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RESIDUAL TREE RETENTION AMELIORATES SHORT-TERM EFFECTS OF CLEAR-CUTTING ON SOME BOREAL SONGBIRDS

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Abstract. Retention of residual trees in ''cutblocks,'' logged blocks of forest, has been proposed as a method to conserve songbirds in landscapes fragmented by clear-cut logging. We examined songbird communities in the boreal mixed-wood forest of Alberta, Canada, to investigate the effect on songbird abundance of (1) logging and (2) retaining variable densities of residual trees in cutblocks (10–133 trees/ha or basal area of 0.50–10.65 m^2). We surveyed songbirds in logged and forested, aspen-dominated, mixed-wood stands in the year before, the year after, and three years after logging. We analyzed changes in abundance of 27 common songbird species: 23 present in the forest prior to logging and four that appeared after logging. Ten species declined with logging and were termed ''forest species.'' Ten more species did not change with logging and were called ''habitat generalists.'' The seven species that increased with logging were called ''cutblock species.'' When the effect of residual tree retention was examined in terms of basal area (rather than density) of residual trees, more songbird species were found to be both positively and negatively affected by residual tree retention, despite the fact that the two tree measures were highly correlated. In the first year after logging, four bird species (two forest, one generalist, and one cutblock) increased, and none decreased with increasing residual tree retention in cutblocks. In the third year after logging, again four species increased with increasing retention, but these were different species than in the first year after logging (one forest and three generalist species). Furthermore, four cutblock species decreased with increasing retention. Based on these findings, we conclude that retention of residual trees may be beneficial to some species, although conservation of unlogged reserves is also important. Most importantly, we recommend that research be continued to examine a larger range of tree retention and longer term effects on the avifauna.

Key words: Alberta, Canada; bird communities; boreal mixed-wood; clear-cut logging; forest management; forest songbirds; neotropical migrants; partial harvesting; Populus tremuloides; *residual tree retention; songbird conservation.*

Introduction

Since the 1970s, there has been growing concern about declines in North American songbird populations (Robbins et al. 1989). These declines have been attributed to forest fragmentation and habitat loss, both of which are largely due to anthropogenic disturbances such as urban development, agriculture, mining, and forestry (Morse 1980, Robbins et al. 1989, Askins et al. 1990). The maintenance of much of North America's avifauna may depend on our ability to mitigate the effects of human development and resource exploitation on wildlife habitat (Terborgh 1989).

The boreal mixed-wood forest of Alberta, Canada, is among many forests undergoing rapid fragmentation by agriculture, logging, mining, and gas and oil exploration and development (Alberta Environmental Protection 1998). Although much of Alberta's forest is considered to be outstanding in its ecological richness and is classified as threatened or vulnerable habitat by the World Wildlife Fund (Ricketts et al. 1999), 195 662 km^2, or ~51%, of this forest has been leased by the Alberta government to companies for logging. This is equal to ~30% of the total area of the province of Alberta. The boreal mixed-wood forest, which accounts for ~76% of the province's forest, is currently being harvested by clear-cutting in two or three stages or passes, with ~10 yr between passes. This system creates a patchwork of ''cutblocks'' (logged blocks of forest) across the landscape. We have little idea what the long-term effects of such a disturbance will be on the wildlife in the area, nor do we know much about logging systems that might mitigate potentially harmful effects.

Partial harvesting, in which some trees and snags are retained in cutblocks, might mitigate the potentially harmful effects of clear-cutting on the forest ecosystem. Residual trees and snags provide structural and age diversity to the regenerating stand, and may facilitate conifer regeneration by providing shelter and seeds (Rose and Muir 1997). For this and other reasons, residual trees and snags may also provide habitat for wildlife such as forest songbirds. Thinning, strip-cutting, and leaving large clumps of residual trees (~0.54

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ha) have resulted in highly diverse bird communities with fewer species losses than produced from clear-cutting in several forest types (Webb et al. 1977, Szaro and Balda 1979, Freedman et al. 1981, Merrill et al. 1998, Chambers et al. 1999). Residual snags also provide foraging and nesting habitat for woodpeckers (Dickson et al. 1983, Niemi and Hanowski 1984). Faced with growing concern over the detrimental effects of clear-cutting, some forestry companies have begun to retain residual trees and snags in logged stands. However, industrial guidelines concerning the amount and structure of residual tree retention lack a strong basis in scientific research.

Norton and Hannon (1997) found that leaving residual trees benefited forest songbird communities in Alberta's boreal mixed-wood forest one year after logging. Partial cuts included 11–39% residual tree retention in the form of numerous single and relatively small patches of trees and snags (range 0.0001–1.1474 ha; 0.0251 ± 0.0018 ha, mean ± 1 SE) distributed at random throughout cutblocks. These cutblocks ranged in size from 10.1 to 30.8 ha. Total bird species richness and abundance were higher in partial cuts than in clearcuts in the year immediately following a winter logging, although both were lower in partial cuts than in forested control stands (Norton and Hannon 1997). Most of the species negatively affected by logging were shrub and tree nesters (Norton and Hannon 1997).

Norton and Hannon's study, however, included only the first breeding season following logging. Songbird communities may continue to change for years or even decades after disturbance (Welsh 1987, Thiollay 1992, Pojar 1995, Schieck and Nietfeld 1995). If sites with high densities of residual trees (partial-cut sites) represent suboptimal habitat for forest-dependent species, abundances of these species may continue to decrease in the first few years following logging because of low reproductive success and recruitment. Norton and Hannon's (1997) study sites were logged in the winter. However, most migratory songbirds identify potential breeding habitat during the dispersal phase at the end of the breeding season, and return to these identified sites in the following season (Brewer and Harrison 1975, Morton 1992). Hence, migratory species that prefer to nest in logged rather than forested stands would probably not be recorded in cutblocks until at least the second year following logging. In addition, breeding philopatry of forest-dependent birds in the first breeding season after logging might increase abundance in cutblocks over that recorded in subsequent seasons.

