala 1990). This reduces further the proportion of years that harvested forests are productive for seed-eaters. In the above example, a conifer that first begins producing cones at 30 years of age may regularly produce many cones only after 90 or more years of age (see Fig. 1).

Third, seed production by late successional conifers will be especially reduced by short rotations. In the Cascade Range, western hemlock tends to be a major component only late in succession (Franklin 1988) and is, therefore, often only a minor seed producer except in old-growth forests. For example, in the Douglas fir–western hemlock forest type in the Cascade Range of Washington, western hemlock seed production in a 100-year-old stand (mostly Douglas fir) was less than one two-hundredth of that in a nearby old-growth stand (Isaac 1943).

Crossbills are expected to benefit from mature and old-growth forests because they produce many more cones much more consistently over longer periods than do regularly logged forests. Furthermore, the small cone crops of younger forests may act as cone failures for crossbills because they require a minimum seed abundance to survive, with small cone crops less likely to meet minimum energy requirements (Benkman 1987*a*, 1992*a*). Moreover, because the larger the cone crop the more crossbills breed (Benkman 1990), the decline in large cone crops should result in smaller rates of increase after cone failures, and hence slower recovery rates.

Spatial Variation in Habitat Productivity

Geographic Variation

Because cone production can vary so much annually within a given area (Fowells 1965), special considerations need to be made to ensure that areas of abundant cones are available every year. Years of good cone production are usually followed by one to several years of little or no cone production; even the most regular cone-producing conifers have occasional cone failures (Fowells 1965). Moreover, large regions often fail to produce many—or any—conifer seeds during one to several years in succession (see Harris 1962), so that crossbills move out of these areas and concentrate where cone crops are produced (see Benkman 1987*a*). During many years, only a fraction of the total potential range can support crossbills. Thus, a reserve or system of reserves encompassing only a restricted geographic area would be inadequate to support nomadic populations of crossbills continuously, and most crossbill populations are nomadic. Indeed, bird species whose foods irregularly fail in a given area (such as tropical frugivores) tend to be "extinction prone" (Terborgh & Winter 1980; also see Janzen 1986).

Climate influences cone crop production and failures (Roeser 1942; Lowry 1966; Lester 1967; Smith & Balda 1979), hence areas experiencing similar climate are more likely to produce cone crops or fail in synchrony (see Kemp & Keith 1970). In western North America, the mountainous terrain affects local climate so that cone crop production is more likely to vary locally (Bock & Lepthien 1976; Smith & Balda 1979). Nevertheless, cone crops can fail synchronously over large mountainous regions (such as the Cascade Range of Oregon and Washington; Franklin et al. 1974).

The greater the number of distinct climatic regions within a forest reserve, the lower the probability of a complete cone failure. For example, six distinct regions, each with an independent probability of cone failure of one in three years (see, for example, Franklin et al. 1974), have a 0.0014 probability of synchronous cone failure (0.333^6). However, if only three of the regions have mature forest, then the probability of synchronous cone failure increases over 26 times to 0.037 (0.333^3). Compounding this further is that the frequency of small cone crops unable to support crossbills through the winter likely increases in younger stands. If as a result of logging the frequency of failure and of small cone crops doubles, the probability that six distinct regions are synchronously unable to support crossbills increases by over 62 times from 0.0014 to 0.088 (0.666^6). The decrease in distinct regions with mature forest and the decline in forest age from logging both compound the probability of synchronous and widespread cone failures, which are pernicious to crossbills (see Newton 1972 and Benkman 1988*a* for the effect of cone failures on crossbills).

Local Variation

Even within a given region, only certain areas may be favorable enough to maintain crossbills over many years (source habitat, for example, [Pulliam 1988]). Areas that consistently produce large cone crops and hold seeds for extended periods most likely represent source habitat for crossbills. Large cone crops result in high intake rates for crossbills (Benkman 1987*a*, 1987*b*), which improves their reproductive rates (Benkman 1990) and presumably their survival (Benkman 1987*a*). The value of reserves in protecting crossbills, therefore, depends critically on the amount of productive land or source habitat protected.

Unfortunately, for the same reasons habitats are most productive for crossbills (edaphic characteristics and climate favorable for cone production) they are likely to be most productive for commercial interests and to be intensively managed. In the Pacific Northwest, for example, lowland forests are the most productive, and they have been largely logged (Norse 1990). Most of the remaining old growth in the Pacific Northwest is at higher elevations on steep slopes (Norse 1990). Here, cone production is less than at lower elevations because it often declines with increases in elevation (as in Douglas fir and ponderosa pine [Jacobsen 1986]). In fact, most protected areas (such as wilderness areas) are confined to higher elevations (Harris 1984), where conifers likely produce fewer seeds than at lower elevations where logging is permitted.

Habitat Fragmentation

As remnant forest patches become smaller and more isolated and fragmented, many forest species decline disproportionately relative to forest loss (Wilcove et al. 1986). Crossbills have been found to decline as forests are fragmented (Helle 1985). However, their decline is more likely in response to the concurrent decline in forest age as a result of logging rather than to habitat fragmentation per se (Helle & Järvinen 1986). Forest fragmentation is likely not as detrimental for crossbills, which regularly may fly distances for greater than 1 km (Benkman, personal observation), as it is for other old-growth species (Norse 1990). Thus, local management policies that account for more area-sensitive species (such as the Northern Spotted Owl *Strix occidentalis caurian* and Northern Goshawk [*Accipiter gentilis*]) would be adequate for crossbills.

Habitat fragmentation, nevertheless, may make crossbills more vulnerable to extinction by reducing the rate at which suitable patches of habitat (those containing a good cone crop)—are colonized. Levins (1969, 1970) has shown that when extinction rates of local individual populations exceed colonization rates, the species or metapopulation may go extinct. Crossbills can be thought of as being composed of many populations repeatedly colonizing patches of habitat containing good cone crops, and then going extinct locally when the cone crop fails. As patches become smaller and more isolated, their rate of colonization by crossbills may decline. Because a nomadic crossbill may need to colonize many patches during its lifetime, even slight declines in colonization rates can be important.

Evidence of Adverse Effects on Crossbills

As expected from much greater seed production in old than young forests, Red Crossbills were more abundant in older than in younger forests in Finland (Helle & Järvinen 1986) and in northern California (Raphael et al. 1988). In another study, Red Crossbills were 30 times more abundant in old-growth (325 to >500 years old) than in younger (65 to 140 years old) forests during two years of poor seed crops in the Cascade Range of southern Washington (Huff et al. 1991). Such a pronounced difference occurred because the only conifer to produce seeds was the late-seral (at least in this forest type) western hemlock, and it was of cone-producing size and age only in the old-growth forests (Manual & Huff 1987; Huff et al. 1991; see previous discussion).

Although this system has not been modeled, diminished cone production and the increased isolation of productive habitats as a result of logging will likely result in declines of crossbills even within mature forests (see Fahrig 1992). Consequently, crossbill abundance should decline disproportionately to forest loss. Evidence of declines in crossbills that are disproportionate to the loss of habitat has been found in northern Finland: as the proportion of land containing older forests (>121 years old) diminished by 27% between the early 1950s and 1970s because of clear-cutting, Red Crossbills declined by 75% (Väisänen et al. 1986). Väisänen et al. (1986) also present evidence that crossbills declined even within an unaltered forest.

In sum, older forests tend to support more crossbills than do younger forests, and as the proportion of the landscape containing older forests declines, crossbills decrease disproportionately in abundance. This could have been anticipated from our knowledge of the natural history of crossbills and of conifer seed production. We should anticipate that if the extent and frequency of logging increases, crossbills will continue to decline and will become increasingly vulnerable to environmental and demographic stochasticity and to losses to genetic variability (see Lande 1988).

Conclusions and Recommendations

Protecting nomadic populations of crossbills presents some of the same challenges confronted in the conservation of migratory species (Myers et al. 1987), where species often concentrate in small areas during their annual cycle. However, protecting nomadic species such as crossbills represents a more formidable challenge because critical habitats are more difficult to recognize since they may be used only once every several years, with different areas crucial during different years. Nonetheless, several practices would aid crossbills and other conifer seed–eating animals.

First, as a general rule, the amount of old-growth forest should be maximized simply because it is consistently the most productive for crossbills. In areas where little old growth remains (such as Siuslaw National Forest in coastal Oregon [Harris 1984]), second-growth should be protected and allowed to mature. Especially beneficial to the crossbills specialized on Douglas fir and western hemlock is the recent proposal to protect an additional 2.4 million ha of U.S. Forest Service and Bureau of Land Management lands in northern California, western Oregon, and western Washington to protect the Northern Spotted Owl (Thomas et al. 1990).

Second, rotation ages of managed forests need to be lengthened. In western North America, the proportion of lands logged and the rotation ages set by the U.S. Forest Service and Bureau of Land Management will have a profound impact on crossbill populations, because these two agencies control most of the federal forests required by crossbills (see Crumpacker et al. 1988). A similar arrangement exists in Canada, where even less old-growth forest is protected from logging than in the United States (for example, only about 17,600 ha of old-growth Sitka spruce–western hemlock forest were protected in British Columbia as of 1987 [Roemer et al. 1988; see also Beebe 1991]).

Third, mature trees should be left in cutover areas (as in partial cuts; see Franklin & Spies 1991). This will act to increase the proportion of the landscape containing trees of cone-bearing age. Furthermore, the remaining trees, released from competition, may then produce larger cone crops (see Fowells 1965). However, because cross-pollination and the number of full seeds per cone declines as mature tree density decreases (Smith et al. 1988), there will be a lower limit to tree density below which crossbills cannot be supported in the long term.

Fourth, because geographically separated areas often produce cone crops asynchronously, a geographically limited reserve of each forest type, (such as spruce-hemlock forest) will be inadequate for protecting nomadic populations of crossbills. Forest reserves should be located among as many distinct climatological regions as possible to avoid synchronous cone failures among all areas, and to minimize the risk from catastrophes (see, for example, Walters 1991). Tree seed zones, each of which represent different climates and elevations, represent such distinct regions (Schopmeyer 1974). Information on geographic variation (morphological or phenological) in conifers can also be used in determining favorable distributions of forests, because morphological variation is correlated with climate (such as ponderosa pine [Fowells 1965] and Douglas fir [Sorensen & Miles 1978]). The policy to protect forest tracts throughout the Northern Spotted Owl's range (Thomas et al. 1990) will reduce the likelihood of a complete cone failure in the Pacific Northwest.

Fifth, reserves should encompass productive forests. This will be difficult where commercial interests are powerful. For example, in the Cascade Range the most productive areas are low-lying valleys, of which few are protected (Norse 1990).

Last, aiding seed-eating animals will seem heretical to silviculturalists, who have invested a considerable amount of energy eradicating these animals (see Fowells 1965). However, by increasing rotation intervals, we enhance seed production and especially the occurrence of large cone crops. Because the greatest fraction and by far the greatest number of seeds remain uneaten during large cone crops (see Fowells & Schubert 1956), they are the most critical for natural regeneration. By increasing rotation ages (and the amount of old growth) we can increase the occurrence and extent of large cone crops and thereby protect the great diversity of conifer seed–eating animals (see Smith & Balda 1979).

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Farmington Daily Times

NEWS

'Hundreds of thousands, if not millions': New Mexico sees massive migratory bird deaths

Algernon D'Ammassa Las Cruces Sun-News
Published 2:41 p.m. MT Sep. 12, 2020 | Updated 2:56 p.m. MT Sep. 12, 2020

LAS CRUCES - Biologists from New Mexico State University and White Sands Missile Range examined nearly 300 dead migratory birds Saturday at Knox Hall on the university's main campus.

Over the past few weeks, various species of migratory birds are dying in "unprecedented" numbers of unknown causes, reported Martha Desmond, a professor at NMSU's Department of Fish, Wildlife and Conservation Ecology.

"It is terribly frightening," Desmond said. "We've never seen anything like this. ... We're losing probably hundreds of thousands, if not millions, of migratory birds."

In August, large numbers of birds were found dead at White Sands Missile Range and at the White Sands National Monument in what was thought to be an isolated incident, Desmond said.

After that, however, came reports of birds behaving strangely and dying in numerous locations in Doña Ana County, Jemez Pueblo, Roswell, Socorro and other locations statewide.

The affected birds have included warblers, sparrows, swallows, blackbirds, flycatchers, and the western wood pewee.

"A number of these species are already in trouble," Desmond said. "They are already experiencing huge population declines and then to have a traumatic event like this is – it's devastating."

On Saturday, Desmond was joined by Trish Cutler, a wildlife biologist at WSMR, and two NMSU students for an initial evaluation of the carcasses.

Desmond said her team also began catching and evaluating living specimens on Friday as residents find birds behaving strangely and gathering in large groups before dying.

"People have been reporting that the birds look sleepy ... they're just really lethargic," Cutler said. "One thing we're not seeing is our resident birds mixed in with these dead birds. We have resident birds that live here, some of them migrate and some of them don't, but we're not getting birds like roadrunners or quail or doves."

On the other hand, numerous migratory species are dying rapidly and it is not immediately clear why, although the cause appears to be recent. Desmond said the birds had moulted, replacing their feathers in preparation for their flight south, "and you have to be healthy to do that; but somewhere after that, as they initiated their migratory route, they got in trouble."

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The biologists guessed the cause might involve the wildfires ravaging the western U.S. and dry conditions in New Mexico.

"They may have been pushed out before they were ready to migrate," Desmond said. "They have to put on a certain amount of fat for them to be able to survive the migration. These birds migrate at night and they get up in the jet stream, and they might migrate for three nights in succession, they'll come down and they'll feed like crazy, put on more fat and go again."

The biologists noted that the majority of the dying birds are insectivores, but that seed eaters were sickening and dying as well.

The birds will be sent to the U.S. Fish and Wildlife Service Forensics Laboratory in Ashland, Ore. for further analysis. Desmond it could be weeks before results come back, and the findings could bear serious ecological implications.

"Over 3 billion birds have died since 1970. Insect populations are crashing, and this is just an unprecedented mortality," she said. "Climate change is affecting the abundance of insects, it's affecting the volatility of the fires, and the scary thing is this may be an indication of the future."

Algernon D'Ammassa can be reached at 575-541-5451, adammassa@lcsun-news.com or @AlgernonWrites on Twitter.

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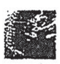

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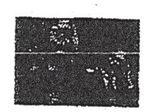

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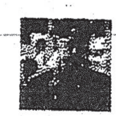


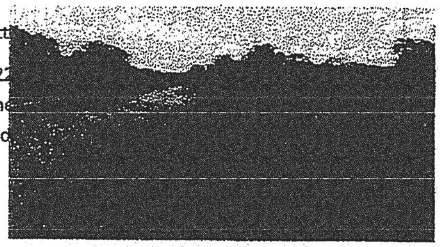
Tweets of the Week May 29-Jun

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.

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.

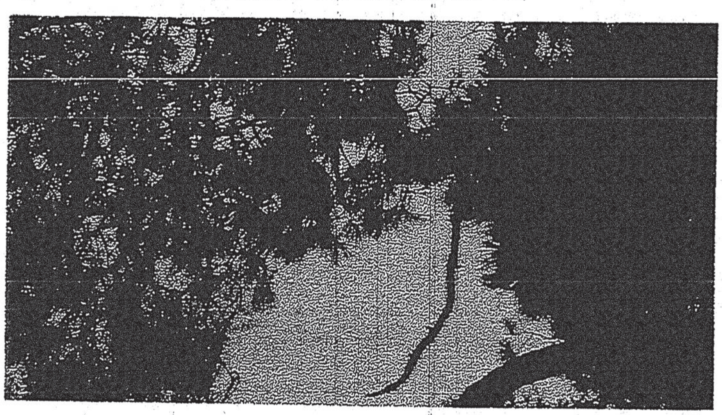
Cal Fire burning next to eagle nest, 2021. How much extra CO_2 is being emitted by extra equipment use and burning unnecessarily?

