Effects of Ecosystem Restoration Treatments on Cavity-nesting Birds, Their Habitat, and Their Insectivorous Prey in Fire-maintained Forests of Southeastern British Columbia[1]

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Abstract

Long-term fire suppression and cessation of aboriginal burning has resulted in forest ingrowth and associated structural and functional changes to dry, fire-maintained forests of southeastern British Columbia. In this experimental study, effects of ecosystem restoration treatments (prescribed fire, partial harvesting, prescribed fire and partial harvesting, and untreated controls) on cavity-nesting birds, their habitat, and their insectivorous prey were investigated. Pre- and post-treatment data on the nesting density of cavity-nesters, the availability of wildlife trees, the foraging intensity of woodpeckers, and the relative abundance and species diversity of insects each breeding season were gathered, beginning in 1996. Preliminary results (1996 to 1998) indicate that harvested treatments were associated with a decline in the nesting density and species richness of cavity nesters over the short term; snag densities decreased in harvested and burned treatments; the number of trees per hectare with fresh woodpecker foraging sign did not differ significantly comparing pre- and post-treatment; and large increases in the relative abundance of selected insect species in some families were apparent in the treated units. Insect response was most pronounced in the burn only treatment, followed by the harvest and burn, and the harvest only treatments. Management implications for future ecosystem restoration efforts are discussed.

Introduction

In the Rocky Mountain Trench and adjoining side valleys of southeastern British Columbia, an estimated 250,000 ha of land within the ponderosa pine (PP) and interior Douglas-fir (IDF) biogeoclimatic zones is classified as Natural Disturbance Type 4 (NDT4). Before 1900, these forests were characterized by "frequent, stand-maintaining fires" resulting in open park-like conditions with widely-spaced large old trees, sparse regeneration, and low incidence of insects and diseases (Arno and others 1995, Covington and Moore 1994, Daigle 1996, Gayton 1996). Systematic and long-term fire suppression and cessation of aboriginal burning has eliminated fire from these forests. This has stimulated forest ingrowth and encroachment, with associated reductions in ungulate forage quantity and quality, increased risk of stand

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replacement fires, and major forest insect and disease outbreaks. Recent operational trials have used mechanical treatments (e.g., partial cutting, slashing pruning) and prescribed fire in an attempt to restore the fire-maintained character of these forests (Gayton and others 1995, Hawe and DeLong 1998, White 1997). Once stands have been thinned and fuels have cured, prescribed burning can be carried without the risk of catastrophic fire, while providing some revenue from sawlogs and pulp. Effects of restoration treatments on wildlife communities have not been well documented in an experimental context (Finch and others 1997, Saab and Dudley 1997).

This study focuses on cavity-nesting birds—birds that are dependent on standing dead and defective live trees for nesting, roosting and feeding (Bull and others 1997, Raphael and White 1984, Steeger and others 1996). In dry, fire-maintained forests, the proportion of cavity-nesting species is comparatively high relative to other forest-dwelling vertebrates (Bunnell 1995). Burns create fire-scorched or killed trees, which are attractive to a host of wood, bark, and cambium-dwelling insects (Amman and Ryan 1991, Fellin 1980, Furniss and Carolin 1977, Samuelsson and others 1994). These insects form the prey base for many cavity-nesting species (Machmer and Steeger 1995). I hypothesized that the temporal and spatial occurrence of burns would have a strong influence on the population dynamics of this wildlife guild.

The objectives of this study are to investigate the individual and combined effects of ecosystem restoration treatments on the nesting abundance and species richness of cavity nesters, the availability of wildlife tree habitat, the foraging intensity of woodpeckers, and the relative abundance and species richness of wood-associated insects. In this paper, preliminary (1996 to 1998) results are reported, and some management implications for future restoration treatments are discussed.

Study Area and Design

The study is being conducted in the Ta Ta Creek range unit (49°46'N, 115°46'W) located 45 km north of Cranbrook in the Rocky Mountain Trench of southeastern British Columbia. The study area is in a flat valley bottom (850 to 900 meter elevation) and consists of moderately dense to semi-open multi-storied stands of 90- to 100-year-old Douglas-fir *(Pseudotsuga menziesii)* and ponderosa pine *(Pinus ponderosa)*. Small pockets of lodgepole pine *(P. contorta)* and trembling aspen *(Populus tremuloides)* are found in draws and wetter sites. Fire has been excluded from these for an estimated 60 years.