With these considerations in mind, we continued and expanded on Norton and Hannon's (1997) original study, repeating the bird survey in the third year after logging and reanalyzing the original data to render it more useful to forest managers. Our first objective was to examine the effect of logging on songbirds by (1) examining changes in abundance of species between the year before and the first and third years after logging, and (2) comparing these patterns of change between cutblocks and controls. In this way, we aimed to identify forest songbird species as those negatively affected, cutblock species as those positively affected, and generalist species as those unaffected by logging. Our second objective was to investigate the relationship between the amount of residual tree retention on cutblocks and the abundance of forest, generalist, and cutblock songbird species.

Methods

Study area and experimental design

This study was carried out in an area of ~ 10 × 15 km in the boreal mixed-wood forest, just north of Calling Lake, Alberta, Canada (55°15′N, 113 ° 35′W). The boreal mixed-wood forest is dominated by a combination of hardwoods (trembling aspen [*Populus tremuloides*] and balsam poplar [*P. balsamifera*]) and softwoods (white spruce [*Picea glauca*], black spruce [*P. mariana*], and jack pine [*Pinus banksiana*]; Rowe 1972). Although the forest is naturally pyrogenic, it is now the focus of extensive logging and gas and oil exploration and development, resulting in a landscape fragmented by cutblocks, seismic lines, pipelines, roads, and clearings for gas wellheads (Alberta Environmental Protection 1998).

Study sites were chosen in the spring of 1994, using harvest plan maps as a guide, in continuous, aspen-dominated, mixed-wood stands >130 yr old. Twelve sites were used in 1994 and 1995 and 18 sites in 1997 (the original 12 plus six additional sites; Fig. 1). All sites were 10–35 ha in size and were a minimum of 300 m and a maximum of 4000 m apart. Forested control sites were ≥400 m from any cutblocks. To control for possible adjacency effects, all sites were located adjacent to old, aspen-dominated, mixed-wood forest, and at least 50 m from any bog, wetland, or conifer-dominated stands. In the original design (Norton and Hannon 1997), nine stands slated for harvest were selected and treatments were applied randomly after being stratified by block size: small (~10–15 ha), medium (~17–26 ha), and large (~28–31 ha). The original treatments (Norton and Hannon 1997) consisted of clearcuts with three levels of residual tree retention (~10%, 30%, 40%). Three forested control stands, one of each size class, were also selected. The design of pre- and post-treatment measures on the same sites with simultaneous controls (BACI design) allows for a powerful test of treatment effects in a varying environment (Stewart-Oaten et al. 1986). The six sites added in 1997 increased the range of tree retention and included three more controls, although we did not have pre-harvest data or first-year harvest data for these sites. In the winter of 1994–1995, nine of the original 12 and three of the six sites added in 1997 were logged using feller-bunchers, skidders, and role-stroke delimbers (Norton and Hannon 1997). Operators were instructed to leave

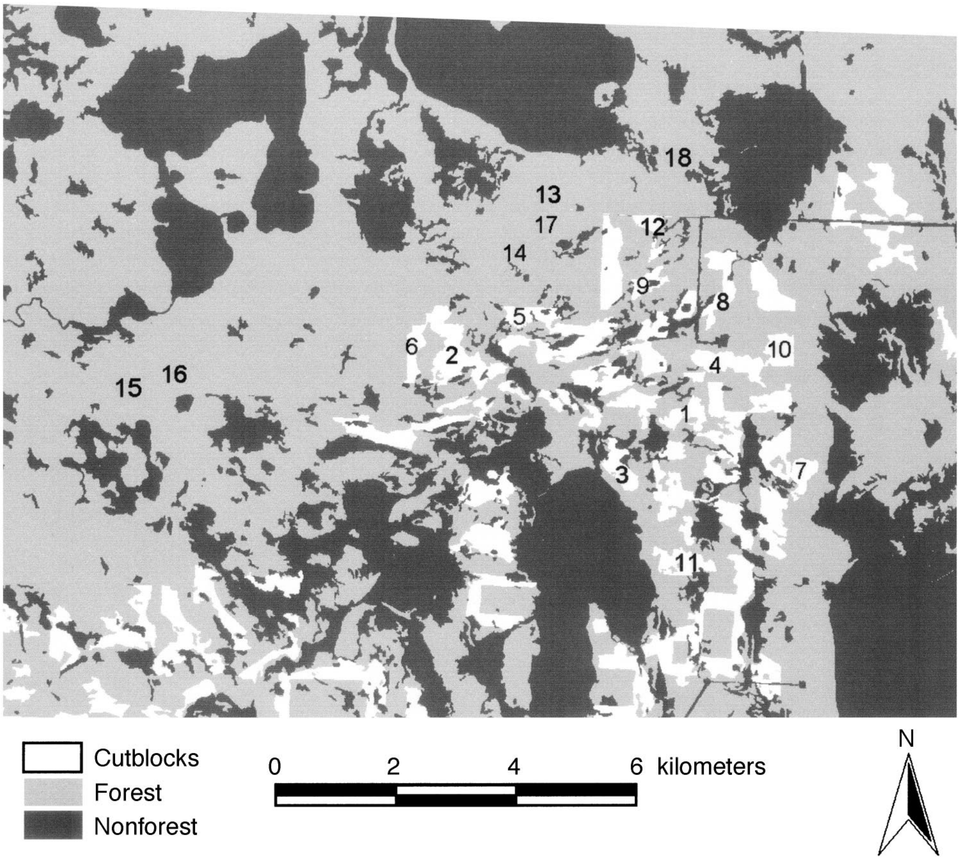

FIG. 1. Study area located east of Calling Lake, Alberta, Canada, showing locations of study sites, numbered in order of increasing tree density. Numbers 1–12 are cutblocks (blocks 1–3 were added in 1997), whereas 13–18 are forested controls (15,16, and 18 were added in 1997).