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.

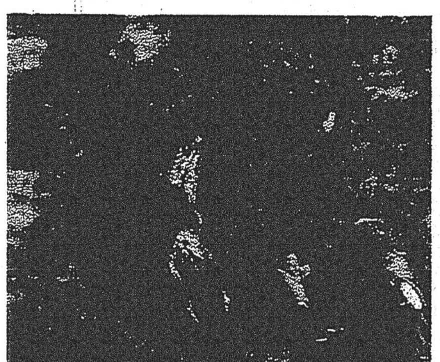
A State Wildlife Health Lab biologist wrote to us later that:

"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.

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A Handbook for Conservation and Management

by

David S. Dobkin

The High Desert Museum
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Schrasker B-84
GTT-B-115
Brewers 119
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92
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CASSIN'S FINCH

Carpodacus cassinii
Fringillidae

Summer, Permanent, or Winter Resident

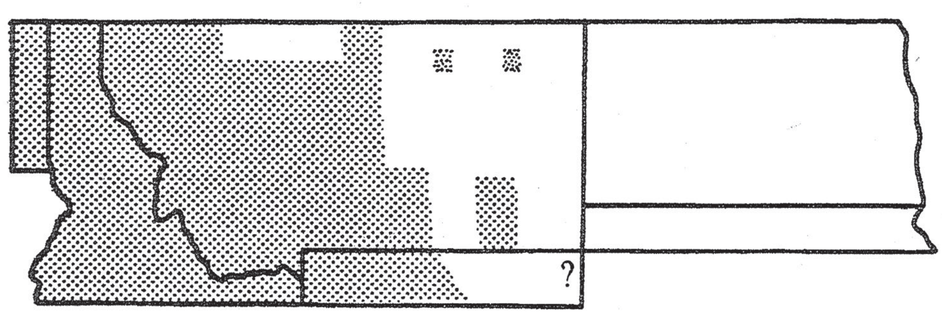
WINTERING AREA: 5

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.



The 2020 Almeda fire in southern Oregon decimated residential neighborhoods.

Wildfire Mitigation

The 2020 wildfire season set a record for acres burned in the U.S. since 1983, and fire season started early for 2021 with drought conditions continuing across most of the country. At this writing, the U.S. Drought Monitor shows much of the West experiencing extreme and exceptional drought along with temperatures of 4-15 degrees above normal. In addition to ramping up firefighting resources, the Forest Service, other federal agencies, state agencies, and local governments are responding to the wildfire threat with increased spending for mitigation projects.

California allocated more than $500 million on wildfire prevention efforts just for springtime projects. Congress is getting involved with various pieces of legislation, and President Biden's budget request calls for spending $1.7 billion "for high-priority hazardous fuels and forest resilience projects," an increase of $476 million. If we learned nothing else from last year's fires, it's that weather-driven fires are unstoppable, which raises questions. Will this increased spending on fire mitigation benefit our forests? Will it benefit at-risk communities?

Most of our readers are likely familiar with the root problem. A century of fire suppression produced dense, overgrown forests that have proven more susceptible to pests, disease, drought, and catastrophic wildfire. The commonsense solution would seem to be simply thinning the forests to reduce wildfire risks, but fire ecologists say the issue is more complicated than that. A 2008 report by Reinhardt, Keane, Calkin, and Cohen cautions against acting on misconceptions about fuel treatments and their use as "a panacea for fire hazard reduction. ... Given the right conditions, wildlands will inevitably burn."

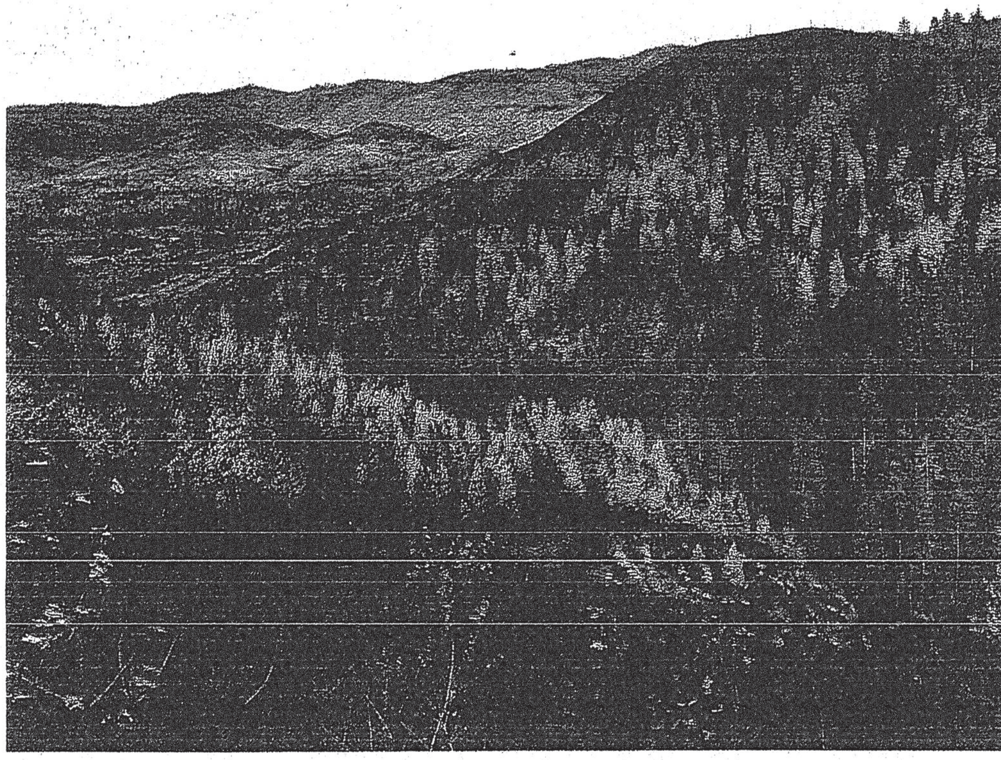
As the 2020 fires demonstrated, those conditions — high temps, low humidity, high wind — have become more common across the West, producing ever larger fires that account for the vast majority of acres burned each year. When fire conditions prevail, high winds carry embers for miles, jumping rivers, lakes, and fire lines.

Reinhardt et al. also note that these fires burn through areas that have been thinned. In fact, without follow-up treatments, thinning can increase the intensity of large fires. The lower density of trees allows winds to blow with less obstruction, and more sunlight on the forest floor dries the ground and encourages flammable shrubs and invasive plants to grow and spread more rapidly. Maintaining the desired conditions requires ongoing, labor-intensive management and, according to a 1994 report by W.L. Baker, may take up to seven treatments before conditions resemble pre-fire-suppression forests.

John Muir Project Director Chad Hanson, a forest ecologist, said a growing body of research suggests that removing trees doesn't protect forests from wildfire but may contribute to more intense wildfires. He and his fellow researchers have conducted multiple studies that support his opposition to logging and mechanical thinning of forests. A 2016 report that he co-authored concludes, "Forests with the highest levels of protection from logging tend to burn least severely."

Mark Finney, a Forest Service research forester, studies the physical processes of fire spread. He led a team that studied the Hayman fire, which was then the largest Colorado wildfire on record. Finney's team determined, "Fuel breaks and treatments were breached by massive spotting and intense surface fires. ... Extreme environmental conditions ... overwhelmed most fuel treatment effects. ... Suppression efforts had little benefit from fuel modifications." The team concluded that the primary objective of fuel treatments should be to make wildfire "less severe, rather than to reduce wildfire extent or make it easier to suppress."

Finney promotes restoring

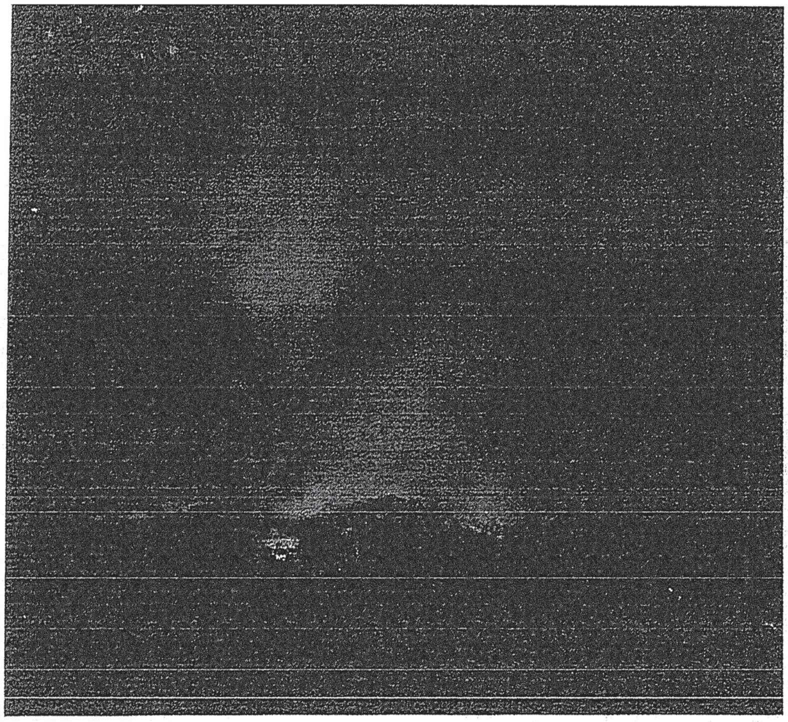


The Camp Fire in California, which destroyed the town of Paradise, burned more intensely in a previously logged area (foreground) than in an adjacent section of unlogged forest where mature trees survived the blaze (photo by Chad Hanson).

fire-prone forests to conditions that mimic the forest structure prior to fire suppression policies. For millennia, he points out, Native American communities in fire-prone regions used fires to manage the landscape, shaping forests and grasslands in ways that minimized the community risk from fires as well as the likelihood of high-severity fires. Because our frequent-fire forests have changed so much since fire was removed from the landscape, Finney advocates for "structural restoration" as a way of returning western U.S. forests to "something that is sustainable." The key, he asserts, is prescribed fire. "Drier forest types were sustained by periodic burning."

As the local tribes recognized, the forest needs repeated management, "not one and done," Finney remarked. "We get all wrapped around the axle because we're not considering the maintenance. The only way to maintain a forest in a low-hazard condition is through repeated burning ... but you can't introduce fire without some mechanical treatment first. ... You can't restore structure without mechanical means."

As Finney elaborates in a recent paper, "We believe that the primary goal of fuel treatment should be to create landscapes in which fire can occur without devastating consequences. Once these conditions have been achieved, wildfire need not be as vigorously suppressed and can itself play a role in maintaining these landscapes. Fuel treatments should not be used to reduce or eliminate fire



The 2020 CZU Lightning Complex fire burns in California's Santa Cruz Mountains. The fire grew to more than 85,000 acres and destroyed over 900 structures (photo by Inklein, Wikipedia).

from landscapes. Fuel treatment programs should ... encourage a return of fire to the landscape and improve the resilience and sustainability of U.S. ecosystems."

For Finney, the bottom line is that we're playing catch-up, and we've got a lot of catching up to do. "We've been ignoring this for a long time. We need to get millions of acres under a treatment regime, and we need to think at landscape scales. We need to strategically begin to introduce treatments onto the landscape to obstruct fires from traveling so far."

Finney contributed to a 2018 study addressing fuel loads in Sierra Nevada forests where severe drought "compounded by forest densification from decades of fire suppression" produced "unprecedented" tree mortality. The study concludes that wildfire severity "may be little affected" in the first decade following widespread tree mortality from bark beetles or drought. However, "extensive loading of large-sized woody fuels in future decades may contribute to dangerous mass fires. ... Such intense fires could prevent forests from becoming re-established."

Finney emphasizes that the combination of historical fire suppression combined with climate change create a lot of unknowns. "Our forests currently have high densities of susceptible trees. It's unprecedented. ... To do nothing would be very naïve. ... It's also important to have private landowners doing their part," which finally gets us to the second question: How much will fire mitigation efforts benefit our communities?

In his most recent paper, Finney observes, "Engineering solutions to reduce vulnerability of buildings and other infrastructure to wildfires face few technical obstacles," yet issues like cost and enforcement challenges "impede widespread adoption of changes in building design and materials."

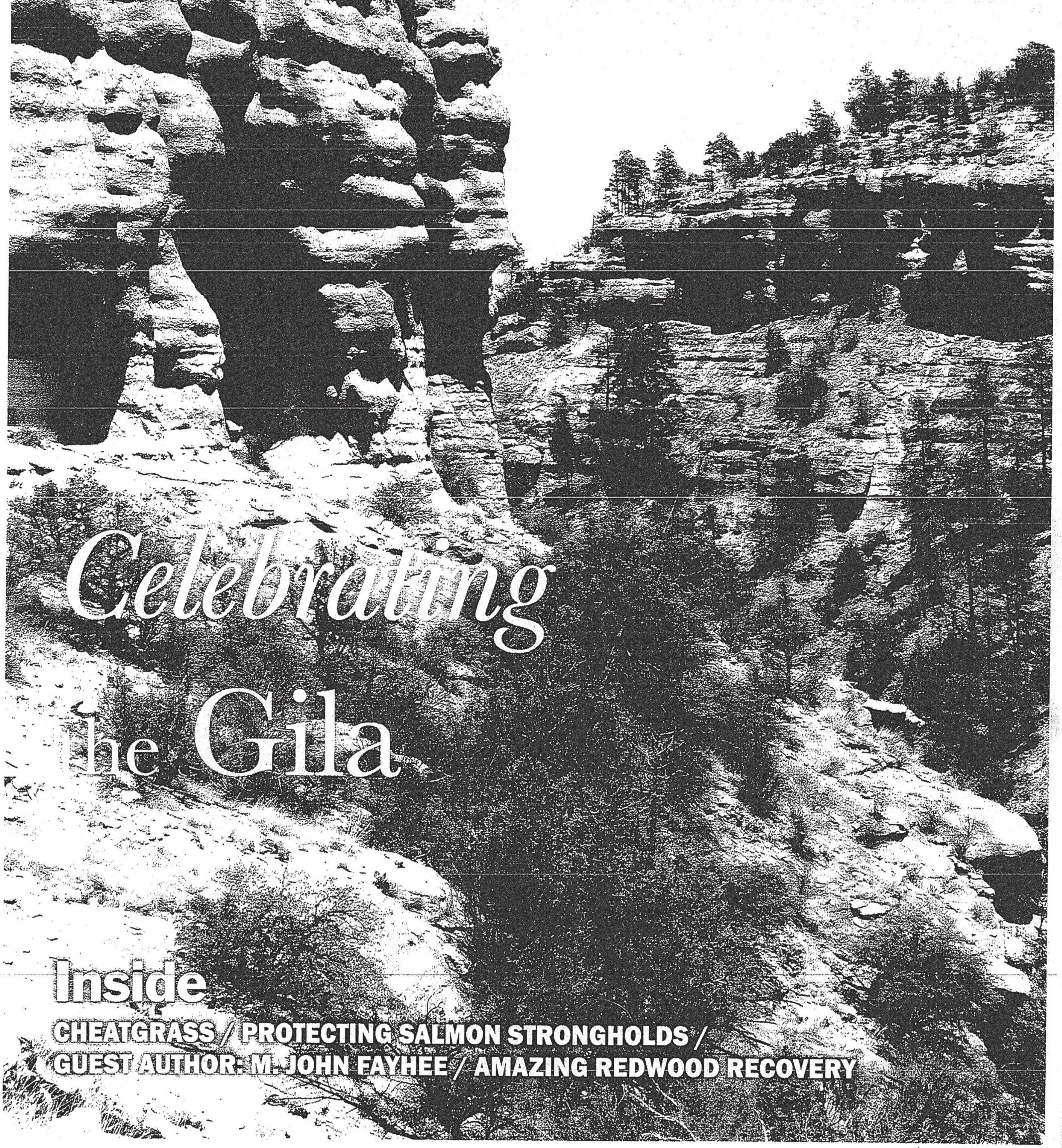
In this regard, Finney and Hanson agree. In fact, Hanson said fire mitigation efforts should start with homes and be limited to a 100-foot radius around each home. "We need to work from the home out to the forest." After fireproofing homes and other buildings by cleaning gutters, installing metal roofs, and so forth, Hanson emphasizes defensible space, especially the 30-foot radius closest to the building, where dry grasses and small trees should be removed. He questions the value of "vegetation management" beyond 100 feet from the house but believes that, with a relatively small investment, most homeowners could be 95 percent effective at saving their homes from wildfire.