Three treatments (25 ha burn only; 40 ha harvest and burn; and 40 ha harvest only) and two control areas (one located adjacent to and a second located 3.5 km away from the treatment units) were tested without replication during operational trials. A uniform "shelterwood with reserves" silvicultural system was used for harvest treatments with the following objectives for post-harvest target densities: 26-36 stems ha^{-1} (sph) of layer 1a (20-65 cm dbh) healthy, well-formed veteran/dominant ponderosa pine and Douglas-fir; 5-20 sph of layer 1b (15-20 cm dbh) ponderosa pine and Douglas-fir; 10-20 sph of layer 2 (10-15 cm dbh) ponderosa pine and Douglas-fir; and 50-60 sph of layer 3 (3-5 cm dbh) ponderosa pine. Post-harvest slashing of small sub-merchantable stems was conducted in the harvest and burn treatment to ensure sufficient fuel to carry the fire. Burning treatments involved low-intensity prescribed fire with the following specific objectives: eliminate 75 and 25 percent of trees in the < 5 and < 15 cm diameter at breast height (dbh) classes, respectively; limit mortality of trees > 30 cm dbh to < 10 percent; prune all trees to a height of 2 m; and reduce ground fuels <10 cm and > 20 cm diameter by 50-75 and 5-10 percent, respectively. Harvesting was completed in February of 1997, and burning was conducted during April of 1998. Resulting stands differed visibly in terms of large stem densities and associated stand structure: post-treatment (1999) densities of trees > 10 cm dbh averaged 160, 48 and 161 sph in the burn only, harvest and burn and harvest only treatments, respectively, while densities of smaller stems were relatively comparable among treated areas.

Methods

We established 20 ha core sampling areas in each of the five treatment units during spring of 1996. These areas were monitored pre- and post-treatment during the 1996 to 1999 breeding seasons.

Cavity Nest Inventory

From early May to late June, we systematically searched the sampling areas for the presence of cavity-nesting species (both primary cavity excavators and secondary cavity users) and cavities in trees. We considered a nest active if parents were observed incubating, tending a brood, feeding young, or if young were heard vocalizing within the cavity. Two observers visited each of the five treatment units weekly on a regular rotation for 7 weeks (70 observer-days/season). We monitored nests weekly to determine their status and fate (i.e., success or failure), and calculated cavity nest densities for each treatment, excluding pairs that failed prior to confirming incubation.

Wildlife Tree Inventory

We assessed the number and characteristics of all standing live healthy trees, dead trees, and wildlife trees (i.e., dead trees, live defective trees, and live trees with evidence of foraging sign) > 10 cm dbh in stratified random plots of 11.28 m radius before (1996) and after (1998) treatment. Tree characteristics included tree species, dbh, height, decay class (British Columbia Wildlife Tree Committee 1997), and presence of particular tree defects and disturbance agents. A total of 100 plots (20 plots per treatment unit) were sampled in late June of each year.

Woodpecker Foraging Inventory

We assessed all trees (no diameter limit) in the random plots for evidence of fresh woodpecker foraging activity before (1996) and after (1998) treatment. Freshly "woodpeckered" bark characterized by newly exposed wood was readily distinguished from old foraging sign that appears discolored. We classified foraging sign as scaling, excavations, or sapsucking. This method is conservative in that it excludes foraging that does not leave quantifiable sign, such as gleaning of insects from bark or foliage. To assist with the interpretation of foraging sign data, we made opportunistic observations on foraging woodpeckers during surveys in the sampling areas.

Insect Trapping

We used Lindgren multiple-funnel traps (baited with an ethanol-releasing device and containing a Vapona pest strip)[3] to assess the abundance and diversity of arboreal beetles and other insects likely to serve as prey items for woodpeckers and other cavity-nesting species. Four traps were suspended (between trees with collection cups at 0.5 to 1.0 meter above ground level) in each treatment unit from late April to early September. Catches were collected every 3 weeks and stored in plastic bags at -10°C. Later, insects were sorted to family, identified to species or genus, and tallied. Insect analyses are still in progress and only preliminary; 1996—1998 results are presented here.