clumps of residual trees evenly dispersed over the cutblocks, regardless of tree species. Nonmerchantable trees were left on the blocks, according to the usual logging practices. Operators were also asked to leave a specific range of retention in each of the nine original sites, but retention on the three added sites was not specified before harvest. Because operators were not very successful in attaining the prescribed levels of retention, we abandoned a blocked statistical design by treatment (with the exception of comparing all cutblocks to controls), and relied instead on a regression approach (amount of trees retained vs. bird abundance).

Bird counts

Count stations were placed 200 m apart in each study site, and birds were counted using a 5-min, 100-m fixed radius point-count technique (Ralph et al. 1993). We established 2–6 count stations per site, depending on the area of the site. This gave us 39 cutblock and 14 control count stations in 1994 and 1995, and 54 cutblock and 26 control stations in 1997. In cutblocks, all stations were $\geq$50 m from the edge of the cutblock. Each site was surveyed between dawn and 1000 at $\sim$10-d intervals, three times from the first week of June to the first week of July 1994, and four times from the last week of May to the first week of July 1995 and 1997. To minimize observer and time-of-day biases, observers rotated so that each site was surveyed by at least two different observers; the order of survey was arranged so that all sites were surveyed at dawn at least once each season. Observers were instructed not to record any individual bird at more than one station; to facilitate this, adjacent stations were surveyed one after the other in all years. Data sheets indicating approximate distance (''within 50 m'' or ''between 50 and 100 m'' from count station) and direction of recorded birds were later reviewed and compared with site maps so that all birds recorded outside of the sample sites were excluded from analyses. Finally, because judgments of distance may vary between controls and cutblocks (a bird singing in a cutblock may seem closer than the same species of bird singing from the same distance in a forested control), observers were trained to judge distance in cutblocks and controls. Visibility may also be greater in cutblocks than in controls, but 97.8% of the observations were auditory; thus, we were not overly concerned with this potential source of error. Singing males were assigned an abundance score of 1, whereas calling and silent birds were assigned an abundance score of 0.5. Counts were not performed under windy (>3 on the Beaufort scale; Ahrens 1993) or rainy conditions.

Because count stations were 200 m apart and the count radius was 100 m, there was some probability of counting one individual as two or more individuals by recording it at different stations on different visits. To minimize ''double counting'' individuals, we calculated abundances per species as the maximum number of observations per site (all stations combined) in any one of the three or four count rounds. We divided this total by the number of stations to give a mean abundance per station. We did not calculate density per hectare because point counting is not an appropriate technique for this. To reduce potential observer error in species identification, we excluded any species recorded only once in either the forested controls or cutblocks. We also excluded all observations of birds flying over the sites, as this does not represent use of the block. Only songbirds (members of the order Passeriformes) were included in analyses. We excluded Black-capped Chickadees (*Poecile atricapillus*) because they breed early in the season and were not adequately sampled when we performed the counts. For comparisons of the same sites over years, we used data from only three count rounds (matched by date), because only three rounds were conducted in 1994.

Residual tree retention

We calculated residual tree retention in two ways: the number of trees per hectare and basal area. Trees and snags with dbh (diameter at breast height) >15 cm were counted on three 0.04-ha vegetation plots 30 m from each point-count station at 0°, 120°, and 240°. We calculated the density of residual trees per hectare by dividing the total number of trees and snags counted in the three plots by 0.12 ha (0.04 ha × 3). We calculated average basal area on the three plots using the formula $[\pi(0.5\text{dbh})^2]$, and then divided this number by 0.12 ha to give the basal area per hectare. Means of both of these measures were calculated for each site. The three sites added to the study in 1997 had low residual tree density and low basal area; they expanded the range studied from 36–133 trees/ha (or 11–39% of the initial trees retained) in 1995 to 10–133 trees/ha (or 3–39% of the initial trees retained) in 1997, or from a basal area of 2.03–10.65 m^2 in 1995 to 0.50–10.65 m^2 in 1997 (from 7–46% basal area retention in 1995 to 2–46% in 1997; approximate retention values for new 1997 sites are calculated by dividing post-cut values by mean pre-cut values for 1995 sites).

Analyses

In all analyses of bird count data, we used an α level of 0.10 to reduce the probability of committing a Type II error (Schmiegelow et al. 1997). We tested the power of each analysis using the program GPOWER (Faul and Ehrdfelder 1992), setting the effect size d at 0.04 birds per point-count station, a difference of three birds between stations. If power was moderate to high ($\geq$0.65, α = 0.10; Cohen 1988), we performed conventional statistical tests, selecting between parametric and nonparametric tests based on homogeneity of variance, examined using a Levene test, and normality, examined using a Kolmogorov-Smirnov goodness-of-fit test (Zar 1984). If power was low (<0.65, α = 0.10; Cohen 1988), we replaced t tests with randomized t tests and linear regressions with randomized linear regressions (Edgington 1987) using the program RT (Manly 1996). We used SPSS version 7.5.1 (Norusis 1995) for all other analyses.