Without doubt, the wildfire situation is complicated, largely because more than a century of fire suppression has taken our forests into uncharted territory, but it has become clear to ecologists that fire is an integral part of our forests. As Finney said, "Fire is an excellent ally, and we've not taken advantage of that, partially through fear. ... We're more vulnerable to fire than the native peoples who had none of our technology. They used it routinely, and they persisted here for upwards of 10,000 years."

A prescribed fire in Yosemite National Park removes flammable undergrowth to mitigate the risk of catastrophic wildfire.

FOREST NEWS

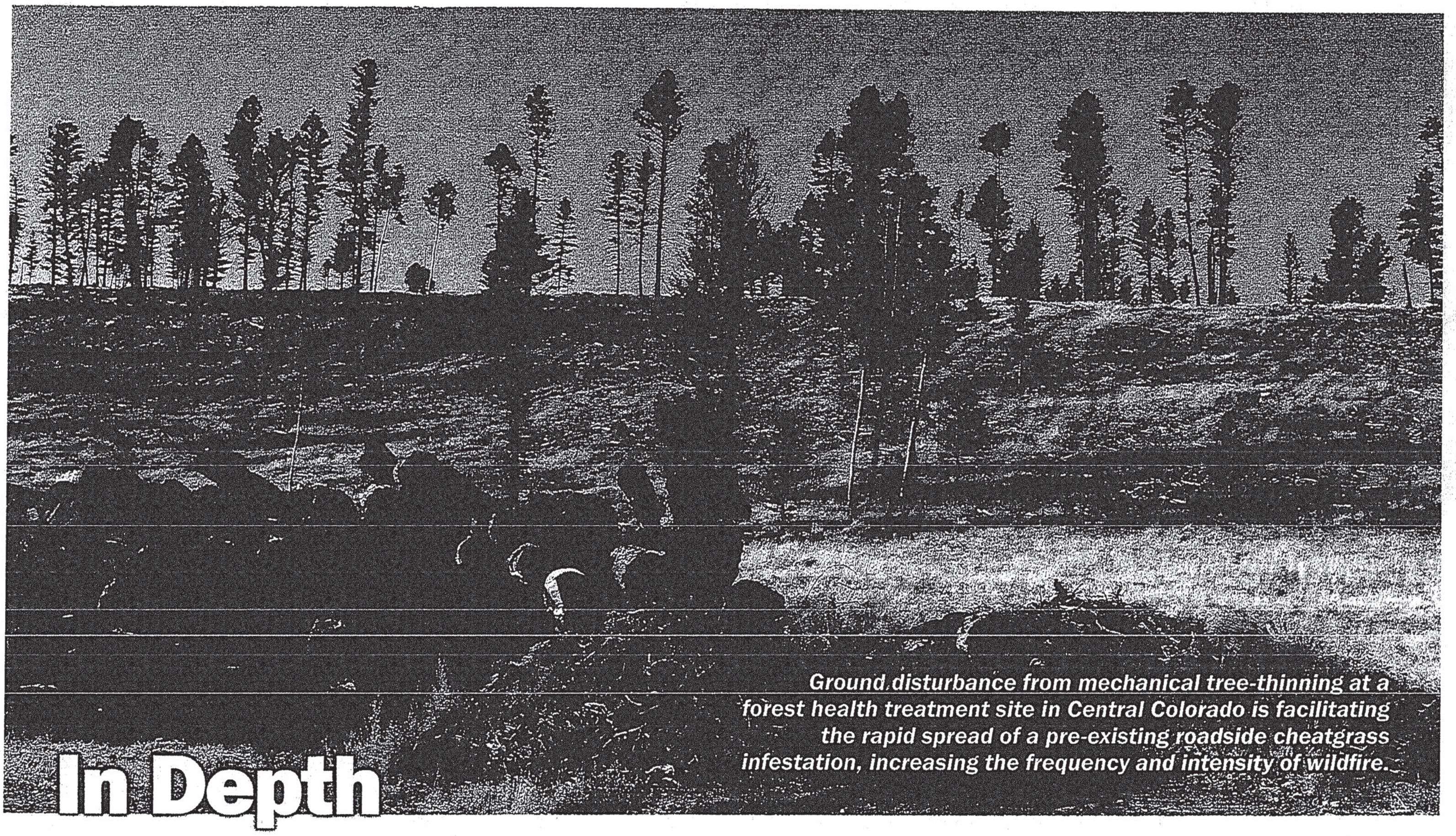
The Newsletter of Forest Service Employees For Environmental Ethics   Spring 2024



Celebrating the Gila

Inside

CHEATGRASS / PROTECTING SALMON STRONGHOLDS / GUEST AUTHOR: M. JOHN FAYHEE / AMAZING REDWOOD RECOVERY



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.

Cheatgrass

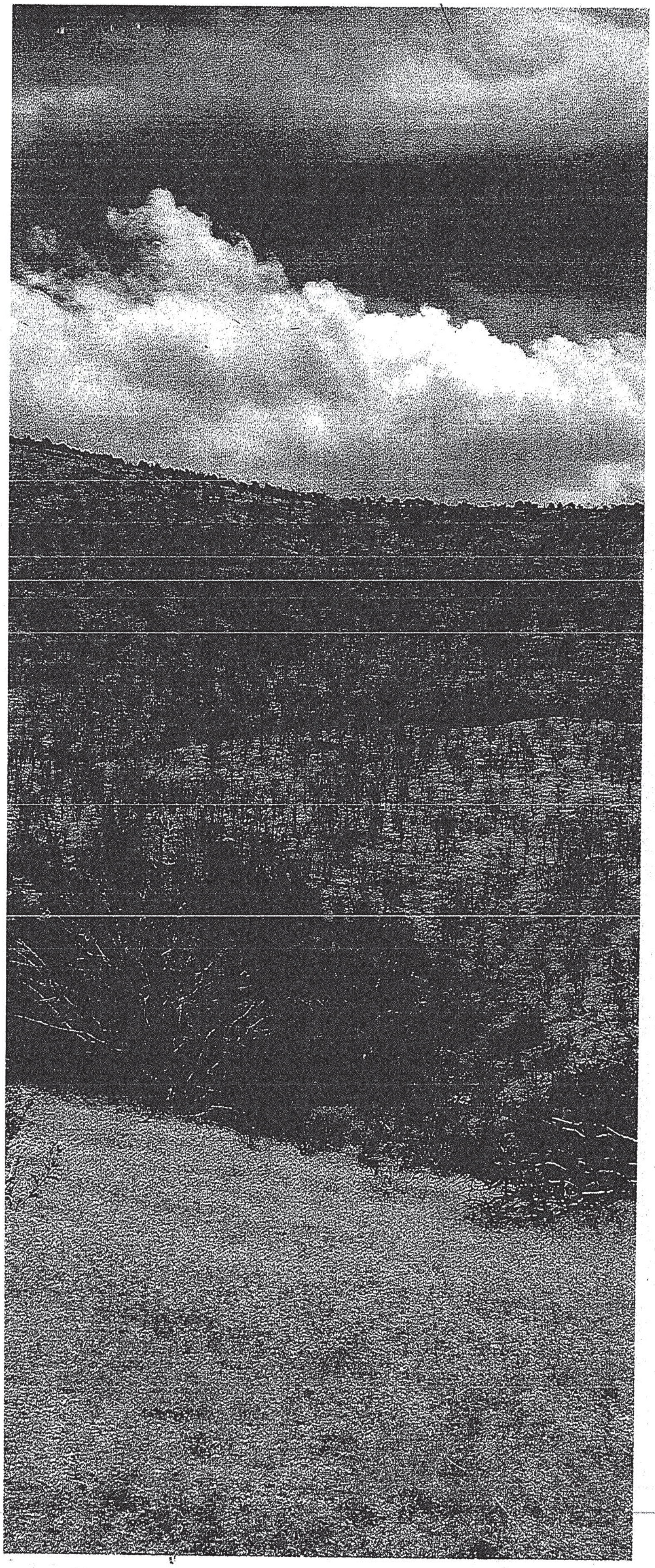
'One of the most significant ecological crises facing land managers in the arid West.'

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



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

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 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.

1988.
ODFW Nongame
Rept 87-3-02.

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

CHARACTERISTICS OF OLD-GROWTH FORESTS IN THE INTERMOUNTAIN REGION

COMPILED BY

RONALD C. HAMILTON
REGIONAL GENETICIST

APRIL 1993

U.S.D.A. - FOREST SERVICE

Standard Summary of Old-Growth Characteristics

Vegetative Series: Engelmann spruce, Subalpine fir, Grand fir
SAF Cover Type: Spruce - fir
Applicable Area: Intermountain Region
Site Productivity: N/A

Live trees

Main canopy			Variation in tree diameter	Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number
Warm/moister environments					
≥20 Utah ≥24 Idaho	≥25	≥220	≥2	Evidence	≥2
Cold/dry environment					
≥15	≥15	≥150 to 180	≥2	≥2-14	≥2
Alpine transition environment					
≥12	≥10	≥150 to 180	≥2	Evidence	≥2

Standard Summary of Old-Growth Characteristics

Vegetative Series: Engelmann spruce, subalpine fir, bristlecone pine, and whitebark pine
SAF Cover Type: Whitebark pine
Applicable Area: Intermountain Region
Site Productivity: N/A

SERAL AND CLIMAX

Live Trees

Main canopy			Variation in tree diameter	Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number
≥18	≥15	≥250	≥2	≥2-15	≥2

Dead Trees

Standing		Down	
DBH/ht ft	TPA	Diameter	Pieces/acre length in ft (min. length
≥15-10	≥5	≥20	5-8 ft

STANDARD SUMMARY OF OLD GROWTH CHARACTERISTICS

Vegetative Series: Douglas-fir, Grand fir, White fir, Engelmann spruce, and Subalpine fir
SAF Cover Type: Interior Douglas-fir
Applicable Area: Intermountain Region
Site Productivity: N/A

SERAL AND CLIMAX

Live Trees

Main canopy			Variation in tree diameter	Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number
Higher productivity sites					
24	≥15	≥200	≥2	Evident	≥2
Lower productivity sites					
18 (14)*	10 (5)**	≥200 (-25yr)	≥2	≥2-15	≥2

Standard Summary of Old-Growth Characteristics

Vegetative Series: Grand Fir
SAF Cover Type: Grand Fir
Applicable Area: Boise, Payette, Salmon NF's
Site Productivity: Moderate-High

Live Trees

Main canopy			Variation in tree diameter	Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number
≥24	≥15	≥200	≥2	Evidence	≥2

Dead Trees

Standing		Down	
DBH/ht ft	TPA	Diameter	Pieces/acre length in ft (min. length
20-20	≥2	≥12	2-25

STANDARD SUMMARY OF OLD GROWTH CHARACTERISTICS

Vegetative Series: Blue Spruce, Subalpine fir, and White fir
SAF Cover Type: Blue Spruce
Applicable Area: Targhee, Bridger-Teton, Caribou, Wasatch-Cache, Uinta, Ashley, Fishlake, Manti-LaSal, and Dixie National Forests
Site Productivity: N/A

Live Trees

Main canopy			Variation in tree diameter	Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number
≥16	≥10	≥250	≥3	≥1-14	≥3

Dead Trees

Standing		Down	
DBH/ht ft (in)	TPA	Diameter (in)	Pieces/acre length in ft (min. length
3-10	0-1	3-10	Infrequent

Standard Summary of Old-growth Characteristics

Vegetative Series: Quaking Aspen
SAF Cover Type: Aspen
Applicable Area: Southern Idaho, N. Wyoming, Utah, and Nevada
Site Productivity: N/A

Live Trees

Main canopy			Variation in tree diameter		Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number	
≥12	10 dry 20 mesic	100	≥2	N/A	≥2	

Dead Trees

Standing		Down	
DBH/ht ft	TPA	Small diameter (in)	Pieces/acre length in ft
≥10-15	2	8	≥10-10

Standard Summary of Old-Growth Characteristics

Vegetative Series:	Douglas-fir, Engelmann spruce, Subalpine fir, Grand fir (rare)
SAF Cover Type :	Lodgepole pine
Applicable Area:	Intermountain Region, except Toiyabe National Forest
Site Productivity:	N/A

SERAL AND CLIMAX

Live Trees

Main canopy			Variation in tree diameter	Tree decadence	Tree canopy layers
DBH*	TPA*	Age*	6-in Classes	TPA-DBH	Number
≥11	≥25	≥140	≥2	≥2-11	2

Dead Trees

Standing		Down	
DBH/ht ft	TPA	Diameter	Pieces/acre length in ft (min. length
≥11	5	≥11	≥50-≥8

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WILDLIFE MONOGRAPHS

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ECOLOGY OF BOREAL OWLS IN THE NORTHERN ROCKY MOUNTAINS, U.S.A.

by

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AND EDWARD O. GARTON

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entrances averaged 102 ± 12.41 mm high (range 64–150 mm) and 95 ± 11.89 mm wide (range 56–148 mm). Nest cavity condition ranged from recently constructed cavities without a feces layer (indicating no prior nesting by woodpeckers or other birds) to old cavities with a deep layer of dried feces, cone scales, and other debris. Inside, the cavities ranged from 7 to 50 cm deep ($\bar{x}$ = 31 ± 7.61 cm) and from 15 to 26 cm diameter ($\bar{x}$ = 19 ± 2.11 cm). The tree diameter at the cavity averaged 41 ± 5.21 cm (range 26–61 cm). Tree dbh averaged 64 ± 11.02 cm (range 33–112 cm).

Excluding nests in nest boxes, owls nested in ponderosa pine 10 times (53%), aspen 7 times (37%), and once each in Douglas-fir (5%) and Engelmann spruce (5%). Ten nests occupied snags, including 8 ponderosa pine, 1 aspen, and 1 Douglas-fir. Snag condition included 3 old branchless snags >11 m tall, 2 hard snags with sloughing bark and only large branches remaining, and 5 young snags with bark and complete limbs. Among nests in live trees, all but 2 cavities occurred in an open area on the tree bole; distance to foliage above the cavity averaged 3.8 ± 1.67 m (minimum 0.3 m). Over 75% of the cavities in live trees had no foliage below the cavity. For those that did, the minimum distance to foliage below was 0.6 m.

The owls chose relatively high cavities, averaging 12.7 ± 2.98 m and ranging from 6 to 25 m. Cavity height averaged 51% of tree height. Boreal owl nests in snags or trees with multiple cavities always occupied 1 of the uppermost cavities suggesting a preference for high nest sites.

The forest immediately around nest trees had an open structure. Density of 2.5–23-cm-dbh trees within a 0.01-ha plot around the nest tree averaged 398 ± 162/ha (range 0–1,482) (Table 5). The density of trees at nests was 3 times lower than the average at winter roost sites. Density of trees larger than 23.1-cm dbh averaged 212 ± 86/ha, similar to average densities measured at winter roost sites.