Results

Cavity Nest Density and Species Richness

We found a total of 10 cavity-nesting species breeding in the study area: hairy *(Picoides villosus)* and pileated woodpeckers *(Dryocopus pileatus),* northern flicker *(Colaptes auratus),* white- *(Sitta carolinensis)* and red-breasted nuthatches *(S. canadensis),* black-capped *(Parus atricapillus)* and mountain chickadees (*P. gambeli)*, mountain bluebird *(Sialia currucoides),* red squirrel *(Tamiasciurus hudsonicus),* and northern flying squirrel *(Glaucomys sabrinus).* Pre-treatment (1996) cavity nest densities ranged from 2 to 10 nests per 20 ha treatment unit (0.1 to 0.5 nests per ha; *fig. 1*). After harvest treatments were completed in 1997, nest densities in the harvest and burn and harvest only treatments decreased by 50 percent. Those in the burn only treatment (still untreated in 1997) increased by 50 percent, relative to pre-treatment. Nest densities in adjacent and distant controls increased and decreased respectively, during this period. After burning treatments were completed in 1998, nest densities remained 50 percent lower than pre-treatment in the harvest only, and were either comparable to or higher than pre-treatment in all other units. Of the post-treatment cavity nests found in the harvested treatments, 50 percent in the harvest only (1997 and 1998) and 33 percent in the harvest and burn (1998 only) were located in wildlife tree patch-reserves, rather than in harvested portions of the treated units. These reserves comprised only 4.2 percent of the total area harvested, yet they supported up to half the breeding cavity nesters. Although sample sizes are low and statistical comparisons are not possible due to lack of replication, these results suggest that over the short term the harvest only treatment was associated with a decrease in cavity nest densities.

Pre-treatment species richness of cavity nesters was low, ranging from two to four species per treatment unit *(fig. 2).* Following harvest in 1997, nesting species richness decreased in both harvested treatments, but remained constant in the adjacent control, and increased in the burn only and distant control. After burning was completed in April of 1998, cavity nesting species richness was 50 percent lower in the harvest only and higher in all other units, relative to pre-treatment. These trends also suggest a short term decrease in cavity-nesting species richness associated with the harvest only treatment.

[3] Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.

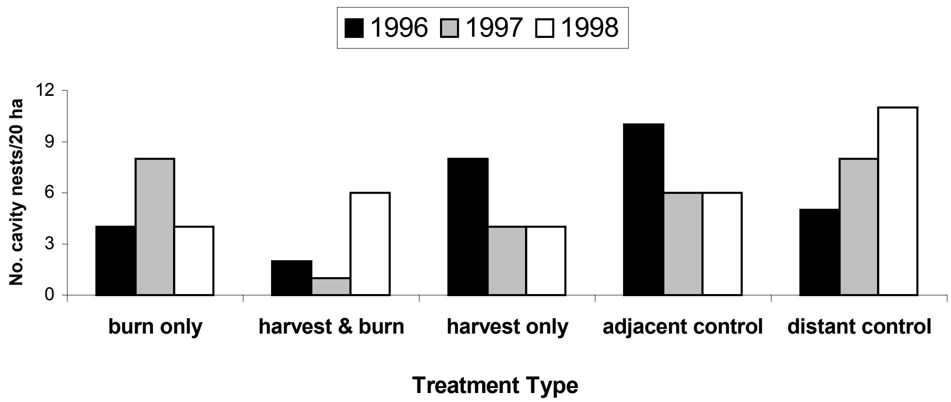

Figure 1—Number of active cavity nests per 20 ha treatment unit by year (1996 to 1998) and treatment type in the Ta Ta Creek operational trial, southeastern British Columbia. Note that 1996 was a pre-treatment, 1998 was a post-treatment year, and 1997 was pre-treatment for burning and post-treatment for harvesting activities.