Although we present data for all species detected in the Appendix (available in *Ecological Archives*), this paper focuses on common songbird species. We define these as species for which $\geq$10 individuals were recorded in all study sites combined in any one of the three years of this study (Schieck and Nietfeld 1995). Using this criterion, the data set could include species recorded only in one site, although they would have to be fairly abundant there ($\geq$10 recordings). Although rarer species are probably an integral part of the community and may disappear rapidly in response to habitat loss (Connor and McCoy 1979, Haila 1983), this study does not have a large enough sample size to include species that are naturally rare in the area. We follow the American Ornithologists' Union (1998) for common and scientific names.

Vegetation left on cutblocks.—To verify how well logging operators had followed the instructions to leave residual trees regardless of timber type, we compared vegetation variables before and after logging. We used Wilcoxon signed-rank tests to perform pairwise comparisons of the percentage of small (15–23 cm dbh), medium (23–38 cm dbh), and large (>38 cm dbh) trees and the percentage of deciduous trees, conifers, and snags recorded before and after logging. Each analysis was performed separately for cutblocks and controls. Percentages were calculated by dividing the number of each type of tree per hectare by the total number of trees per hectare. We also performed a Spearman rank correlation between basal area and the number of trees per hectare in cutblocks and controls.

Differences in bird abundance between years and between controls and cutblocks.—We examined changes in abundances of common songbird species and total songbirds from pre- to post-harvest in all cutblocks combined vs. controls using Friedman analyses of variance by ranks, the nonparametric equivalent of repeated-measures ANOVAs, with post hoc analyses according to Zar (1984). The Friedman test did not allow us to test for interactions between year and treatment (logged vs. controls); hence, we could not determine whether abundance changes over the years were caused by logging or were the reflection of a landscape or regional change in abundance, not related to treatment.

TABLE 1. Mean (and 1 SE) values for vegetation variables, and results of pre- and post-logging pairwise comparisons (Wilcoxon signed ranks) in three plots of boreal mixed-wood forest in Alberta, Canada.

Variable†	Pre-Cut	Post-Cut	Z	P
Cutblocks				
No. trees/ha	301.04 (19.39)	78.24 (12.48)	−2.660	0.008
Basal area (m^2)	25.71 (1.44)	6.21 (1.08)	−2.660	0.008
Small trees (%)	29.66 (3.26)	48.90 (9.13)	−1.481	0.139
Medium trees (%)	45.13 (3.34)	34.21 (6.76)	−1.125	0.260
Large trees (%)	25.21 (2.57)	16.89 (6.50)	−1.125	0.260
Conifer (%)	4.93 (1.53)	2.86 (2.86)	−1.400	0.161
Deciduous (%)	77.41 (3.05)	73.56 (8.53)	−0.296	0.767
Snags (%)	17.67 (2.36)	23.59 (7.94)	−0.889	0.374
Controls				
No. trees/ha	303.21 (10.28)	333.82 (26.11)	0	1.000
Basal area (m^2)	26.94 (0.98)	28.72 (1.72)	−0.365	0.715
Small trees (%)	30.89 (7.03)	40.7 (3.63)	−1.069	0.285
Medium trees (%)	45.03 (2.82)	35.50 (4.21)	−1.604	0.260
Large trees (%)	24.08 (4.23)	23.08 (5.06)	0	1.000
Conifer (%)	8.96 (4.14)	20.4 (4.99)	−0.535	0.593
Deciduous (%)	64.42 (8.72)	61.4 (6.26)	0	1.000
Snags (%)	26.62 (4.95)	18.21 (2.25)	0	1.000

Notes: Vegetation data were collected, before and after logging, 30 m from each point-count station at 0°, 120°, and 240°. Pre-logging data were not available for the three cutback and three control sites added in the third year after logging. Hence, they are not included in this analysis.

† Tree sizes are defined as follows: small, 15–23 cm dbh; medium, 23–38 cm dbh; large, >38 cm dbh.

Therefore, we examined differences in abundances between cutblocks and controls in each year, using *t* tests, Mann-Whitney *U* tests, or randomized *t* tests, depending on normality, homogeneity of variance (Zar 1984), and power (Cohen 1988). Because only three count rounds were performed on nine future cutblocks in the year before logging, only data from three count rounds and these nine cutblocks were considered in post-logging years. To be conservative in our interpretation (i.e., to avoid a Type II error), we report trends for analyses where $0.10 < P < 0.20$.

Effect of residual tree retention.—Depending on the power of the tests, we used linear regression or randomized linear regression models to investigate the effect of tree density and basal area on common species abundances, total songbird abundance, and total abundance of species grouped with respect to whether they declined, increased, or did not change in abundance after logging. We analyzed data from all nine cutblocks surveyed in 1995 and all 12 cutblocks surveyed in 1997. Analyses included data from all four count rounds performed in both post-logging years.

RESULTS

Amount and composition of trees left on cutblocks

The number of trees per hectare and basal area of trees were reduced considerably by the harvesting (Table 1). Basal area and number of residual trees per hectare were highly correlated in both cutblocks (r_S = 0.987, P = 0) and controls (r_S = 0.975, P = 0.001). The size distribution of trees left in cutblocks did not change with logging, nor did the makeup of the tree community. The percentages of small, medium, and large trees were the same after as before logging in cutblocks and in controls (Table 1). Similarly, the percentages of conifers, deciduous trees, and snags did not change with logging (Table 1).