Nest sites occupied forest stands in 3 habitat series (based on Steele et al. 1981). We found 17% of nest sites in Engelmann spruce (*Picea engelmannii*) series, specifically the Engelmann spruce–common horsetail (*Equisetum arvense*) habitat type; 39% in the subalpine-fir series, specifically subalpine fir–twisted stalk (*Streptopus amplexifolius*), subalpine fir–bluejoint (*Calamagrostis canadensis*), subalpine fir–beargrass (*Xerophyllum tenax*), and subalpine fir–grouse whortleberry (*Vaccinium scoparium*) habitat types; and 44% in the Douglas-fir habitat series, specifically Douglas-fir–common snowberry (*Symphoricarpos albus*), Douglas-fir–pinegrass (*Calamagrostis rubescens*), and Douglas-fir–elk sedge (*Carex geyeri*) habitat types. The slope at the nest ranged from flat to 49%, averaging 28 ± 6%. Nest trees were distributed relatively evenly from bottoms

Table 5. Forest structure at 19 different nest trees used by boreal owls in the Frank Church River of No Return Wilderness during 1984–88. Tree densities are reported for 2-concentric circular plots—an inner circle 5.2-m radius and an outer "donut" extending from 5.2 m to 11.4 m.

Site characteristic	$\bar{x}$	±95% CL
Tree density (No./ha)		
Inner plot		
2.5–7.6-cm dbh	174	111.9
7.7–15-cm dbh	98	48.1
15.1–23-cm dbh	114	60.1
23.1–38-cm dbh	136	73.7
38.1–68-cm dbh	60	42.5
>68-cm dbh	11	15.6
Outer plot		
2.5–7.6-cm dbh	242	107.3
7.7–15-cm dbh	178	70.1
15.1–23-cm dbh	124	49.5
23.1–38-cm dbh	130	56.6
38.1–68-cm dbh	51	25.0
>68-cm dbh	10	7.8
Snag density (No./ha)		
2.5–38-cm dbh	79	42.2
>38-cm dbh	10	11.5
Basal area (m^2/ha)	33.7	3.62
Canopy cover (%)	55	7.7
Topographic features		
Distance to water (m)	201	98.9
Slope (%)	28	5.8

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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 Canada (Godfrey 1986). South of the continuous transcontinental band, disjunct populations occur in the

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Broad Scale Predictions. Predicting the response of Boreal Owls to differing landscape scale patterns is more difficult. The lack of information on patterns of Boreal Owl abundance at the landscape and broader scales precludes extensive predictions at broad scales. I would argue that a primary focus of adaptive management approaches should be at this scale.

The issue of fragmentation seems to dominate much of the discussion of landscape scale impacts, so preliminary predictions regarding fragmentation may be useful in stimulating inquiry. In referring to potential response to fragmentation, I explicitly separate the influence of habitat loss from the influence of increased landscape heterogeneity. Fragmentation effects result from the process of changing the characteristics of the landscape mosaic and must be considered after eliminating the direct influence of reducing habitat area.

The high mobility and the extensive areas used on a daily basis by Boreal Owls suggests they may react to fragmentation differently from passerines. For instance, timber harvest of 30% of a basin through clear-cutting mature lodgepole pine (*Pinus contorta*) in 1–5 ha patches dispersed throughout the area may not significantly reduce habitat quality if the remaining forest is dominated by mature and older spruce-fir forest. The forests used by Boreal Owls exhibit a patchy mosaic under natural disturbance (Knight 1994). In a natural forest mosaic, owls move between distant patches on a daily basis (Hayward et al. 1993). This hypothesis assumes that timber harvest would not significantly reduce small mammal populations in the unharvested stands.

Aside from fragmentation, it is important to consider the impact of harvest schemes that target different forest types: aspen, lodgepole pine or old spruce-fir forests. I hypothesize that the negative impacts of any stand replacement harvest scheme will be decreased if stands of mature and older spruce-fir or aspen forest remain dispersed throughout the landscape.

Predicting the consequences of management at the broadest spatial scales is challenging. Conservation strategies at the regional scale should focus on maintaining the continuity of Boreal Owl metapopulations. This involves identifying subpopulations and landscapes that likely play key roles in the persistence of owls within the region and neighboring regions. These subpopulations would receive special attention to assure that management actions either favored the owl or did not negatively impact the subpopulation. Spatial modeling and good information on dispersal will be necessary to make sound management predictions at this scale.

Thermal Cover

STRATEGIES TO APPROACH FOREST MANAGEMENT FOR BOREAL OWLS

I began this discussion by emphasizing the extent of uncertainty in our understanding of Boreal Owls and noted the substantial geographic variation in Boreal Owl ecology across North America. In combination, these factors produce a discouraging management environment where predictions must be made tentatively. Therefore, the response of Boreal Owls to forest management was framed as a series of hypotheses to be tested and likely only testable through adaptive management. Despite the degree of uncertainty and the extent of geographic variation, I believe some general points can be made concerning approaches to forest management and planning for Boreal Owls.

Limiting Factors. Site-specific forest management for Boreal Owls must consider the factors most likely limiting the population in a particular setting. Thermal stress likely limits the elevation distribution of Boreal Owls in the central and southern Rocky Mountains. Therefore, availability of cool microsites, which often occur in mature and older forests, may be important in many regions.

The availability of nest cavities and prey likely limit populations of Boreal Owls in different situations. In regions with few or no Pileated Woodpecker (*Dryocopus pileatus*) or Northern Flicker (*Colaptes auratus*) cavities, nest-site availability will limit Boreal Owl abundance. Even within the geographic range of Pileated Woodpeckers, the absence of these woodpeckers at higher elevations may limit Boreal Owl abundance (Hayward et al. 1993). If cavity availability limits Boreal Owl populations, management of primary cavity excavators as well as the forest processes that support large snags will influence Boreal Owls.

In some forests, cavities are abundant and prey availability may play a strong role in Boreal Owl population dynamics. It is unclear whether absolute abundance or variation in prey populations is more important in owl population regulation. However, small mammal populations appear to be

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Rocky Mountains extending from Canada to northern New Mexico (Palmer and Ryder 1984, Hayward et al. 1987, Whelton 1989, Stahlecker and Rawinski 1990). Throughout this broad distribution the owl occurs in a variety of boreal and subalpine forests: conifer and mixed forests in Canada (Bondrup-Nielsen 1978), transition forests in Minnesota (Lane 1988) and subalpine forests in the Rockies (Hayward et al. 1993). Boreal Owl populations are intimately linked to the composition, structure and dynamics of these forests (Hayward and Hayward 1993, Hayward and Verner 1994). Therefore, the distribution and abundance of Boreal Owls may be strongly influenced by forest management practices.

How do populations of Boreal Owls respond to alternative approaches in forest management? In this paper I provide a perspective on the potential impacts of forest management on the owl. Forest management represents the human activity most likely to influence the long-term distribution and abundance of Boreal Owls. Among Holarctic raptors, Boreal Owls, at least in the North American Rockies, may represent the species whose ecology is most universally tied to the forest system. An understanding of the potential response of Boreal Owls to various changes in forest structure and dynamics is a critical step in designing management.

In the U.S., management of Boreal Owls has become an important task on public lands. Four National Forest Regions and the Superior National Forest which represent most of the species' range south of Canada have designated the Boreal Owl as a "sensitive species." Within the National Forest System, sensitive species are plants and animals whose population viability is identified as a concern by a Regional Forester. Sensitive species require special management and programs are underway to develop management plans for Boreal Owls (J. Friedlander pers. comm.).

Unfortunately, the knowledge needed to develop a sound management strategy may be lacking (Hayward 1994a). To date, only four major published investigations from North America provide the ecological basis for management planning (Bondrup-Nielsen 1978, Palmer 1986, Hayward et al. 1992, Hayward et al. 1993). None of these investigations represent experimental approaches to ecological questions, none of these was designed to directly address forest management issues and all extended for 4 yr or less—a temporal scale insufficient to address important issues in forest management or the ecology of a long-lived vertebrate. The Boreal Owl in North America represents a classic example of uncertainty in wildlife management.

Over 14 yr ago, Romesburg (1981) admonished wildlife managers for the development of management plans built upon unreliable knowledge. Management built on poor science leads to a loss of credibility and poor resource management. Current understanding of Boreal Owl ecology and biology is poor. Management built on this foundation alone will invite criticism and loss of credibility. Recently though, Murphy and Noon (1991) discussed an approach to deal with the inherent uncertainty associated with management of a forest raptor, the Spotted Owl (*Strix occidentalis*). They advocate applying the hypothetico-deductive approach to management. Through a rigorous assessment of the assumptions that form the basis of management, they reduce the uncertainty clouding an evaluation of the efficacy of various management options. Walters' (1986) adaptive management concepts are another attempt to deal with the uncertainty that accompanies wildlife management.

My perspectives on forest management for Boreal Owls is guided by a philosophy that combines the concepts of the hypothetico-deductive method and Walters' adaptive management to develop management in the face of poor knowledge. Therefore the statements I make regarding the potential response of Boreal Owls to forest management, must be regarded as hypotheses. I would advocate the testing of these hypotheses through multi-scale experiments in the spirit of adaptive management.

To provide a perspective on forest management and Boreal Owls, I will review the conservation status of Boreal Owls in North America including a discussion of trends in forest management, examine our understanding of the ecology of Boreal Owls as it relates to the owl's potential response to forest management, present some hypotheses concerning how different forest management approaches may influence Boreal Owls on different geographic and temporal scales and provide some ideas concerning strategies to approach forest management for Boreal Owls.

The perspective I present is biased by the geographic limits of my field experience with Boreal Owls—I have worked in the Rocky Mountains. More important, the literature on Boreal Owl ecology in North America is limited. Literature from

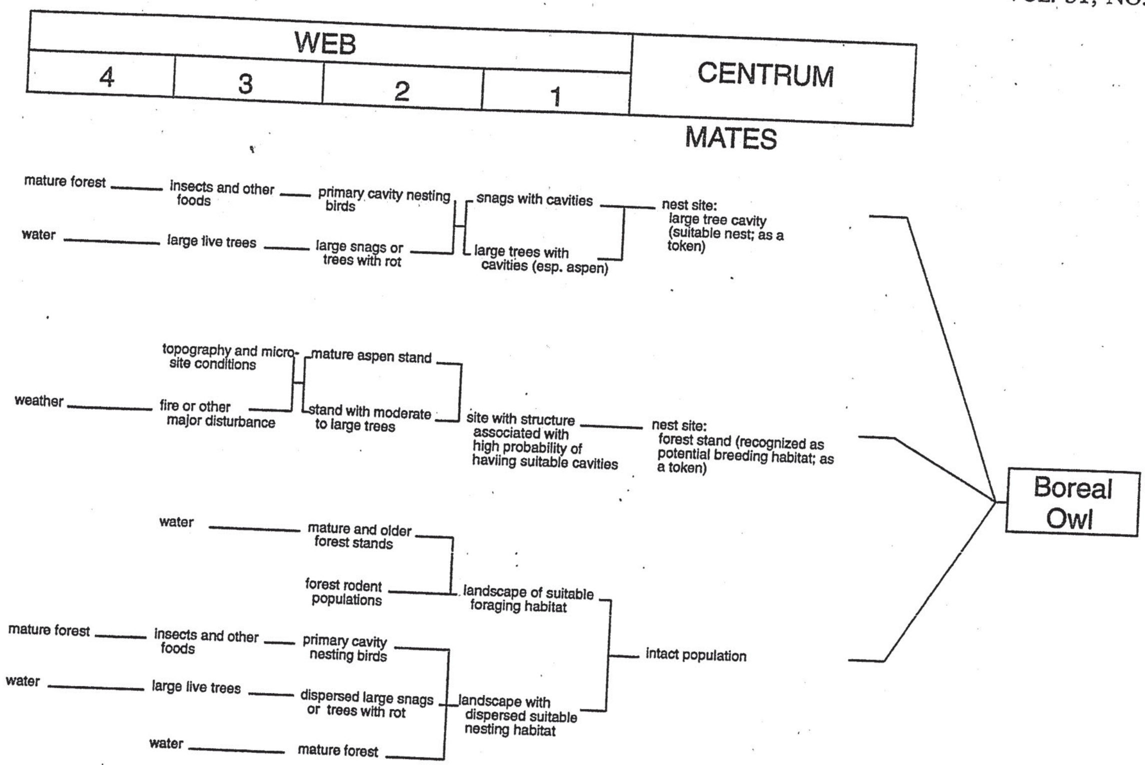

Figure 1. Envirogram (Andrewartha and Birch 1984) illustrating the relationship between Boreal Owls and specific components of the forest system. This portion of a larger envirogram (Hayward 1994b) focuses on Boreal Owl nesting ecology.

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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Europe significantly broadens our understanding of the species. However, the ecology of Boreal Owls differs geographically within Europe (Korpimäki 1986) and within North America (Hayward et al. 1993). I suspect that the response of Boreal Owls to forest management differs between the Old and New Worlds and geographically within both.

Although our understanding of Boreal Owl ecology in North America is limited to three forest systems (one in each of northcentral Canada, central Idaho and northern Colorado), the Boreal Owl appears to occupy a variety of forest types. These forests range from deciduous and mixed forests to subalpine conifer forests (Meehan and Ritchie 1982, Palmer 1986, Lane 1988). The dynamics of these forests differ substantially due to differing patterns of forest growth and different disturbance regimes (Knight 1994). Likewise, Boreal Owl population dynamics, relationships with primary cavity nesters and relationships with prey populations differ among these forest types (Hayward 1994b). Therefore, the response of the owl to alternative forest management patterns almost certainly differs geographically. Any forest management scheme must be cognizant of the differences among the forest systems.

STATUS OF BOREAL OWLS IN NORTH AMERICA

Trends in population abundance or trends in habitat conditions are often used to assess status (Anderson 1991). In 1994, the U.S. Forest Service published an assessment of Boreal Owl status (Hayward and Verner 1994). That document concluded that Boreal Owls were not in immediate peril throughout their range but that over the long-term and in local areas over the short-term, Boreal Owls likely face significant conservation problems in the absence of conservation planning. To reach this conclusion the assessment examined evidence concerning trends in the distribution and abundance of the owl and the habitat relationships of the owl.

Distribution and Abundance of Boreal Owls. Little evidence exists to assess changes in the distribution of Boreal Owls in North America. Prior to 1979 the owl was not recognized as a breeding bird south of Canada (Eckert and Savaloja 1979). Since then numerous published reports have extended the recognized range of Boreal Owls in North America (Palmer and Ryder 1984, Hayward et al. 1987, Whelton 1989). Today, evidence exists for breeding populations throughout the Rocky Mountains south to southwestern Colorado and northern New Mexico (Stahlecker and Rawinski 1990, Stahlecker and Duncan 1996). Do these records indicate an extension of the species range?

I suggest that the actual distribution of Boreal Owls has not changed recently, but our knowledge of distribution has increased because of an increase in survey effort. Historical records indicate that Boreal Owls were recorded in the western United States but not recognized as breeding. A closer look at the literature indicates that Boreal Owls were documented in Colorado for nearly 100 yr (Ryder et al. 1987). Despite the occurrence of Boreal Owls in the western U.S., checklists and field guides did not list the species even after breeding populations were documented in 1983. Biologists in Europe also located new populations of Boreal Owls during the past three decades and attributed these to increased interest in the species (Cramp 1977).