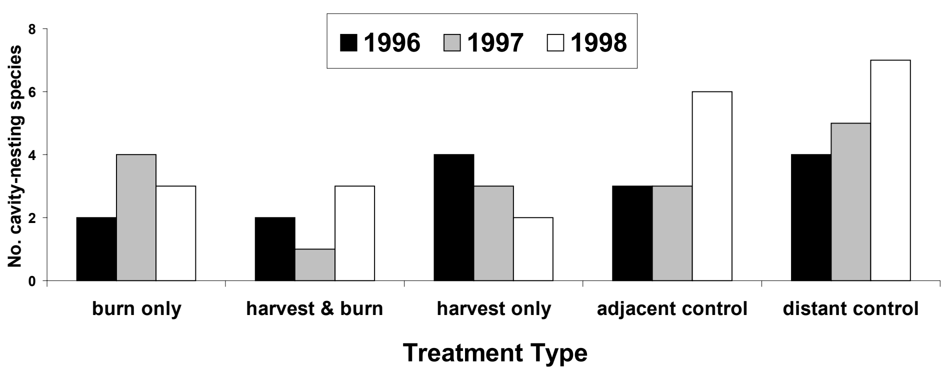

Figure 2—Number of cavity-nesting species per 20 ha treatment unit by year (1996 to1998) and treatment type in the Ta Ta Creek operational trial, southeastern British Columbia. Note that 1996 was a pre-treatment, 1998 was a post-treatment year, and 1997 was pre-treatment for burning and post-treatment for harvesting activities.

The mountain bluebird was the only species to nest in a treated area (burn only) after treatment that was not breeding there before treatment. Conversely, the northern flying squirrel and white-breasted nuthatch did not breed in treated units (harvest only and burn only, respectively) once treatments were completed, although they were nesting there before.

Availability of Wildlife Trees

The harvest and burn and harvest only treatments showed reductions in basal area of 60.6 and 66.4 percent, respectively, as determined during a companion study monitoring vegetation response to treatments (Penniket & Associates 1998). As expected, total densities of trees measuring > 10 cm dbh differed significantly between years and among treatments *(fig. 3)*. In particular, the two harvested treatments showed significant post-treatment declines (two-way Anova: $F = 11.2$, df = 1, $P = 0.001$ for year; $F = 3.66$, df = 4, $P = 0.05$ for treatment; $F = 3.69$, df = 4, $P = 0.006$ for treatment × year).

Snag densities ranged from 2.5 to 12.5 snags per hectare in the treatment units in 1996 compared with 0.1 to 15.0 snags per ha in 1998 *(fig. 3)*. All three treated units had lower snag densities post-treatment, i.e., decreases of 97 percent in the burn only, 98 percent in the harvest and burn and 50 percent in the harvest only, while densities in adjacent and distant controls were 48 percent lower and 50 percent higher, respectively, relative to pre-treatment. Differences among treatments and between years were not statistically significant (two-way Anova: $F = 0.718$, df = 4, $P < 0.581$ for treatment; $F = 1.13$, df = 1, $P = 0.289$ for year; $F = 0.732$, df = 4, $P = 0.571$ for treatment × year).

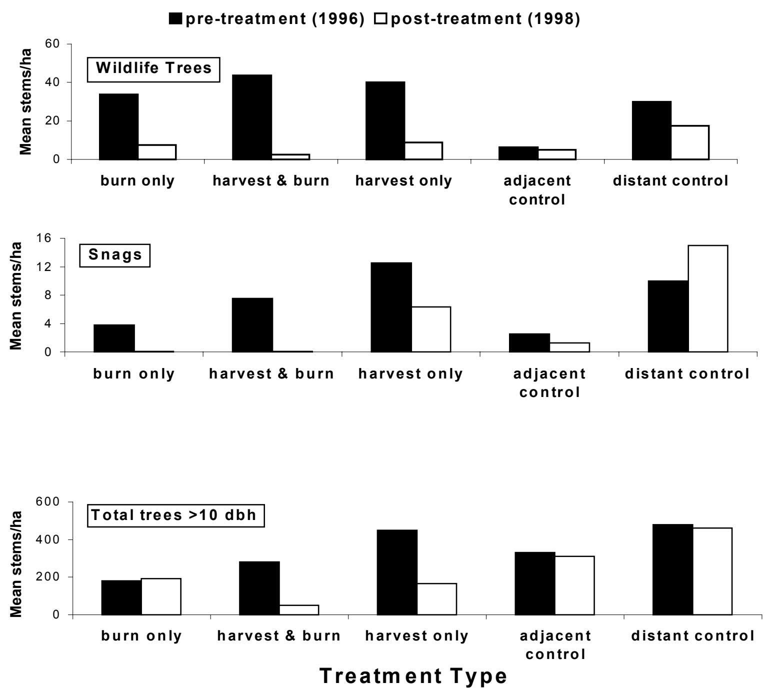

Figure 3—Densities (stems ha^{-1}) of wildlife trees, snags, and all trees (> 10 cm dbh) by treatment type in 1996 (pre-treatment) and 1998 (post-treatment).