Total songbird abundance after logging

The total abundance of songbirds decreased significantly between 1994 and 1995 in cutblocks, but increased between 1995 and 1997, so that mean abundances per point-count station in the third year after logging were not significantly different from pre-logging abundances (1994, 11.4 ± 0.47; 1995, 5.7 ± 0.95; 1997, 10.0 ± 0.41; all values are mean ± 1 SE; Friedman χ^2 = 15.6, df = 2, $P < 0.01$). The total abundance increased gradually in controls over the three years, being significantly greater in 1997 than in 1994 (1994, 10.4 ± 0.60; 1995, 12.8 ± 0.91; 1997, 13.6 ± 0.71; Friedman χ^2 = 4.5, df = 2, P = 0.10). Furthermore, although abundance did not differ between controls and future cutblocks before harvest (t = 1.198, df = 10, P = 0.26), numbers were significantly higher in controls than cutblocks in both years after harvest (1995, t = −4.036, df = 10, $P < 0.01$; 1997, t = −4.419, df = 10, $P < 0.01$). This suggests a general depression of songbird abundance on the logged sites.

Species negatively affected by logging

Of the 27 common species analyzed, 23 were present in the forest prior to logging (see the Appendix). Of these, 10 species appeared to be negatively affected by logging (Fig. 2). Four of the 10 species either disap-

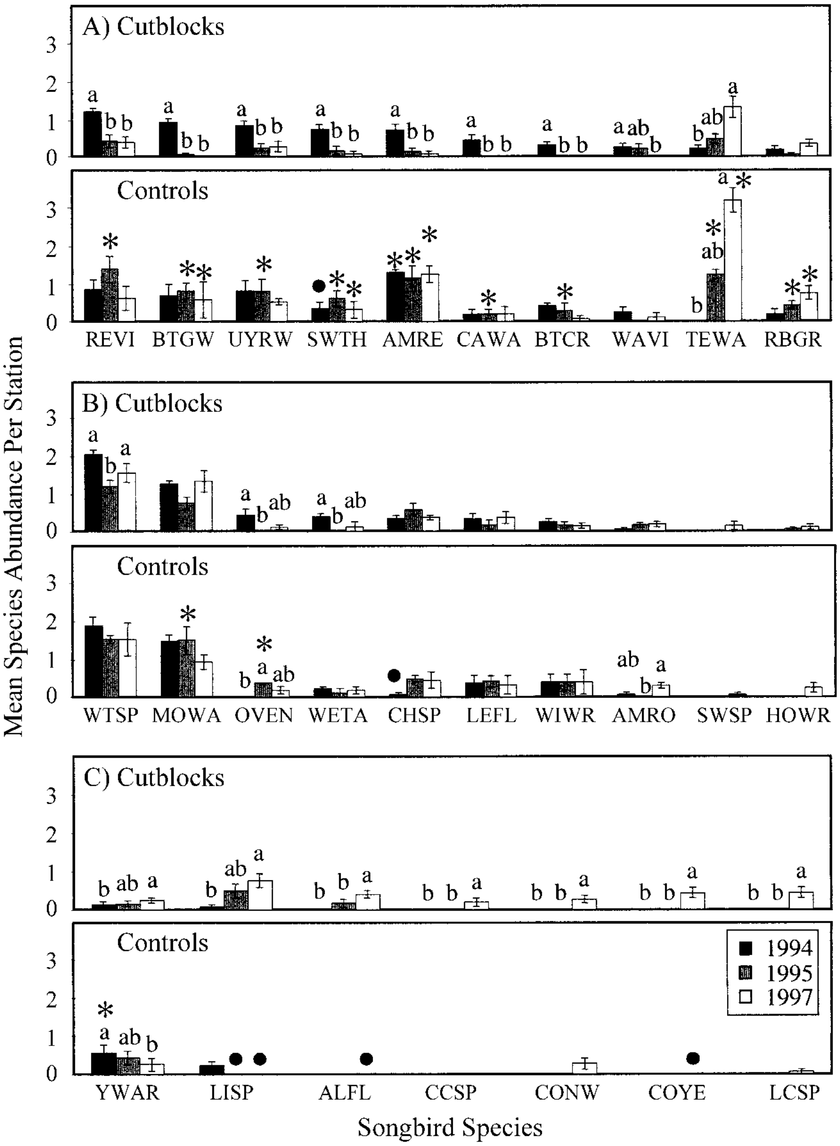

FIG. 2. Songbird species abundances (mean ± 1 SE) in cutblocks vs. controls in the year before logging (1994) and the first and third year after logging (1995 and 1997) for: (A) forest species, (B) habitat generalists, and (C) cutblock species. Asterisks indicate a significantly greater abundance in controls than in cutblocks, and solid circles denote a significantly lower abundance in controls than in cutblocks ($P \leq 0.10$). Friedman tests were used to compare abundances between years. Letters (a, b, and ab) mark homogeneous groups where differences were significant ($P \leq 0.10$). Species codes follow Gustafson et al. (1997); see Table 2 for common names and the Appendix for scientific names.

peared or were extremely rare in cutblocks after logging, while they were still present in control sites (Black-throated Green Warbler, Canada Warbler, Brown Creeper, and Warbling Vireo). Four other species declined in abundance in the cutblocks after harvest, while remaining stable or significantly higher in controls (Red-eyed Vireo, Yellow-rumped Warbler, Swainson's Thrush, and American Redstart). The Red-eyed Vireo showed a trend ($P = 0.18$) to decline in control sites but not cutblocks in 1997, suggesting a regional decline in numbers. Two species increased either significantly (Tennessee Warbler) or only slightly (Rose-breasted Grosbeak) in abundance in cutblocks and controls, but were more abundant in controls than cutblocks following logging. This suggests that a regional increase in these species had occurred, but the increase was not as high in cutblocks as in controls. We defined ''forest species'' as those that either declined in abundance in cutblocks or increased significantly less in cutblocks than in controls.