Direct evidence concerning trends in Boreal Owl abundance is completely lacking for North America. Breeding populations of Boreal Owls were only recently documented throughout most of the species' range in the U.S. Studies in North America generally have not focused on demography, precluding any assessments of trend in the near future. I am aware of only two populations (one in Idaho and one in Montana) that have been sampled using methods that will facilitate rigorous assessment of trends within the next 5 yr (Hayward et al. 1992). The prospects for assessing trends in the near future appear bleak.

Abundance and Distribution of Important Habitats. Information on trends in condition of forest habitats used by Boreal Owls offers an indirect method to infer population trends. Gathering and summarizing the necessary information at a broad geographic scale is not feasible for this paper. Furthermore, most statistics on timber harvest do not include the information necessary to evaluate the pattern in distribution and abundance of important forest types. For instance, stand-replacement harvests (clear-cuts) create stands without habitat value for Boreal Owls for a century or more, while partial cutting may leave stands with high habitat value if dominant trees are not removed. An objective evaluation of habitat trends relies not only on knowledge concerning recent timber harvest but knowledge on succession of lands that experienced large disturbance events 100–150 yr ago.

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culties in predicting future harvest. As the availability of timber has declined on lower elevation forest lands in western North America, focus is shifting to high elevation spruce-fir forests used by Boreal Owls. Furthermore, the rules regulating timber harvest in the U.S. have changed recently regarding salvage after fire (U.S. Public Law 104-19). The consequences of these changes are difficult to predict. As they might say in a prospectus, the extent of future harvest and therefore impact on Boreal Owl habitat may not be related to past trends.

Summary. There is little direct evidence concerning trends in North American Boreal Owl populations. In a Boreal Owl conservation assessment (Hayward 1994c), evaluation of habitat use patterns, life history and trends in habitat condition were used to infer owl trends.

HABITAT RELATIONSHIPS OF BOREAL OWLS

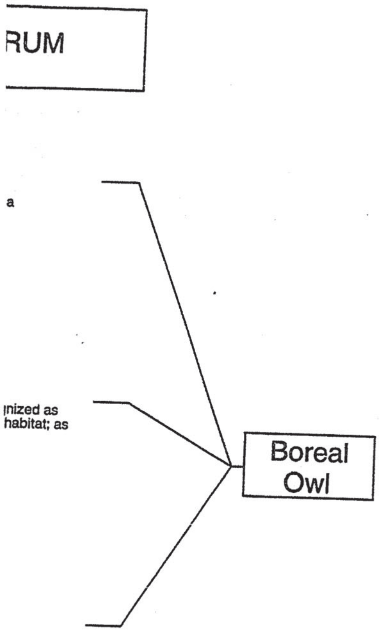
I review the habitat relationships of Boreal Owls. My goal is to establish the relationship between the owl and the forest to form hypotheses concerning the potential response of Boreal Owls to forest management.

Habitat relationships of Boreal Owls and habitat relationships of principal prey species will, in large part, dictate the potential response of Boreal Owls to timber management. The realized impact of forest management in a particular situation will be determined by the interaction of habitat relationships of the owl and prey populations mediated by those factors currently limiting population growth. Nesting habitat conditions (especially cavity availability), prey availability (winter and summer) and microclimatic conditions related to owl thermoregulation likely limit the distribution and abundance of Boreal Owls in different populations (Hayward 1994b). Management that focuses on these limiting factors, after examining evidence suggesting which factor may be most critical in a particular setting, will most effectively target management actions.

As I have emphasized, the ecology of Boreal Owls varies geographically. For instance, daily and annual movement patterns, relationship with principal prey populations, population stability and limiting factors vary from the boreal forests of Canada to southern New Mexico (Hayward et al. 1993). Despite this variation, Boreal Owls are forest owls throughout their range and their ecology is linked to forest habitats with particular structural characteristics. I also consider nesting, roosting and foraging habitat separately because each of these may be limiting in different management settings. I will review the evidence describing the link between forest conditions and Boreal Owl populations. In my review I move from fine scale habitat characteristics to more broad scale relationships.

Fine Scale Habitat Relationships. *Nesting habitat.* The requirement for a large tree cavity constrains the range of sites used by Boreal Owl for nesting habitat. As secondary cavity nesters, boreals are intimately linked with the organisms and processes associated with formation of large tree cavities. An envirogram (Andrewartha and Birch 1984) emphasizes the linkage between forest structural conditions, primary cavity nester populations (woodpeckers), forest insects and pathogens (Fig. 1). The elements of the centrum relate directly to the owl while the web depicts components of the system important to maintaining the centrum. Elements of this envirogram are forest characteristics associated mainly with the presence/absence of suitable nesting cavities.

Beyond cavity availability, observations in the Rocky Mountains suggest that forest structural characteristics are important in nest-site selection. In Idaho, comparisons of forest structure at nest sites and random sites indicated use of stands with mature and older forest structure. Forest structure at nest sites differed from the random sample (101 sites) of available forest. Used sites occurred in more complex forest, with higher basal area, more large trees and less understory development than available sites (Hayward et al. 1993). Also in Idaho, a small nest-box experiment evaluated whether choice of nest sites is driven solely by cavity availability or if forest structure per se is important when a range of alternatives are available (Hayward et al. 1993). In this experiment nest boxes were hung in three forest types that differed significantly in structural characteristics. Owls used boxes in two forest types with complex structure (e.g., multiple canopy layers, many tree size classes) but did not use boxes in the forest type with a more simple structure (e.g., single canopy layer, more uniform tree diameters). Based on our observations I hypothesize that forest structure is important in an indirect way. Owls first search for nest sites in forests of a particular structure because the probability of finding cavities is highest in those types. So selection of old forest for nesting may be



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in mature forests facilitating access to small mammals during critical winter periods (Sonerud 1986, Sonerud et al. 1986). In Idaho, mortality and significant movement events most often occurred during warm winter periods when snow crusting became severe.

An envirogram further emphasizes the link between Boreal Owl foraging habitat and particular features of the forest, especially features linked with mature forests (see Hayward 1994b). The envirogram illustrates the indirect tie between Boreal Owl fitness and abundance of lichen, fungi and *Vaccinium* ground cover—all of which can be influenced by various forest management practices.

The evidence regarding habitat use for nesting, roosting and foraging in the Rockies suggests that at a fine scale, Boreal Owls rely on particular characteristics of mature and older forests. This relationship suggests that forest management at the level of stands will likely influence abundance of Boreal Owls.

Landscape Scale Habitat Relationships. Analysis of patterns of Boreal Owl abundance in relation to landscape patterns is not available for North America. Indirect evidence from Europe and North America suggests that Boreal Owls differentiate among forest habitats at the landscape scale. Our observations of owls in Idaho suggest that landscapes dominated by mature spruce-fir forest or those with mature spruce-fir juxtaposed with mature larch (*Larix sp.*), ponderosa pine (*Pinus ponderosa*) or aspen (*Populus tremuloides*) sites will have the greatest abundance of boreals (Hayward et al. 1992, 1993). In other words, an interspersion of forests that generally support high density of cavities in mature spruce-fir forest will provide quality habitat.

More direct evidence from Europe supports the notion that landscape scale forest cover influences Boreal Owl density and productivity. As the proportion of Scotch pine (*Pinus sylvestris*) forest decreased and the proportion of Norway spruce forest (*Picea abies*) and agricultural land increased, quality of territories (those with more frequent nesting) increased (Korpimäki 1988). The conclusion that territories with spruce forest and agricultural land (in small patches) were the highest quality habitat was corroborated by evidence on breeding frequency and clutch sizes.

Regional Scale Habitat Relationships. At very broad geographic scales, distribution patterns of Boreal Owls may also have important implications for management. In portions of the boreal forest, distributions of Boreal Owls may be quite continuous. Along the southern and northern borders of the boreal forest and in the Rockies, the owl may occur in an interesting geographic pattern which likely results in a strong metapopulation structure (Hayward et al. 1993). In Idaho, patches of suitable habitat occur throughout the mountainous landscapes in a wide range of patch sizes (Fig. 2). Assuming that subpopulations of owls occupy habitat as hypothesized in Figure 2, the metapopulation structure of the owl in the region is a complex mix of subpopulations. Because of this structure, management of forest at the scale of individual national forests may have important implications for neighboring national forests over a broad geographic region.

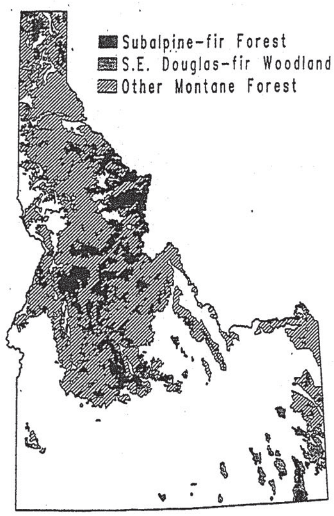

Figure 2. Pattern of potential Boreal Owl habitat in Idaho suggesting the distribution of a portion of the metapopulation extending along the Rocky Mountains. Potential habitat is defined as forested sites in the subalpine-fir zone throughout the state and Douglas-fir woodland in southeastern Idaho. Other montane forests are not considered potential habitat (adapted with permission of *Wildl. Monogr.* from Hayward et al. 1993).

HYPOTHESES: BOREAL OWL RESPONSE TO FOREST MANAGEMENT

Stand-Replacement Harvest. The importance of mature forest to Boreal Owls for nesting, roosting and foraging suggests that the short-term impact of stand-replacement harvest (clear-cut) will be negative. Open habitats as well as young, even-age forests provide few resources for Boreal Owls. Furthermore, these habitats generally do not enhance habitat for woodpeckers or small mammals. Large clear-cuts appear to provide no resource values for Boreal Owls except along edges where owls may capture prey (Hayward 1994b). However, impacts will depend upon the size and spacing of cuts and the forest type being harvested. Furthermore, long-term impacts may not parallel short term response.

I hypothesize that small, patch clear-cuts implemented with long rotations may not negatively impact Boreal Owl habitat over the short- or long-term. Boreal Owls generally attack prey within 30 m of a perch (Hayward et al. 1993), so most of a 1–3 ha patch cut will be accessible for foraging. Furthermore, in small patch cuts, ground cover, which could reduce prey availability, often does not change significantly from that found under the forest, snow crusting affects only a small proportion of a small forest opening and small patch cuts emulate, to some extent, the landscape structure of mature spruce-fir forests (Knight 1994). In cases where small patch cutting is employed, I hypothesize that potential negative impacts will be reduced if the patch cutting is concentrated in a portion of each watershed rather than dispersed throughout entire watersheds and mature forest remains in the matrix between cuts.

Larger clear-cuts in conifer forest most often will reduce habitat quality for 100 to 200 yr. However, clear-cutting of aspen may be important in maintaining the long-term availability of cavities in some systems. In many forest systems aspen is a pioneer species that is lost through succession (DeByle and Winokur 1985). Restoration of aspen forests through silviculture may be an important management tool to maintain Boreal Owl habitat in forest systems where aspen provides a majority of the nesting habitat. Through coordinated timber harvest, large aspen which provide cavities for nesting may be maintained over the long-term, at the landscape scale, despite loss from individual stands. Focus on aspen management may even be more important in systems where aspen occupies a small proportion (<1%) of the landscape and occurs in small patches associated with particular microsites.

The shape of clear-cuts will likely influence both the short- and long-term impact on Boreal Owls. Although no direct evidence is available, I hypothesize that more complex shaped cutting units, especially those with stringers of forest extending into cutting units in upland areas, riparian buffers and patches of forest remaining within the cu[t] unit, will have fewer negative impacts than larg[e] rectangular or circular cuts. This hypothesis stem[s] from the pattern of habitat use by Boreal Owl pre[y] species (Williams 1955, Merritt 1981, Wells-Goslin[g] and Heaney 1984) and observations that Borea[l] Owls will nest in small patches of forest (G. Hay[ward] unpubl. data).

Based on the same arguments, sloppy clear-cut[s] (clear-cuts with residual standing dead and live trees, especially aspen and patchy slash), and cuts that retain standing and downed wood on the site will have fewer negative impacts, especially over the long-term. The mitigating qualities of retaining patches of live trees and shrubs, snags and woody debris arise from several factors. These elements will accelerate the rate at which the future stand attains mature and older forest characteristics (Knight 1994). In particular, recovery of fungi and lichen populations may be accelerated by maintenance of residuals (Ure and Maser 1982, Hansen et al. 1991).

Partial Cutting and Uneven-Age Management. Discussion of sloppy clear-cuts or irregular shelterwood prescriptions leads logically to discussion of partial cutting and uneven-age regeneration prescriptions. I hypothesize that group selection (harvest of small groups of trees in an uneven-age stand, maintaining the uneven-age properties) may not significantly reduce Boreal Owl habitat quality in many situations if, over the long-term, mature and old forest qualities are maintained and tree species composition does not exclude important cavity trees. Timber harvest prescriptions such as group selection and single tree selection (harvest of individual trees from an uneven-age stand in a pattern that maintains the size structure of the original stand) that retain forest structure, are compatible with developing owl nesting habitat. Thinning from below (harvest which removes individuals smaller than the dominant size class) and single tree selection that reduces competition among dominant trees and increases tree growth, could accelerate the process of developing suitable nest structures. While clear-cutting eliminates red-backed voles in Rocky Mountain forests (Campbell and Clark 1980, Scrivner and Smith 1984, Ramirez and Hornocker 1981), preliminary results of an experiment examining clear-cuts and group selection harvests indicate that red-backed voles remain abundant in partial cut stands when many large

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linked to forest conditions (Hayward and Hayward 1995) and forest management will influence the abundance of potential prey, and in turn, affect owl population persistence. Forest structure will also influence the availability of prey by changing owl access to prey. For instance, forests with dense ground cover or a high density of small trees will reduce the efficiency of foraging Boreal Owls. Furthermore, forest structure affects snow conditions which influence prey availability (Sonerud 1986).

Cavity availability and prey availability likely interact to influence Boreal Owl population growth. Tree cavities occur nonrandomly across the landscape as do small mammal populations. The spatial arrangement of cavities and prey (relative to one another) are important in determining Boreal Owl abundance. The conservation status of Boreal Owls will be intimately tied to the interaction of these resources.

While cavities and prey likely limit Boreal Owl populations in most landscapes, predation and competition may influence populations in certain circumstances. In local situations, mustelids destroy a high proportion of owl nests in some years (Sonerud 1985). The influence of these losses on population abundance is unknown. Evidence also indicates that interactions with other owls may influence the distribution of Boreal Owls suggesting that competition may be an important limiting factor in some situations (Hakkarainen and Korpimäki 1996).

Boreal Owl Management Within Ecosystem Management. In western North America the ecology of Boreal Owls is linked with many characteristics of mature and older spruce-fir forests (Hayward 1994b). Management which facilitates the long-term maintenance of a landscape with significant representation of mature and older forest habitat will provide quality Boreal Owl habitat. Therefore, management schemes which promote the processes that maintain productive spruce-fir forests, and management which facilitates the stand dynamics necessary to produce old spruce-fir forest, will provide the habitat characteristics necessary for Boreal Owls. As indicated earlier, this is not incompatible with timber harvest.

Most applications of ecosystem management strive to manage systems to emulate natural disturbance patterns and processes. As reviewed by Knight (1994), spruce-fir forests experience a variety of disturbance agents that act at scales ranging from single trees to hundreds of hectares. Development of old forest conditions following stand-replacement disturbance proceeds slowly; succession to mature forest conditions takes >150 yr. However, old forest stands represent a mosaic resulting from the frequent action of small scale disturbance. Partial cutting emulates (to some extent) insect mortality and windthrow, two common disturbances integral to the formation of old spruce-fir forest structure. Alexander (1987:59) indicated that "uneven-aged cutting methods—individual tree and group selection—have seldom been used in spruce-fir forests, they appear to simulate the natural dynamics of these forests." Therefore, careful harvest of trees from spruce-fir forest may not be incompatible with maintaining important elements of old forest and habitat characteristics linked with Boreal Owls.