Pre- and post-treatment, the densities of wildlife trees were significantly lower in 1998, but differences among treatments and interactions between treatment and year were not (two-way Anova: $F = 20.4$, df = 1, $P < 0.0001$ for year; $F = 2.03$, df = 4, $P = 0.092$ for treatment; $F = 2.00$, df = 4, $P = 0.097$ for treatment × year). The significant year-effect is related to the way wildlife trees were defined (i.e., dead trees, live defective trees, and live trees with evidence of foraging sign or other wildlife use), and reflects the significantly lower level of woodpecker foraging activity found on trees in the overall study area in 1998 (see below).

Woodpecker Foraging Intensity

Woodpecker scaling and excavations accounted for approximately 80 and 20 percent, respectively, of foraging observations in both years (81.4 scaling and 18.6 percent excavations in 1996 compared with 79.3 scaling and 20.7 percent excavations in 1998). The number of trees with fresh woodpecker foraging sign in the treatment units ranged from 5 to 22.5 trees/ha in 1996 and 2.5 to 15 trees/ha in 1998 *(fig. 4)*. Differences between years were significant; however, treatment effects and interactions between treatment and year were not (Two-way anova: F = 4.69, df = 1, $P < 0.0317$ for year; F = 0.444, df = 4, $P = 0.777$ for treatment; F = 0.444, df = 4, $P = 0.777$ for treatment × year). Woodpecker foraging activity on trees was lower in the winter, spring, and early summer preceding our June 1998 observations than in the corresponding seasons prior to 1996. These differences were not treatment related and may be linked to abiotic factors (e.g., temperature, snow pack) affecting the entire study area.

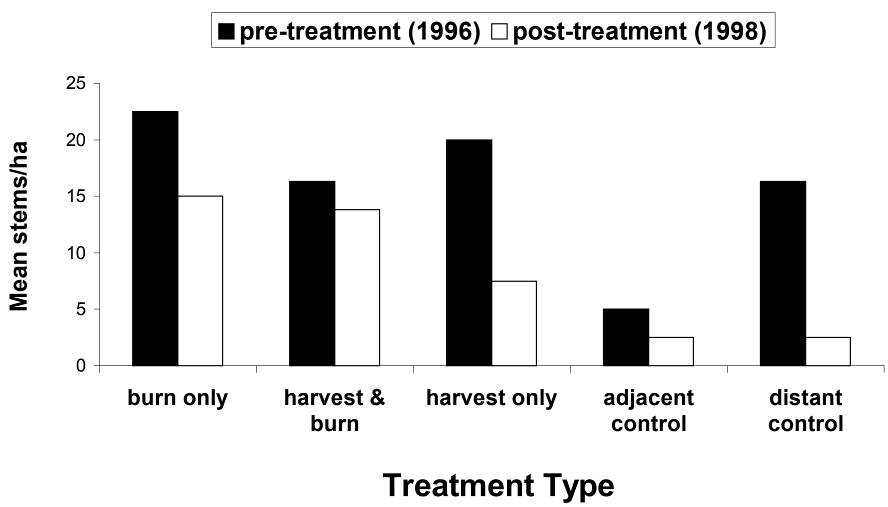

Figure 4—Density (stems ha^{-1}) of trees with fresh woodpecker foraging sign by treatment type in 1996 (pre-treatment) and 1998 (post-treatment).