Species that appeared to be unaffected by logging

The abundances of 10 of the 23 species that had been present prior to harvest appeared to be unaffected by logging. Least Flycatcher, Chipping Sparrow, Swamp Sparrow, Winter Wren, and House Wren showed no significant change in abundance in either cutblocks or controls following logging, and abundances in controls were not higher than in cutblocks after harvest (Fig. 2). Similarly, the American Robin did not change in abundance in cutblocks and showed no differences in abundance between cutblocks and controls in either post-logging year, although it was significantly more abundant in controls in the third than in the first year after logging. The White-throated Sparrow declined in the first year after logging in cutblocks, but rebounded by the third year after harvest; this resulted in similar abundances between controls and cutblocks by the third year. The Mourning Warbler showed no significant change in post-harvest abundance in cutblocks and controls. Although abundances in controls were higher than in cutblocks in the first year after harvest, they were not significantly different by the third year after harvest. Both the Ovenbird and Western Tanager declined in abundance in cutblocks in the first post-harvest year, but then increased by the third year and were not significantly different from controls, suggesting a recovery after an initial decline. Ovenbird numbers, however, tended to be higher in the future cutblocks in 1994 than in the controls ($P = 0.15$), were higher in controls than cutblocks in 1995, and had similar but very low abundance in controls and cutblocks in 1997. Thus, for Ovenbirds and Western Tanagers, the evidence for no effect of logging is weak. Nevertheless, we have termed these 10 species ''habitat generalists,'' bearing in mind the reservations about Ovenbirds and Western Tanagers.

Species positively affected by logging

Three of the 23 species present prior to logging increased in cutblocks relative to controls (Yellow Warbler, Lincoln's Sparrow, and Alder Flycatcher; Fig. 2). An additional four species (Clay-colored Sparrow, Connecticut Warbler, Common Yellowthroat, and LeConte's Sparrow) invaded the cutblocks by the third year after harvest. The LeConte's Sparrow showed a trend for higher abundance in cutblocks than controls in 1997 ($P = 0.14$). The Connecticut Warbler, however, was also found in similar abundance on the controls and cutblocks in 1997, suggesting that it might be better classified as a habitat generalist. However, for the purposes of this study, we subsequently refer to these seven species as ''cutblock species.''

Effect of residual tree retention on species abundance

We analyzed the effect of both number of trees per hectare and basal area of trees left in the blocks on the abundances of birds. Despite the fact that the two measures were highly correlated, more significant effects were documented using basal area in 1997 (in 1995 results were similar). Hence we present results for basal area to be more conservative. In the first year after logging, abundances of American Redstart, Mourning Warbler, Swainson's Thrush, and Yellow Warbler, total songbird abundance, and total abundance of generalist species increased with increasing basal area on cutblocks (Table 2). In the third year after logging, the abundance of Yellow-rumped Warbler, Least Flycatcher, Winter Wren, and Ovenbird, and the total abundance of generalist species increased with increasing basal area (Table 2). Lincoln's Sparrow, Connecticut Warbler, Common Yellowthroat, and LeConte's Sparrow, and the total abundance of cutblock species decreased in abundance with increasing basal area (Table 2).

Discussion

Forest managers have two options for retaining forest songbirds that are sensitive to logging in harvested landscapes: they can leave unharvested forest reserves or they can partially or selectively harvest stands in hope of retaining some forest species. Our study indicates that retaining trees on cutblocks, at least at the levels we studied (10–133 trees/ha, basal area 0.50–10.65 m^2), will benefit some songbirds up to three years after logging, although unharvested areas are more beneficial to more species. Furthermore, we found more species to be affected by retention when it was measured in terms of basal area rather than in terms of number of trees per hectare, despite the fact that the two measures were highly correlated. Thus, it is important to consider the size, and not just the number, of trees retained. Our data provide some support for the conclusions of several other studies of tree retention (Webb et al. 1977, Freedman et al. 1981, Merrill et al. 1998, Schieck et al. 2000) and the finding of Norton and Hannon (1997) that retention of residual trees is beneficial to songbirds. However, it is interesting to note that the species positively affected by retention in the first year after logging were not the same as those affected in the third year.

Forest species

Of the 10 forest species negatively affected by logging, only the American Redstart and Swainson's Thrush benefited from high retention in the first year after logging, and only the Yellow-rumped Warbler benefited in the third year after logging. Nine of the species that we defined as ''forest species'' (Black-throated Green Warbler, American Redstart, Brown Creeper, Rose-breasted Grosbeak, Tennessee Warbler, Warbling Vireo, Yellow-rumped Warbler, Red-eyed Vireo, and Swainson's Thrush) are foliage gleaners and/or nest in shrubs or trees (Ehrlich et al. 1988). Clearly, for most of these species, retention of >40% of the trees and >46% of tree basal area in cutblocks would be required to maintain them in the forest. More study is required to determine what that level of tree retention would be. Given the restrictions of working with feller bunchers, it might not be operationally fea-

TABLE 2. Results of linear and randomized linear regressions of abundances of common songbird species vs. basal area in logged sites the year after (1995) and three years after logging (1997) in Alberta, Canada. Significant test results ($P < 0.10$) are presented in boldface.