The paucity of information available on the response of Boreal Owls to specific forest management actions presents an obstacle to the formulation of management within an ecosystem framework. A strong conservation strategy for Boreal Owls cannot be produced without new knowledge on Boreal Owl ecology. Management based on current knowledge must contend with uncertainty and be devised specifically to deal with this uncertainty. Adaptive management (Walters 1986), then, must be built into any approach to manage the species, particularly an ecosystem management strategy.

CONCLUSIONS

Based on my review of the habitat relationships of Boreal Owls and management considerations, I offer the following conclusions: (1) Maintaining Boreal Owls on a local scale is not incompatible with timber harvest but is incompatible with extensive, stand replacement silviculture implemented over entire watersheds, employing large cutting units; (2) Forests with high habitat value for Boreal Owls develop through long successional trajectories. Therefore forest management must consider long-term forest patterns on broad spatial scales; (3) The hypothesized metapopulation structure of Boreal Owls in North America suggests that forest management must be coordinated at a regional scale; (4) Adaptive management which links managers and research ecologists is necessary to produce the knowledge needed to understand the response of Boreal Owls to alternative management approaches at a variety of spatial scales; (5) As a top carnivore that preys upon the dominant small mammal species in subalpine forests and nests in

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large tree cavities, the Boreal Owl integrates into its ecology many aspects of forest dynamics. As such, the owl may represent a good model to aid in developing ecosystem management; (6) At all spatial scales, an eye to restoration management must be taken in landscapes that have experienced intensive harvest in the past. Restoration may be particularly appropriate in aspen forests of the Rocky Mountains.

Forest management which sustains mature subalpine and boreal forests likely will conserve Boreal Owls. Such management, however, must consider (among other things) the successional dynamics of spruce-fir forests including the detritus food chain, the consequences of various disturbances and the long-term (post-glacial) trends in these forests. Management must focus as much on the long-term condition of the plant communities used by Boreal Owls as on the population dynamics of the owl.

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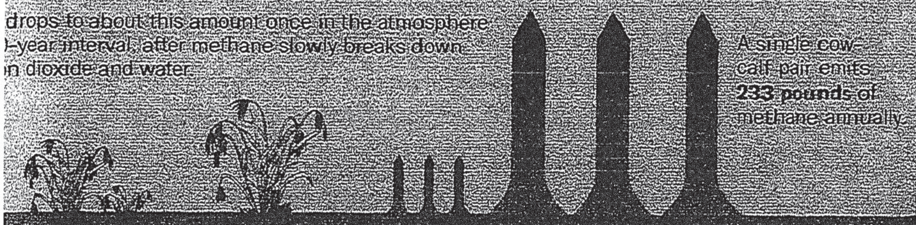
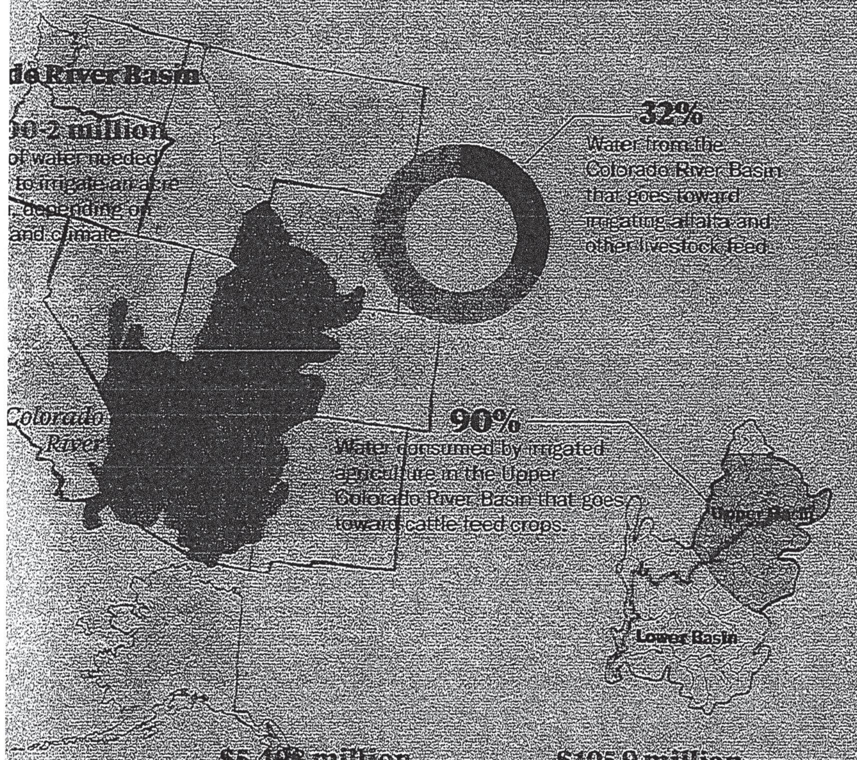


31.3 million acres
Minimum amount of land in the Western U.S. dominated by cheatgrass, a noxious, **fire-prone** weed spread by grazing, as of 2000.

123.5 million
Tons of carbon lost to the atmosphere as of 2000 due to the conversion of native rangelands to cheatgrass in the Wyoming big sagebrush biome.



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Administrative
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n public lands.

$5.498 million
Amount that industry, including livestock lobbying groups, donated to Frank Mitloehner, a UC Davis animal science professor who downplays cattle's contribution to climate change.

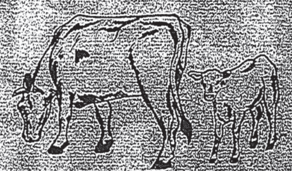
$36
Social cost of greenhouse gas — the estimated cost of damage done to the climate — for one AUM on Western public lands.

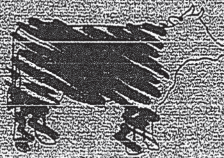
$105.9 million
Amount budgeted to the Interior Department for rangeland management in 2020, meaning taxpayers are subsidizing grazing operations to the tune of $90 million per year.

$2.5 billion
Total amount of federal conservation, disaster, commodity and crop insurance subsidies paid to ranchers and farmers in the 11 Western states between 1995 and 2020.

Animal Unit Month (AUM)

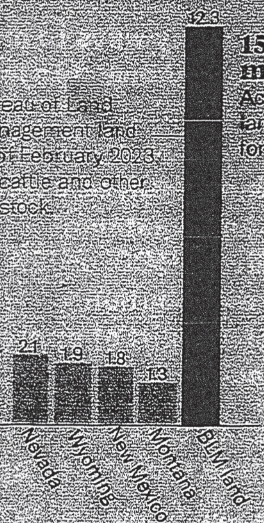
Forage required to feed one 1,000-pound cow-calf pair grazing on public land for one month.





Percentage of a cow's body weight the animal will eat in one month. This can increase in desert environments where cows must walk farther to reach forage.

Number of active AUMs (in millions)



155 million
Acres of BLM land allotted for grazing.

Bureau of Land Management land as of February 2023 for cattle and other livestock.

$15.9 million
Revenue the BLM received from grazing fees for all livestock categories in 2020.

SOURCES: U.S. Department of Agriculture, Bureau of Land Management, Bureau of Reclamation, Environmental Working Group, Environmental Protection Agency, "Water Scarcity and Fish Imperilment Driven by Beef Production," by Brian Richter, et al., "The animal agriculture industry, US universities, and the obstruction of climate understanding and policy," by Viveca Morris and Jennifer Jacquet, "Livestock Use on Public Lands in the Western USA Exacerbates Climate Change: Implications for Climate Change Mitigation and Adaptation," by J. Boone Kauffman, et al.

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Red Squirrels Snags 2006

Research Article

Sciurid Habitat Relationships in Forests Managed Under Selection and Shelterwood Silviculture in Ontario

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Abstract

*Although partial forest harvesting is practiced over large areas, managers know little about its impacts on sciurid rodents, particularly on northern (*Glaucomys sabrinus*) and southern flying squirrels (*G. volans*) in the northeastern United States and Canada. We examined habitat relationships of sciurid rodents (northern flying squirrels, southern flying squirrels, red squirrels [*Tamiasciurus hudsonicus*], and eastern chipmunks [*Tamias striatus*]) at 2 spatial scales in managed and unmanaged coniferous and hardwood forests of Algonquin Provincial Park, Ontario, Canada. We live-trapped rodents in 26 northern hardwood stands and in 16 white pine (*Pinus strobus*) stands from 2002 to 2004. Northern flying squirrel and red squirrel densities were significantly lower in recently harvested (3–10 yr since harvest) shelterwood stands than in unmanaged stands. In contrast, southern flying squirrel densities were higher in selection-harvested stands than in old-forest areas. The densities of northern flying squirrels and red squirrels had a strong relationship with the density of large spruce (*Picea *sp.) and hardwood trees and snags in conifer sites. Southern flying squirrel numbers had a positive association with the density of mast trees at the landscape level but not at the stand level in hardwood forests. Eastern chipmunk density had a positive correlation with the volume of old downed woody debris and the stems per hectare of declining trees. We recommend forest managers retain more large spruce and hardwood trees to mitigate the impacts of shelterwood harvesting on northern flying squirrels and red squirrels, and that they maintain high mast availability at the landscape level to ensure the persistence of southern flying squirrels.* (JOURNAL OF WILDLIFE MANAGEMENT 70(6):1735–1745; 2006)

Key words

eastern chipmunk, Glaucomys sabrinus, Glaucomys volans, *habitat use, northern flying squirrel, Ontario, partial harvesting, red squirrel, southern flying squirrel, stepwise regression,* Tamias striatus, Tamiasciurus hudsonicus.

Many jurisdictions in North America, including Ontario, Canada, have selected flying squirrels (*Glaucomys* spp.) as indicators of sustainable forest management practices. This designation has resulted in a relatively large body of research on these and other tree squirrels in landscapes managed under clearcut logging (Rosenberg and Anthony 1992, Witt 1992, Carey 1995, 2000, Martin and Anthony 1999, Cote and Ferron 2001). However, partial harvesting techniques, such as selection and shelterwood logging, have received less attention. These are common silvicultural techniques employed in temperate mixedwood forests in northeastern Northern America. In these systems, forest operators remove a portion of the overstory at relatively shorter intervals (approx. 20 yr), creating a more frequent, but less intensive disturbance regime, than under clearcut logging. The effects of partial harvesting on canopy-dwelling organisms are likely to differ from those resulting from clearcutting because partial harvesting maintains a relatively closed-canopy mature forest throughout the harvest cycle. Unfortunately, only 2 studies have examined the effects of partial harvesting (shelterwood harvesting) on flying squirrels (Waters and Zabel 1995, Taulman et al. 1998). These studies found that relatively high harvest intensities (<10 m^2/ha residual basal area) negatively affected flying squirrel populations. Researchers have not examined the effects of selection harvesting systems in hardwood forests, which typically leave greater residual basal areas than shelterwood logging.

Although partial harvesting systems retain canopy cover on sites, impacts on tree squirrel populations may manifest through other logging-induced changes in forest structure. Partial harvesting typically involves a reduction in the abundance of diseased and dead trees (McComb and Lindenmayer 1999, McGee et al. 1999, Costello et al. 2000) and often results in more homogenous forest structure, with reduced tree density and size (Costello et al. 2000). These changes may be important for arboreal mammals (Gerrow 1996, Carey 2000) and could result in negative effects for cavity nesters (Imbeau et al. 2001).

Most past studies on sciurids have only considered local (site-level) effects; however, organisms may demonstrate different responses to the same factor at different scales (Wiens 1989). Studies in fragmented landscapes suggest that flying squirrels may be sensitive to area effects (Nupp and Swihart 2000) and indicate that large clearcuts may act as barriers to dispersal and movement (Bendel and Gates 1987). Taulman (1999) found that flying squirrels nested in adjacent unharvested forest following partial harvesting, suggesting that the amount and configuration of unharvested stands might modulate flying squirrel responses to forest harvesting. In concert, these studies raise the possibility that local responses to forest harvesting might,

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Table 3. Mean (± SE) values for site-level habitat variables that differed among logged and old hardwood and pine forest sites in Algonquin Provincial Park, Ontario, Canada, 2002–2004. Variable units are the number of stems per hectare unless otherwise stated.

Habitat variable[c]	Recent cut n = 8, 7[a] Mean	Recent cut SE	Old cut n = 8[b] Mean	Old cut SE	Old forest n = 10, 9[a] Mean	Old forest SE	P
Hardwood							
BA (m^2/ha)	19.0	1.2	20.7	0.8	31.0	1.4	<0.001
Conifer	13.1	7.7	11.5	5.9	39.9	8.4	0.020
Con ≥ 25	8.1	5.4	5.0	2.5	23.6	5.8	0.028
Hwd ≥ 25	54.5	3.8	62.4	5.0	70.4	4.0	0.044
Spruce ≥ 25	0.5	0.4	0.5	0.3	3.5	1.1	0.014
Snags	10.8	3.7	13.3	3.5	28.8	4.6	0.008
Snags ≥ 25	3.0	0.7	6.0	1.4	10.1	1.1	<0.001
Pine							
BA (m^2/ha)	20.0	1.8	NA	NA	30.7	1.3	<0.001
CanClos (%)	76.2	3.2	NA	NA	89.4	1.6	0.001
Conifer	153.8	21.5	NA	NA	229.3	25.4	0.047
Hwd ≥ 25	12.4	3.9	NA	NA	27.4	3.9	0.018
Spruce ≥ 25	3.2	1.0	NA	NA	8.1	2.1	0.080
Snags ≥ 25	4.7	1.1	NA	NA	11.1	2.0	0.022

[a] Sample sizes for hardwood and pine forests, respectively.
[b] We sampled old cut stands only in hardwood forests.
[c] See text for definition of habitat variables.

11.1 → 4.7 = -58%

logging histories evenly between years, and hence allow equal application of any biases among comparisons.

Habitat variable reduction and variance partitioning.—The various sciurid species tended to be more or less abundant in one forest type (hardwood or conifer), but uncommon in the other; hence, we undertook community analyses separately for hardwood and white pine sites. Because squirrel densities differed between trap years (see Table 1), we first partialled it out of the models. To examine species–habitat relationships, we performed partial redundancy analysis (RDA), which served to reduce the number of site-level habitat variables (some of which were highly correlated with each other). Initially, we entered the 17 site habitat variables into the RDA (see Habitat Measurements, above). We computed the statistical significance of each variable using Monte Carlo simulations and a forward selection routine (9,999 permutations). We sequentially removed variables that contributed little (highest nonsignificant P-value) until the inflation factor was <10 (ter Braak and Smilauer 1998). We retained 11 variables in the hardwood model and 9 in the white pine model.

We investigated the relative importance of site and landscape variables using variance partitioning (Borcard et al. 1992, Cushman and McGarigal 2004). This analysis measures the variation in a community matrix, explained independently and jointly by different sets of explanatory variables. We calculated the unique variation at each level (site and landscape) by partialling out (as a covariate) variation due to the other level. We performed all multivariate analyses with CANOCO 4.5 for Windows (ter Braak and Smilauer 1998).

Individual species responses.—We plotted the density of each squirrel species against all habitat variables to ensure that assumptions of normality and homogeneity were satisfied and to check for the possibility of curvilinear relationships. We ln-transformed southern flying squirrel densities in hardwood sites and red squirrel densities in pine sites to normalize the variance. We performed analysis of variance (ANOVA) with Bonferroni-corrected post hoc tests for each species to compare densities among years. Where a significant year effect existed ($P \leq 0.050$), we included it in all the following regression and ANOVA analyses. We analyzed the influence of logging history on squirrel density and habitat variables with ANOVA.