Abundance and Diversity of Insects

Pre-treatment insect diversity and abundance was similar across treatment units, with a total of 1,919 individuals, 84 species, 57 genera, and 25 families represented in trap catches (Wilson 1999). After treatments were completed in 1998, 10,642 individuals representing 98 species, 68 genera, and 26 families were tallied in catches. Relative to controls, all three treated units showed large (>100 percent) post-treatment (1998) increases in abundance of selected species in the following families:

Buprestidae, Cerambycidae, Cleridae, Hymenoptera, Nitulidae, Oedemeridae, Salpingidae, Scolytidae and Trogositidae. Increases in species abundance, especially *Hylastes* spp., *Hylurgops* spp., *Dendroctonus valens, Asemum nitidum,* and *Rhagium inquisitor,* were greatest in the burn only treatment. Relative to controls, total numbers of insects captured were 170, 54, and 45 percent higher in the burn only, harvest and burn and harvest only treatments, respectively. Increases in post-treatment insect species diversity were also apparent in the families Buprestidae (flat-headed wood borers), Cerambycidae (round-headed wood borers) and Scolytidae (bark beetles). The latter families inhabit dead and dying wood and are associated with forest disturbances (Furniss and Carolin 1977). Preliminary results suggest that insect response to restoration treatments is quite species-specific, but that selected species increased dramatically, with responses most pronounced in the burn only treatment, followed by the harvest and burn and harvest only.

Discussion

Overall, species composition of breeding cavity nesters found in this study was similar to that in nearby Idaho Douglas-fir forests (Medin 1985), although nest densities reported here are considerably lower. Lack of replication prevented me from detecting treatment effects; however, preliminary results do suggest that harvesting (either through greater disturbance and/or direct effects on the landscape) was associated with a decline in cavity nest densities and species richness. The white-breasted nuthatch was the only species to breed at a site pre-treatment and then vacate after treatment, while continuing to nest in controls. This response is consistent with Hejl and others (1995) who reported that in the Southwest, this species was less abundant in clearcut and partial cut ponderosa pine stands relative to unharvested forests.

Several species, such as red-breasted nuthatch, mountain chickadee, and red squirrel present in the harvested areas shifted into untreated wildlife tree patches to nest after harvest treatments were completed. Overall, 40 percent of post-treatment cavity nests were located in wildlife tree patches, compared with only 10 percent pre-treatment. These shifts imply that for some species, untreated patch reserves provide preferred habitat relative to partially logged areas. My findings are consistent with studies demonstrating that red-breasted nuthatches and mountain chickadees are more abundant in unharvested forest than in partial cuts (Medin 1985, Szaro and Balda 1979, Waterhouse and Dawson 1998) or in clearcut stands (Hejl 1994, Hejl and others 1995). Both species have been shown to use residual patches. Their detection rates in patches are comparable to those in contiguous forest and greater than in adjacent logged areas (Gyug and Bennett 1995, Seip and Parker 1997). Similarly, red squirrel populations tend to be highest in mature to old unmanaged forests (Carey 1995), and Klenner (1998) demonstrated that partial cutting treatments involving 20-50 percent volume removal are associated with moderate declines, although this species does use patches of residual snags and green trees.

Shifts of cavity nesters into wildlife tree patches suggest that declines in nest density and species richness were partially offset through patch retention. Patches represented only 4.2 percent of the total area harvested in this operational trial. This is 0.3 percent less than area-based retention requirements still in effect (Province of British Columbia 1999). A greater level of patch retention (or a lighter level of treatment) could potentially minimize impacts of harvesting on the resident cavity-nesting guild. Retention should be based on pre-treatment surveys to delineate the most valuable wildlife tree habitat, rather than operational ease.

I did not detect short-term effects of burning treatments on cavity nest densities or species richness, with the exception of some anecdotal shifts in species composition. This lack of effect is consistent with lesser severity of the burn treatment on stand structure and with other studies investigating effects of prescribed fire on breeding birds in pine forests (Bock and Bock 1983, Finch and others 1997, Horton and Mannan 1988). After treatment in 1998, mountain bluebirds moved into the burn only to breed, whereas northern flying squirrels, which bred there in 1997, vacated in 1998. Mountain bluebirds are closely associated with early post-fire conditions (Finch and others 1997, Hutto 1995), and Bock and Bock (1983) found a greater abundance of this species in burns when compared with unburned forests. Conversely, northern flying squirrels are more abundant in mature to old forests (Carey 1995, Klenner 1998, McDonald 1995). Thus, this species may have been displaced in my study owing to reduced habitat suitability or short term disturbance directly associated with prescribed burning. We did not detect the Lewis's woodpecker *(Melanerpes lewis)* in our study area from 1996 to 1998. In 1999, this species was observed incidentally for the first time, which is consistent with its reported association with early post-fire habitats (Hutto 1995, Saab and Rich 1997).