Species	Code†	1995 R^2	1995 B‡	1995 F	1995 P	1997 R^2	1997 B‡	1997 F	1997 P
Forest species									
Warbling Vireo	WAVI	0.056	0.236	0.41	0.540	···	···	···	···
Red-eyed Vireo	REVI	0.072	0.269	0.55	0.484	0.037	0.191	0.38	0.551
Brown Creeper	BRCR	0.004	0.067	0.03	0.865	···	···	···	···
Swainson's Thrush	SWTH	**0.442**	**0.665**	**5.55**	**0.051**	0.242	0.492	3.19	0.172
Tennessee Warbler	TEWA	0.008	0.088	0.06	0.822	0.036	0.190	0.37	0.554
Yellow-rumped Warbler	UYRW	0.199	0.446	1.74	0.229	**0.353**	**0.594**	**5.46**	**0.042**
Black-throated Green Warbler	BTGW	0	0.014	0.01	0.971	0.047	−0.218	0.50	0.497
American Redstart	AMRE	**0.429**	**0.655**	**5.25**	**0.056**	0.008	−0.089	0.08	0.783
Canada Warbler	CAWA	···	···	···	···	···	···	···	···
Rose-breasted Grosbeak	RBGR	0.050	0.224	0.37	0.562	0.013	−0.112	0.13	0.729
Total forest species		0.286	0.535	2.80	0.142	0.183	0.427	2.22	0.154
Cutblock species									
Alder Flycatcher	ALFL	0.158	0.397	1.31	0.290	0.166	−0.407	2.00	0.176
Yellow Warbler	YWAR	**0.629**	**0.793**	**11.86**	**0.011**	0.005	0.068	0.05	0.833
Connecticut Warbler	CONW	···	···	···	···	**0.237**	**−0.486**	**3.08**	**0.091**
Common Yellowthroat	COYE	···	···	···	···	**0.290**	**−0.538**	**4.08**	**0.071**
Clay-colored Sparrow	CCSP	0.057	−0.239	0.43	0.535	0.061	−0.247	0.65	0.439
LeConte's Sparrow	LCSP	···	···	···	···	**0.218**	**−0.467**	**2.80**	**0.099**
Lincoln's Sparrow	LISP	0.238	−0.488	2.19	0.182	**0.368**	**−0.607**	**5.82**	**0.037**
Total cutblock species		0.048	0.219	0.35	0.572	**0.315**	**−0.561**	**4.59**	**0.058**
Generalist species									
Least Flycatcher	LEFL	0.266	0.516	2.56	0.137	**0.271**	**0.521**	**3.71**	**0.083**
House Wren	HOWR	0.311	−0.558	3.19	0.127	0	−0.020	0.01	0.951
Winter Wren	WIWR	0.271	0.521	2.61	0.139	**0.247**	**0.497**	**3.26**	**0.090**
American Robin	AMRO	0.004	−0.065	0.03	0.867	0.002	0.042	0.02	0.897
Ovenbird	OVEN	···	···	···	···	**0.254**	**0.504**	**3.40**	**0.095**
Mourning Warbler	MOWA	**0.626**	**0.791**	**11.69**	**0.011**	0.054	0.232	0.57	0.469
Western Tanager	WETA	0.261	0.511	2.48	0.111	···	···	···	···
Chipping Sparrow	CHSP	0.207	0.455	1.82	0.219	0.185	0.430	2.23	0.173
Swamp Sparrow	SWSP	···	···	···	···	0.055	−0.234	0.58	0.464
White-throated Sparrow	WTSP	0.028	0.168	0.20	0.666	0.046	0.213	0.48	0.506
Total generalist species		**0.541**	**0.736**	**8.26**	**0.024**	**0.353**	**0.594**	**5.46**	**0.042**
Total		**0.429**	**0.655**	**5.27**	**0.055**	0.026	−0.160	0.26	0.619

Notes: Data were analyzed from nine cutblocks in 1995 and 12 cutblocks in 1997; all data sets include four point-count rounds. Ellipses indicate that the species was not recorded in that year.

† Species codes follow Gustafson et al. (1997); common names follow American Ornithologists' Union (1998); see the Appendix for scientific names.

‡ A positive slope (B) indicates an increase in bird species abundance with increasing basal area; a negative slope indicates a decrease in abundance with increasing basal area.

sible to retain high levels of trees on blocks; unharvested reserves might be a better solution.

Although none of the forest species that we identified is declining in the Canadian province of Alberta, four of them are declining across North America (Canada Warbler, Rose-breasted Grosbeak, Tennessee Warbler, and Swainson's Thrush; Sauer et al. 1999). Retention of residual trees did not consistently affect the abundance of any of these species. Our study indicates that logging of the boreal mixed-wood forest may contribute to the decline of these four species. In particular, Canada Warbler and Swainson's Thrush have been shown to prefer old-growth stands in the boreal mixed-wood forest (Schieck and Nietfeld 1995). Therefore, maintenance of old-growth reserves may be crucial to the conservation of these two species. The only forest species to benefit from residual tree retention in our study, the Yellow-rumped Warbler, has increased in abundance in Alberta and across North America (Sauer et al. 1999).

Generalist species

Total generalist species abundance increased with higher retention in 1995 and 1997, and Least Flycatcher, Winter Wren, and Ovenbird also increased on high-retention stands in 1997. These three generalists have been associated with mature (50–65-yr-old) or old (120-yr-old) mixed-wood stands in Alberta's boreal forest (Schieck and Nietfeld 1995) and elsewhere (Pojar 1995), and have been found to be negatively affected by various types of logging (Webb et al. 1977, Robinson and Robinson 1999). The high residual tree retention in some sites in our study allowed the retention of these species in cutblocks. Of the three generalist species that increased with increasing residual tree basal area in the third year after logging, the Ovenbird and Winter Wren have increased in abundance across North America, but the Least Flycatcher has decreased (Sauer et al. 1999). Thus, residual tree retention may be beneficial to the conservation of at least one declining species.