We used linear regression to develop sciurid habitat relationship, except in 2 instances (northern flying squirrels and red squirrels in hardwood sites) where densities were low and we therefore used logistic regression (on presence/absence). For all species, if a year effect was significant, we included it in all models. To develop site-level habitat models, we used 2 methods, stepwise regression and a priori selection of candidate variables followed by use of Akaike's Information Criterion (AIC; Burnham and Anderson 1998) to compare among models created from these candidate variables. In the former method, we used the site-level habitat variables remaining after the RDA forward selection routine. In the latter method, we picked 5–6 variables for each species that we reasoned to be most important based on relationships observed or hypothesized in the literature (listed in Table 1). From these candidate variables, we constructed all possible models of up to 3 terms and used AIC_c to compare among them. Specifically, we calculated the difference between the ith model and the minimum AIC_c (i.e., ΔAIC_c) and the Akaike weight (w_i) for each model.

To incorporate landscape-level variables into these analyses, we first forced the best site-level model into the analysis and then evaluated the value of the landscape variables in explaining additional variation in squirrel densities. We followed this hierarchical procedure for 2 reasons: 1) site-level variables proved to be more important than landscape-level variables in explaining squirrel captures abundance of flying squirrels and hypogeous fungi was significantly lower in shelterwood stands than in uncut old-growth stands in California (Waters and Zabel 1995). Beyond the local site-level effects of harvesting, we observed additive effects of logging on the surrounding landscape. This suggests that the interspersion of large harvested and unharvested blocks on the landscape may be important to ensure the persistence of red squirrels and northern flying squirrels.

We developed habitat models using both stepwise regression and models developed a priori. Stepwise regression has fallen into disfavor recently because of misuse of the technique (excessive data dredging), its reliance on arbitrary cutoffs (e.g., $\alpha = 0.05$), and model over-fitting (Burnham and Anderson 1998, Stephens et al. 2005). However, models developed a priori may be equally prone to problems when considering a large number of potential variables (i.e., model dredging; Stephens et al. 2005). Thoughtful model development is critical in both stepwise techniques and a priori model building. If researchers take care in developing models, we believe stepwise regression is a powerful tool and should continue to be used in habitat studies. Habitat studies often lead to the development of large sets of potential variables, and the inclusion of novel variables can lead to new insights.

Management Implications

Shelterwood harvesting decreased the density of large spruces and hardwoods below critical thresholds for northern flying squirrels and red squirrels. In order to mitigate the structural changes resulting from logging, we recommend managers retain at least 4.5 spruce trees/ha $\geq$25 cm dbh and 15 hardwood trees/ha $\geq$25 cm dbh in white pine stands (levels predicted to maintain northern flying squirrel densities in shelterwood stands at 50% of their mean level in old-forest areas). The abundance of mast trees on the landscape appears to limit southern flying squirrel density. In hardwood stands, we recommend foresters maintain a minimum of 17% of the landscape area (to maintain 80% of their mean level in hardwood forests) with high mast tree abundance (i.e., where $\geq$20% of the stand composition is a mast tree).

Acknowledgments

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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-*

Paper submitted October 11, 1994; revised manuscript accepted April 19, 1995.

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 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.

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.*

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 Diameter at Breast Height (cm)					
Tree Status	*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.*

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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.

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 bird species. In addition to fire severity, the number of years following a fire event (time-since-fire) can have significant influence on bird response. Again, there are some important studies that have included early vs later stages of succession after fire in their analyses of bird occurrence patterns (Taylor 1973, 1979; Taylor and Barmore 1980; Raphael *et al.* 1987; Breininger and Smith 1992; Hutto *et al.* 1995; Pyke *et al.* 1995; Ganey *et al.* 1996; Woinarski *et al.* 1999; Bechtoldt and Stouffer 2005; Covert-Bratland *et al.* 2006; Saab *et al.* 2007; Vierling and Gentry 2008; Brown *et al.* 2009; Chalmandrier *et al.* 2013; Holmes *et al.* 2013; Hutto *et al.* 2015). Taken together, results from these two kinds of studies suggest that the simultaneous consideration of severity and time-since-fire might allow us to detect fire effects that are even more nuanced.

The purpose of this study was to document the response of native bird species to the Black Mountain fire, a 3500-ha, lightning-caused, mixed-severity fire that burned through a lower-elevation dry, mixed-conifer forest immediately west of Missoula, Montana, in 2003. Using point-count data from each of 10 years following the fire, we compared the occurrence rates of bird species in a variety of burned forest conditions across a space-time continuum with their occurrence rates in surrounding unburned forest of the same vegetation type. This study was designed to test whether a combination of fire severity and time-since-fire is necessary to expose some positive effects of fire on birds in dry, mixed-conifer forest, which is common across the mountainous West.

Methods

Study area

Situated at the south-west edge of the city of Missoula, Montana, United States, the 2003 Black Mountain fire started in mature, low-elevation, mixed-conifer forest dominated by Ponderosa Pine (*Pinus ponderosa*), Western Larch (*Larix occidentalis*), Lodgepole Pine (*Pinus contorta*), and Douglas-fir (*Pseudotsuga menziesii*) on 8 August as the result of a lightning strike. The fire burned slowly until it grew rapidly on 16 August when it spread across 1500 ha in 2 h and prompted the evacuation of 130 homes; it eventually burned ~3500 ha. The spatio-temporal effects of the Black Mountain fire and their relationship with wildland fire hazard and risk were discussed by Hardy (2005), the fire was also the subject of several studies that were designed to measure the effectiveness of the severity classifications associated with the Burned Area Reflectance Classification (BARC) map that was generated following the fire (Hudak *et al.* 2004; Hudak *et al.* 2007; Lentile *et al.* 2007).

Overall design

There are limits to case history studies of single fire events due to the lack of treatment replication, but it is nearly impossible to attain true treatment-level replication of severe-fire events in either an experimental or natural fashion; case studies are *sine qua non* in fire ecology (Hargrove and Pickering 1992). Therefore, fire ecologists must take advantage of individual opportunities that arise, and then rely on meta-analyses at some point in the future to understand the extent to which results can be safely generalised more broadly beyond any single event.

To assess fire effects, we sought to compare bird occurrence rates in the burned forest to those we would have expected to observe in the absence of fire. As is true with most unplanned natural disturbance events, no bird surveys had been conducted within the burned area before the fire, so we estimated what the occurrence rates of bird species in the study area were likely to have been before the fire by drawing samples from a subset of the Northern Region Landbird Monitoring Program database (Hutto and Young 2002). Point-count locations were positioned in unburned but otherwise similar (dry, low-elevation, mixed-conifer) forest within a 100-km^2 block centred on the fire using methods identical to those used in this study (Fig. 1). We used data from the most recent year sampled at each point, which yielded 717 points that were surveyed sometime between 1992 and 2009. Although bird occurrence rates certainly vary across space and time, that variation is very small relative to variation among vegetation types. Therefore, the large number of points drawn from a variety of locations and years should serve to swamp outlier places or years that might otherwise bias an estimate of the 'average' occurrence rate for each species in a typical unburned mixed-conifer forest.

We used a digital orthophoto of the fire perimeter to initially position 279 bird survey points throughout the burned area, spacing points no closer than ~200 m from any other point. Beginning 9 months after the fire and for 10 of the first 11 years following the fire, one of the authors (RLH) visited an average of 100 (range = 77–127) points every post-fire year except 2008 within a 6 × 2 km rectangle that covered the south-east portion of the fire (Fig. 1). The survey locations were well distributed across the study area in each year, although the precise locations varied somewhat from year to year because of variation in survey routes taken by the observer. A given point may or may not have been visited in more than one year; specifically, the numbers of points visited from 1 to 10 times across years were 80, 37, 29, 14, 17, 41, 39, 12, 9 and 1, respectively. In most instances, points were visited on a single occasion in any given year (of 1087 point visits across all years, only 29 were visited multiple times in a given year). In instances where points were visited more than once in a given year, we randomly selected one of the visits for analysis. Thus, data associated with each point are represented by a single visit in each year. A summary of survey effort (numbers of independent survey points) across the combinations of fire severity and time-since-fire are presented in Table 1.

Bird surveys

Point counts were conducted during the height of the breeding season every year (last week of May to the first week of July) and lower elevations were visited earlier in the season than higher elevations. On a given visit to each point, we used standard 10-min point-count methodology (Hutto *et al.* 1986; Ralph *et al.* 1995) to record the distance to and identity of each bird detected by either sight or sound between 0630 and 1130 hours. We also recorded an on-the-ground visual estimate of tree mortality percentage (1–20%, 21–40%, 41–60%, 61–80% and >80%) within 100 m of each survey point during each of the first 2 years after the 2003 fire and used the mean value as an index of fire severity surrounding the point.

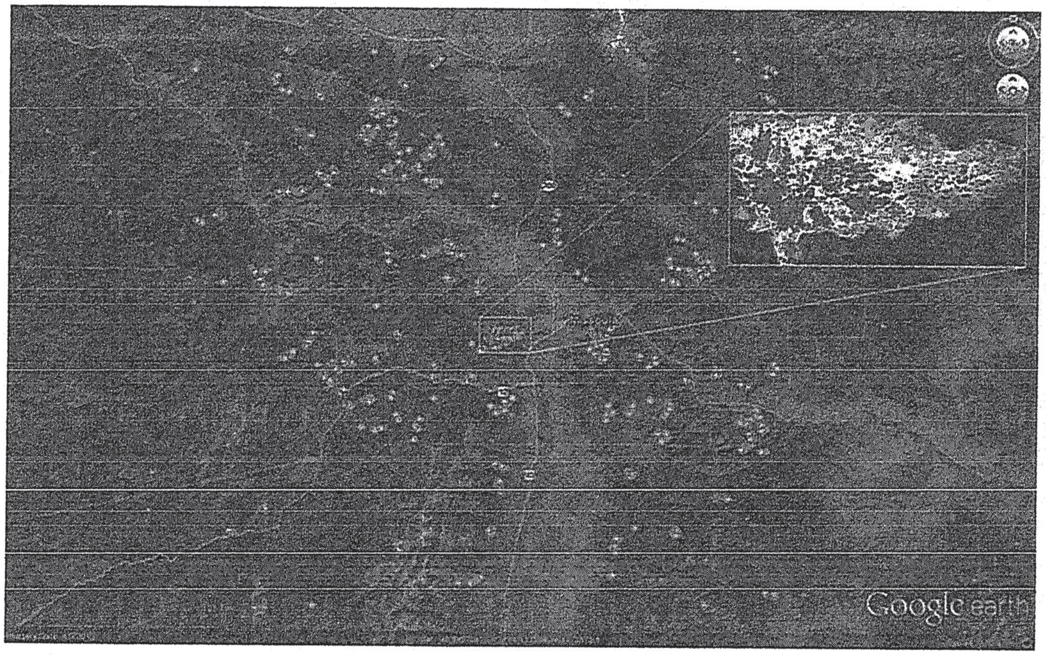

Fig. 1. The small rectangle encloses the bird survey points that were positioned within the 2003 Black Mountain fire perimeter, 5 km west of Missoula, Montana. The larger 100-km^2 area surrounding the rectangle shows the locations of the 717 unburned-forest bird survey points outside the fire perimeter. Source: Google Earth.

Table 1. The number of independent surveys (point-counts) that were conducted in each combination of time-since-fire and fire severity within the Black Mountain fire near Missoula, Montana, USA

Fire severity (tree mortality)	Two-year interval				
	2004–2005	2006–2007	2009–2010	2011–2012	2013–2014
1–40%	35	36	40	59	57
41–80%	47	51	65	85	97
>80%	100	97	84	108	126

Statistical analyses

Because the numbers of individuals of any one species are notoriously difficult to estimate during a 10-min count, the most reliable information that can be obtained from a point count is the presence or absence of a species (Hutto 2016). Therefore, we recorded the species detected within a fixed, 100-m radius surrounding each survey point and used the proportion of points at which a species was detected (naïve occupancy) as a response variable to reflect bird abundance. We did not employ occupancy modelling (MacKenzie *et al.* 2006) or distance sampling to estimate bird density (Buckland *et al.* 2001) because points were visited only once in a given year, sample sizes for most species were too small to estimate reliable detection functions and mean detection distances to all bird species in the unburned and burned forest types (the most important potential source of detectability bias) were not significantly different (46.6 m vs 47.1 m; $t = -1.14$, d.f. = 15 810, $P = 0.26$). In addition, several fundamental assumptions associated with distance sampling could not be met; these include the assumptions that (1) there is no bird movement in response to the observer, (2) changes in vegetation with distance from the observer do not confound the effect of distance alone, (3) observers can obtain accurate estimates of the number of individuals of each species surrounding a point and (4) observers can obtain accurate estimates of distances to birds that were heard but not seen (90% of all detections). Welsh *et al.* (2013) discussed why these and other problems can lead to biases that are as bad as or worse than those that might be present in unadjusted data. Johnson (2008) also discussed instances where simple indices, such as percentage occurrence, are likely to be more reliable than distance-adjusted density estimates and a multi-species survey is one of those instances. To confirm that the patterns we describe are not artefacts of detection bias, we conducted an additional analysis based on data drawn from a very limited 50-m radius, where detections can be assumed to be very near 100% for most species, none of the bird distribution patterns was affected by the use of this more restricted dataset. Analyses were conducted on 50 species that were detected on at least 10 point-counts (either inside or outside the fire perimeter), were not wide-ranging large raptors and were not restricted to riparian or wetland habitats embedded within the mixed-conifer forest.

We started with a traditional analytical approach, where we used Chi-square test analyses to investigate whether the probability of occurrence of any given bird species differed significantly between burned and unburned points. We then conducted a more refined analysis to assess whether there were differences in percentage occurrence between the surrounding unburned mature forest and any combination of fire severity and time-since-fire. To smooth out smaller-scale variability and to achieve adequate sample sizes in each combination of fire severity and time-since-fire, we first aggregated the counts for each species into 3 fire-severity levels (1–40%, 41–80% and 81–100% tree mortality) and 5 2-year periods (2004–2005, 2006–2007, 2009–2010, 2011–2012 and 2013–2014), which produced counts for 15 combinations of fire severity and time-since-fire. We then computed the odds ratio of seeing a species in each of the 15 combinations relative to the surrounding unburned (equivalent to pre-fire) forest. Since there were some 0 counts, we added 0.5 to all counts for all species (Agresti 2002). The odds ratio for fire severity level i and time period j for a given species is:

$$\hat{\theta}_{ij} = \frac{(x_{ij1} + 0.5)/(x_{ij0} + 0.5)}{(x_{b1} + 0.5)/(x_{b0} + 0.5)}$$

where x_{ij1} and x_{ij0} are the numbers of points where the species was and was not detected, in fire severity level i during time period j, and x_{b1} and x_{b0} are the analogous counts for the outside-fire base (labelled 2003 along the year axis, but representing data from outside the burn, regardless of year that the data were collected). An approximate standard error for $\log \hat{\theta}_{ij}$ is (Agresti 2002):

$$\mathrm{SE}\left(\log \hat{\theta}_{ij}\right) = \left(\frac{1}{x_{ij1} + 0.5} + \frac{1}{x_{ij0} + 0.5} + \frac{1}{x_{b1} + 0.5} + \frac{1}{x_{b0} + 0.5}\right)^{1/2}.$$

To assess the statistical significance of the odds ratio we computed $z_{ij} = \log \hat{\theta}_{ij}/\mathrm{SE}(\log \hat{\theta}_{ij})$ (the expected value of the log-odds ratio is 0 when the true-odds ratio is 1). Because we ran over 700 comparisons in total, we calculated Bonferroni-adjusted P-values to provide an estimate of the statistical significance associated with each odds ratio. All data manipulation and plots were conducted in R (R Core Team 2014) using packages dplyr (Wickham and Francois 2015), tidyr (Wickham 2014) and ggplot2 (Wickham 2009).