Snag densities differed in pre- and post-treatment settings. Specifically, snag densities were 97, 98, and 50 percent lower in the burn only, harvest and burn and harvest only, respectively. In the two harvested treatments, snags were removed to comply with occupational health and safety regulations (Workers' Compensation Board of British Columbia 1998). At the time of the harvest, the regulations required that all snags in reach of work areas be felled during harvesting operations, unless buffered in wildlife tree patches or surrounded by no-work zones. Snag densities in the burn only treatment were very low prior to treatment (approximately 3.8 sph). However, I observed several smaller (< 30 cm dbh) fallen burnt snags with advanced decay and previously active nests in late April of 1998 after the burn. These fallen snags were most likely destabilized by prescribed fire, and similar declines in snag density have been documented for other prescribed burns: 56 percent decrease by Gaines and others (1958); 35-48 percent decrease by Gordon (1996); and 45 percent decrease by Horton and Mannon (1988). In 1999, it became apparent that prescribed burning killed many smaller (<15 cm dbh) trees in the two burned treatments (82.5 and 87.3 percent reduction in mean layer 4 sph in the burn only and harvest and burn, respectively; Penniket & Associates Ltd. 1998). These smaller trees were intensively used for foraging by woodpeckers in 1999, but were too small to provide suitable cavity nesting habitat (Machmer 2000).

The increased abundance and diversity of insects found in the burn only (and to a lesser extent in the harvest and burn and harvest only) is consistent with other studies that have reported a greater abundance of buprestid and cerambicid beetle larvae in recent fire-killed and/or harvested trees (Amman and Ryan 1991, Evans 1964, Fellin 1980). I anticipate an increase in woodpecker foraging activity on trees in burned areas in future years, as pupae and larvae of the elevated 1998 insect populations become available prey. Monitoring at this site will continue for another 1 to 2 years.

Management Implications

Successful management of the cavity-nesting bird community during broad-scale application of ecosystem restoration treatments will require a multi-faceted approach that satisfies the diverse requirements of various guild members. Recently revised occupational health and safety regulations in British Columbia (Workers' Compensation Board of British Columbia 1998) permit retention of dead trees in all forestry operations, provided they do not pose a hazard to forest workers. To promote populations of cavity nesters that are associated with semi-open stands (e.g., red-breasted nuthatch, mountain chickadee, northern flying squirrel) during restoration treatments, I recommend that managers attempt to achieve a patchy rather than uniform treatment severity during mechanical treatments, and that wildlife tree patches be established in all areas planned for restoration treatment. Consideration should be given to exceeding current area-based retention requirements (Province of British Columbia 1999) in an attempt to approach historical stand conditions and minimize the likelihood of adverse impacts on the cavity-nesting guild. Furthermore, patch-reserves should be delineated based on pre-treatment surveys to identify patches with features of highest demonstrated value to cavity nesters, such as dead trees with broken tops, evidence of internal decay and/or previous cavity use in the largest available diameter classes (Bull and others 1997, Steeger and Dulisse 2002).

To address the needs of species associated with open habitats, such as Lewis's woodpecker and mountain bluebird, an abundance of high value dead trees should be retained as individuals or clumps in the main portion of treatment areas. The latter should represent a range of decay classes favoring the largest diameter stems, which have the greatest longevity during fire and satisfy the requirements of the most species (Finch and others 1997, Raphael and White 1984).

To minimize loss of snags during prescribed fire treatment, constructing fire lines around them (Gordon 1996) and/or using fire retardant to protect these habitat features should be considered.

Preliminary insect responses to the treatments suggest that enhancement of food supplies for cavity nesters associated with burning may be greater for partial cutting in combination with burning than for partial cutting alone. If further monitoring substantiates this difference, then consideration should be given to the establishment of "burn only" patch-reserves within areas planned for "harvest and burn" restoration treatment. After treatment, these patch-reserves would contain a higher density of standing dead trees charred by fire to support dependent insect populations that serve as prey for cavity nesters.

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