Cutblock species

Three cutblock species (Yellow Warbler, Lincoln's Sparrow, and Alder Flycatcher) were present at low numbers in the forest prior to logging and moved into the cutblocks in the first year after logging. Four species (Clay-colored Sparrow, LeConte's Sparrow, Common Yellowthroat, and Connecticut Warbler) appeared after the first post-logging year, but were present in other habitat types in the landscape (e.g., riparian areas). All but the Yellow Warbler have been associated with early successional stages or young boreal mixed-wood stands in other studies in the area (Schieck and Nietfeld 1995, Hobson and Schieck 1999). The delay in colonization of cutblocks by some species until after the first post-logging year may be related to the increasing suitability of the habitat as aspen in the cutblocks regenerated (up to 1–2 m in height by the third year after logging; R. Tittler, *personal observation*) or to a lag in finding the cutblocks by species not found in adjacent unharvested stands. Clay-colored Sparrow, Yellow Warbler, and Alder Flycatcher were unaffected, and the other four species decreased in abundance with higher tree retention in cutblocks in the third year after logging, as did the total abundance of cutblock species. Hence, retention of high levels of residual trees did not benefit, and may have harmed, cutblock species that otherwise benefit from clear-cut logging.

Of the seven cutblock species, Clay-colored Sparrows have declined over the long term (1966–1998) across North America and in Alberta, and the Common Yellowthroat and Connecticut Warbler have declined slightly over the short term (1980–1998) and long term across North America (Sauer et al. 1999). Cutblock species in general, and the Connecticut Warbler and the Lincoln's Sparrow in particular, may benefit from logging in the boreal forest, and some may also benefit from retention of low, rather than high, densities of residual trees. However, most of these species are typical of the aspen-parkland rather than the boreal mixed-wood forest (Hobson and Schieck 1999), and may be better managed in areas farther south. One exception is the Connecticut Warbler, which is associated with postfire stands in the boreal mixed-wood forest (Hobson and Schieck 1999). To retain this species, it would be better to leave fewer, rather than more, residual trees on cutblocks or to allow some natural fires to occur.

Limitations of study design

The presence of a few individual forest songbirds in cutblocks may indicate that some suitable habitat was available there. However, the limitations of the point-count technique must be considered in interpreting these results. Although this technique assumes that singing birds represent breeding pairs (Ralph et al. 1993), individuals may not be breeding where they are singing (Ambuel and Temple 1983). Selecting singing posts in cutblocks may allow for greater song transmittance (Kroodsma 1984), but the nest sites may be in adjacent forested stands. In our study, all cutblocks were adjacent to old forested stands, which would provide adequate nesting opportunities for forest species. Forest birds singing in cutblocks also may have been simply passing through (although we excluded those simply flying over) or may have been unmated males, as recorded for Ovenbirds in other suboptimal habitats (i.e., small forest tracts) in Missouri (Gibbs and Faaborg 1990, Van Horn et al. 1995). Ovenbirds nesting in boreal mixed-wood stands adjacent to cutblocks have been observed flying out into cutblocks to sing from residual trees (Lambert 1998). A rigorous nest monitoring program would be necessary to ascertain whether species recorded in cutblocks were nesting there. In a companion paper based on an artificial nest experiment (Tittler and Hannon 2000), we found no effect of logging or of residual tree retention on nest predation, but information on nesting and breeding success for each species is lacking.

Recommendations for future research

In general, we would expect to find a curvilinear relationship between forest bird abundance and residual tree retention if a full range of retention were examined. As density and basal area of trees approach those of unharvested stands, the abundance of forest birds should level off, producing an inflection point in the bird abundance vs. residual tree curve. For management purposes, it would be useful to find this inflection point. With this in mind, we initially performed logistic as well as linear regressions of bird abundance vs. residual tree density and basal area in our study. Finding that linear regressions invariably provided better fits (higher R^2 values), we have presented these and conclude that the inflection points are not within the range of residual tree retention studied. We therefore recommend that research be expanded to include retention beyond the range studied (basal area 0.50–10.65 m^2) in order to detect the inflection point. We also suggest that the range of study be expanded to include levels of retention below the level studied to address the issue of whether little residual is better than no residual tree retention.

We further recommend that future studies document the reproductive success of birds nesting in logged stands. In addition, it would be useful to return to the sites once the old stands adjacent to the cutblocks have been logged. Because the study area is designated for this second pass of logging in the year 2004, just 10 years after the initial logging, we predict that any species using cutblocks for singing, but relying on adjacent old stands for nesting, will disappear at that time. Revisiting the study at a later date might also show longer term benefits of leaving high densities of residual trees in cutblocks. One of the possible long-term benefits of residual tree retention is improved conifer regeneration (Bella and Gál 1996), which may provide habitat for forest songbird species such as the Black-throated Green Warbler, which may be dependent on the spruce component of the mixed-wood forest (Robichaud and Villard 1999). Furthermore, residual trees provide structural and age diversity to the regenerating stands, and this, in turn, may benefit forest songbirds and other forest birds over the long term (Rose and Muir 1997). Finally, considering our finding that more birds were affected when residual tree retention was examined in terms of basal area rather than number of trees per hectare, we recommend that future studies concentrate on the former measure.

Conclusions and Management Recommendations

Based on the results of this study, we recommend that managers in the boreal mixed-wood forest do not rely solely on increased tree retention in blocks to conserve forest songbirds. Managers should also maintain unharvested reserves on the landscape. We further recommend that research be continued so that firm conclusions can be drawn about exactly how much retention is advisable in cutblocks. Unharvested stands are still better for forest birds than are cutblocks with residual tree retention (within the range studied), but on a landscape level, if trees are to be harvested, retention of residual trees may be a useful method of providing habitat for some forest birds in cutblocks. Some species appear to benefit from logging in general, and to prefer low rather than high tree retention. With one exception, however, these birds are aspen-parkland species not typical of the early successional stages of the boreal mixed-wood forest; perhaps these species would be better managed in the parkland south of our study area.

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