Results

We detected a total of 107 bird species in the combined dataset drawn from burned and unburned forests, and 50 of those species met target requirements for inclusion in analyses as described in the methods (Table 2). By grouping points into two categories (burned-forest and unburned-forest points) and then calculating the percentage occurrence rates of each species inside and outside the burned forest (Table 2), we found 25 species to be more abundant in the burned forest (23 significantly so; $P < 0.05$), and 25 to be more abundant outside the burned forest perimeter (21 significantly so; $P < 0.05$). In contrast, a majority of species (60%) was significantly more likely to be detected in at least one category representing a particular combination of fire severity and time-since-fire within the burned forest than within mature, unburned, green-tree forests of the same type (Fig. 2). Although most species responded positively at 1 or more combinations of fire severity and time-since-fire, not all species responded positively in the same way. To help visualise the positive and negative responses to fire and the differences in patterns of response among species, we colour-coded the odds ratios associated with a comparison of the occurrence rates for a species in each combination of fire severity and time-since-fire and its occurrence rate in unburned forest outside the fire perimeter (Fig. 2).

Fourteen (28%) of the 50 species revealed significantly greater abundances within than outside the burned forest within 2 years following fire, most commonly in the moderate or severely burned forest patches (Fig. 2). These included four woodpecker species (Black-backed Woodpecker, Hairy Woodpecker (*Picoides villosus*), American Three-toed Woodpecker and Northern Flicker (*Colaptes auratus*)) several thrush species (Western Bluebird (*Sialia Mexicana*), Mountain Bluebird, Townsend's Solitaire (*Myadestes townsendi*)), two flycatcher species (Western Wood-Pewee (*Contopus sordidulus*) and Dusky Flycatcher (*Empidonax oberholseri*)), and two seed-eating specialists (Cassin's Finch (*Haemorhous cassinii*), Pine Siskin (*Spinus pinus*)), among others (e.g. Rock Wren (*Salpinctes obsoletus*), Lazuli Bunting (*Passerina amoena*), Chipping Sparrow and (*Spizella passerina*)).

Several additional species exhibited significant but delayed increases in abundance within the burned forest (e.g. Tree Swallow; Lewis's Woodpecker (*Melanerpes lewis*), Pygmy Nuthatch (*Sitta pygmaea*), Dusky Grouse (*Dendragapus obscurus*), Vesper Sparrow (*Pooecetes gramineus*), White-breasted Nuthatch (*Sitta carolinensis*), Calliope Hummingbird (*Selasphorus calliope*) and Williamson's Sapsucker (*Sphyrapicus thyroideus*); Fig. 2).

The significantly positive response to fire was, for several species (e.g. Pygmy Nuthatch, Calliope Hummingbird, Cassin's Finch, Brown-headed Cowbird (*Molothrus ater*), Red Crossbill (*Loxia curvirostra*), Pine Siskin, Red-breasted Nuthatch (*Sitta Canadensis*), Hammond's Flycatcher (*Empidonax hammondii*) and Hermit Thrush (*Catharus guttatus*)), relatively restricted to the lowest fire severity category (Fig. 2). A relatively large number of additional species (MacGillivray's Warbler (*Geothlypis tolmiei*), Common Raven (*Corvus corax*), Pileated Woodpecker (*Dryocopus pileatus*), Western Tanager (*Piranga ludoviciana*), Evening Grosbeak (*Coccothraustes vespertinus*), Yellow-rumped Warbler (*Setophaga coronata*), Mountain Chickadee (*Poecile gambeli*), Cassin's Vireo (*Vireo cassinii*) and Ruby-crowned Kinglet (*Regulus calendula*)) showed a similar but non-significant response to fire.

Only six (12%) of the 50 species (Steller's Jay (*Cyanocitta stelleri*), Black-capped Chickadee (*Poecile atricapillus*), Gray Jay (*Perisoreus canadensis*), Townsend's Warbler (*Setophaga townsendi*), Swainson's Thrush (*Catharus ustulatus*) and

Table 2. Bird species detected at survey points within burned or surrounding unburned mixed-conifer forest

The list excludes species detected on fewer than 11 points, riparian specialists, and raptors. The location where a species was more frequently detected is shown in bold. The last column shows the level of significance of any difference based on a Chi-square test. The four-letter mnemonic code for each species is provided in parentheses after the Latin binomial. Statistically significant differences are based on Chi-square likelihood ratio ($P < 0.05$*, $P < 0.01$**, $P < 0.001$***)

Species	Burned ($n = 1087$)		Unburned ($n = 717$)		P^A
	Hits	%	Hits	%	
Dusky Grouse, *Dendragapus obscurus* (DUGR)	46	**4.23**	2	0.28	***
Calliope Hummingbird, *Selasphorus calliope* (CAHU)	75	**6.90**	14	1.95	***
Lewis's Woodpecker, *Melanerpes lewis* (LEWO)	15	**1.38**	0	0	***
Williamson's Sapsucker, *Sphyrapicus thyroideus* (WISA)	58	**5.34**	14	1.95	***
Hairy Woodpecker, *Picoides villosus* (HAWO)	244	**22.45**	21	2.93	***
American Three-toed Woodpecker, *Picoides dorsalis* (ATTW)	38	**3.50**	4	0.56	***
Black-backed Woodpecker, *Picoides arcticus* (BBWO)	76	**6.99**	0	0	***
Northern Flicker, *Colaptes auratus* (NOFL)	275	**25.3**	35	4.88	***
Pileated Woodpecker, *Dryocopus pileatus* (PIWO)	10	0.92	20	**2.79**	**
Olive-sided Flycatcher, *Contopus cooperi* (OSFL)	33	3.04	30	**4.18**	ns
Western Wood-Pewee, *Contopus sordidulus* (WEWP)	209	**19.23**	2	0.28	***
Hammond's Flycatcher, *Empidonax hammondii* (HAFL)	100	9.20	131	**18.27**	***
Dusky Flycatcher, *Empidonax oberholseri* (DUFL)	285	**26.22**	102	14.23	***
Cassin's Vireo, *Vireo cassinii* (CAVI)	94	8.65	195	**27.20**	***
Warbling Vireo, *Vireo gilvus* (WAVI)	96	8.83	142	**19.8**	***
Gray Jay, *Perisoreus canadensis* (GRAJ)	8	0.74	64	**8.93**	***
Steller's Jay, *Cyanocitta stelleri* (STJA)	2	0.18	19	**2.65**	***
Clark's Nutcracker, *Nucifraga Columbiana* (CLNU)	43	**3.96**	15	2.09	*
Common Raven, *Corvus corax* (CORA)	16	1.47	22	**3.07**	ns
Tree Swallow, *Tachycineta bicolor* (TRES)	22	**2.02**	0	0	***
Black-capped Chickadee, *Poecile atricapillus* (BCCH)	10	0.92	57	**7.95**	***
Mountain Chickadee, *Poecile gambeli* (MOCH)	108	9.94	189	**26.36**	***
Red-breasted Nuthatch, *Sitta Canadensis* (RBNU)	341	31.37	386	**53.84**	***
White-breasted Nuthatch, *Sitta carolinensis* (WBNU)	79	**7.27**	10	1.39	***
Pygmy Nuthatch, *Sitta pygmaea* (PYNU)	22	**2.02**	1	0.14	***
Brown Creeper, *Certhia Americana* (BRCR)	12	1.10	17	**2.37**	*
Rock Wren, *Salpinctes obsoletus* (ROWR)	41	**3.77**	2	0.28	***
House Wren, *Troglodytes aedon* (HOWR)	384	**35.33**	0	0	***
Golden-crowned Kinglet, *Regulus satrapa* (GCKI)	2	0.18	120	**16.74**	***
Ruby-crowned Kinglet, *Regulus calendula* (RCKI)	152	13.98	321	**44.77**	***
Western Bluebird, *Sialia Mexicana* (WEBL)	123	**11.32**	0	0	***
Mountain Bluebird, *Sialia currucoides* (MOBL)	403	**37.07**	8	1.12	***
Townsend's Solitaire, *Myadestes townsendi* (TOSO)	194	**17.85**	116	16.18	ns
Swainson's Thrush, *Catharus ustulatus* (SWTH)	48	4.42	280	**39.05**	***
Hermit Thrush, *Catharus guttatus* (HETH)	38	3.50	58	**8.09**	***
American Robin, *Turdus migratorius* (AMRO)	241	22.17	164	**22.87**	ns
Orange-crowned Warbler, *Oreothlypis celata* (OCWA)	118	10.86	115	**16.04**	***
MacGillivray's Warbler, *Geothlypis tolmiei* (MGWA)	151	13.89	151	**21.06**	***
Yellow-rumped Warbler, *Setophaga coronate* (YRWA)	394	36.25	462	**64.44**	***
Townsend's Warbler, *Setophaga townsendi* (TOWA)	55	5.06	281	**39.19**	***
Chipping Sparrow, *Spizella passerine* (CHSP)	676	**62.19**	319	44.49	***
Vesper Sparrow, *Pooecetes gramineus* (VESP)	14	**1.29**	1	0.14	**
Dark-eyed Junco, *Junco hyemalis* (DEJU)	441	40.57	390	**54.39**	***
Western Tanager, *Piranga ludoviciana* (WETA)	295	27.14	372	**51.88**	***
Lazuli Bunting, *Passerina amoena* (LAZB)	163	**15.00**	11	1.53	***
Brown-headed Cowbird, *Molothrus ater* (BHCO)	164	**15.09**	74	10.32	**
Cassin's Finch, *Haemorhous cassinii* (CAFI)	93	**8.56**	20	2.79	***
Red Crossbill, *Loxia curvirostra* (RECR)	77	**7.08**	39	5.44	ns
Pine Siskin, *Spinus pinus* (PISI)	184	16.93	131	**18.27**	ns
Evening Grosbeak, *Coccothraustes vespertinus* (EVGR)	3	0.28	16	**2.23**	***

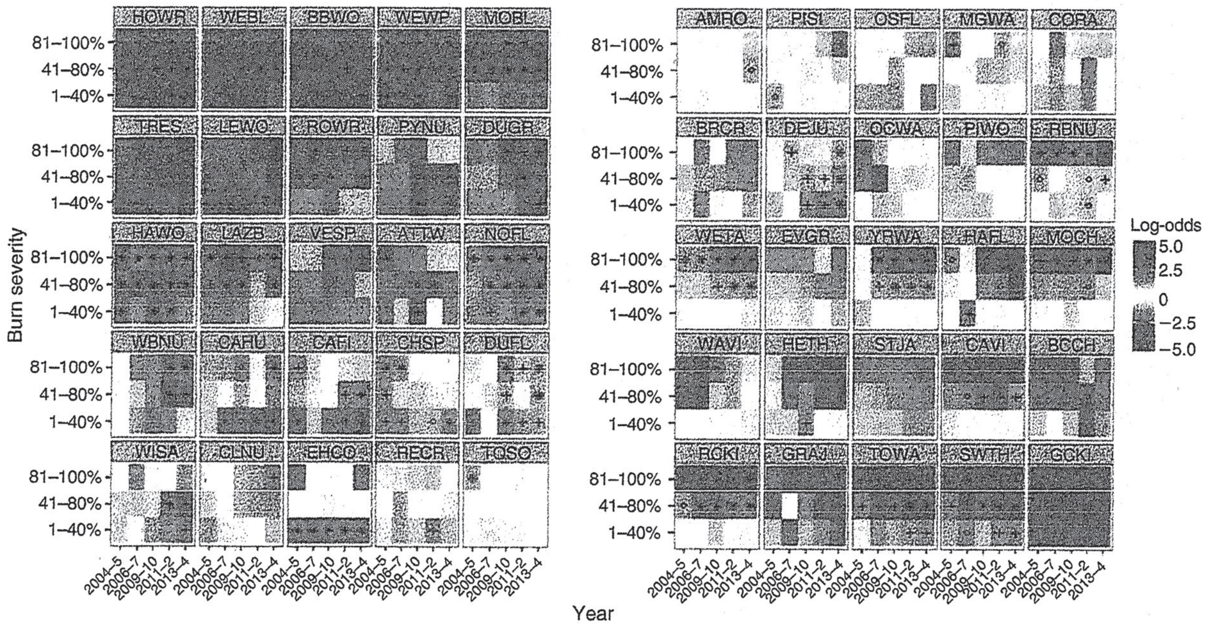

Fig. 2. Heat maps reflecting the log-odds ratio associated with the percentage occurrence in each combination fire severity and time-since-fire in comparison with the percentage occurrence in unburned forest outside the fire perimeter for each of 50 bird species (four-letter mnemonic codes provided in Table 2; species are organised by their average log-odds scores, from those that had a large average positive response to those that had a large average negative response to fire). Hotter (more red) blocks represent positive responses to fire and cooler (more blue) blocks represent negative responses to fire. The symbols correspond with Bonferroni adjusted P-values ($\diamond = 0.01 < P < 0.05$; $+ = 0.001 < < 0.01$; $* = P < 0.001$). Thirty of 50 species (60%) were significantly more abundant in burned forest at some combination of severity and time-since-fire than in unburned, mature green-tree forest.

Golden-crowned Kinglet (*Regulus satrapa*)) were detected less frequently after fire, regardless of fire severity, and their detection rates generally continued to decrease over time (Fig. 2).

Discussion

Following the most common approach to assessing fire effects, we first looked at whether there were significant differences in bird abundances between the burned and surrounding unburned forest. The results from this analysis were consistent with those reported in many other studies of fire effects on birds (see Kotliar *et al.* 2002) – roughly half the bird species appeared to benefit and half did not (Table 2). Unfortunately, this kind of analysis hid positive responses that became apparent only after accounting for fire severity and time-since-fire. By dividing the burned-forest data into 15 combinations of fire severity and time-since-fire, we found results that were more nuanced than those obtained from a simple 'burned vs unburned' analysis. Specifically, 30 of 50 (60%) of the bird species considered were significantly more likely to occur inside the burned forest (at 1 or more combinations of fire severity and time-since-fire) than outside the burned forest. The distinct location of the greatest probability of detection for any 1 species across the 2-dimensional fire-severity and time-since-fire gradient, combined with differences in those locations among species (Fig. 2), suggests that the bird occurrence patterns are accurate reflections of bird abundance and not artefacts of some kind of sampling bias that might affect all species similarly. Other recent work (Stephens *et al.* 2015) has also revealed that the locations of peak abundances across a fire-severity/time-since-fire gradient differ among species.

Many of these significantly positive responses would not have been evident without partitioning the data into multiple severity and time-since-fire categories. This kind of analysis is difficult to conduct with data from any one fire because sample sizes (the number of independent survey points in each severity-by-year category) are generally much smaller than what we were able to achieve here (Table 1). Even with the sample sizes we achieved, we were still forced to use fewer categories than the number used in the field to assess the statistical significance of fire effects. Although each bird species responded uniquely to the combination of fire severity and time-since-fire (Fig. 2), four general classes of response are worth noting, along with some of the most probable biological underpinnings behind each.

Response Pattern 1

This pattern is illustrated by species that showed an abrupt increase in abundance within the first few years following fire, and the elevated abundance persisted until the end of the 11-year study primarily (but not exclusively) in locations that burned at higher severities. Several woodpecker species (Black-backed Woodpecker, Hairy Woodpecker, American Three-toed Woodpecker and Northern Flicker) showed this response pattern (Fig. 2). The biological basis behind the abrupt increase in woodpecker populations is well established and unambiguous: bark and wood-boring beetle populations increase as individual beetles detect the newly created abundance of fire-killed trees.