THE EFFECTS OF THINNING ON FOREST BIRD COMMUNITIES IN DRY INTERIOR DOUGLAS-FIR FORESTS

by

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Abstract

I studied the effects of the Thompson/Nicola Mule Deer Forage and Slashing Project on vegetation structure and bird species abundance in the summers of 1990 and 1991. I sampled stand structure and bird abundance on three thinned and three unthinned study sites, each 25 ha in size. Thinning was targeted at smaller size classes, and as a result, there were significantly fewer small Douglas-fir trees (<10 cm dbh) in treatment plots. Individual tree canopy volumes were not significantly different between treatment and control sites. Canopy volumes per hectare (m^3/ha) of Douglas-fir trees <10 cm dbh were significantly lower (60% lower) in treated sites. There was no significant increase in herb and shrub cover in treatment sites. Percent cover of down and dead woody debris was significantly higher in treated sites.

This thinning trial had little effect on the forest bird community. Poor understory response, either because of the effects of cattle grazing, insufficient thinning, leaving slash on site, or an insufficient amount of time since treatment, or all four, may explain the failure of ground-and shrub-feeding bird species to increase as predicted. Elevated levels of spruce budworm and Douglas-fir tussock moth, and potential foraging habitat in other forest strata may have prevented the predicted reduction in abundance of foliage-feeding species. The increased amount

of down and dead wood likely accounted for the modest increase in woodpecker use of thinned sites.

Chi-square and discriminant function analysis suggested that several within-site bird/habitat associations exist. Northern flickers, Vesper sparrows and Chipping sparrows were associated with open forest habitats in control sites. Yellow-rumped warblers, and Dusky flycatchers were associated with dense, unthinned habitat in treatment sites. Ruby-crowned kinglets, Orange-crowned warblers and Dusky flycatchers were associated with riparian habitats in control sites.

I recommend that grazing regimes be modified to assess the effect of grazing on vegetation response. Slash could be piled to benefit ground foraging/nesting species instead of allowing it to lie where it was felled. Even if not required for snag management, unthinned patches should continue to be left as part of the treatment to maintain spatial habitat heterogeneity. Long-term monitoring of both vegetation and bird communities is recommended particularly if the Thompson/Nicola Mule Deer Forage and Slashing Project becomes a model for thinning projects in these forest types.

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1. Introduction

Thinning of overstocked stands is common in many areas of British Columbia. The primary silvicultural objective of thinning is to increase yield of merchantable timber volume (Smith 1986). Thinning reduces competition from neighbouring trees and allows greater uptake of water and nutrients from the roots. The subsequent increase in the foliage volume, and carbohydrate production allows for crown and root extension which ultimately lead to increases in diameter growth (Mann and Lohrey 1972, Smith 1986). The temporary reduction of gross total productivity due to the initial reduction of stand density is then offset by increases in diameter growth.

Thinning has potentially important consequences for wildlife because of the changes in habitat structure that it can induce. It is widely acknowledged that habitat structure can affect the composition of bird communities (MacArthur and MacArthur 1961, Anderson and Shugart 1974, Balda 1975, Maurer et al. 1981, Morgan et al. 1989). Alterations in habitat structure that leads to increases in habitat complexity usually result in increases in bird species diversity (MacArthur and MacArthur 1961, Karr 1961, Meslow 1978, Morrison and Meslow 1983). Conversely, changes that lead to the elimination or reduction of specific habitat components, such as shrubs or standing dead trees, can lead to the reduction in abundance and potential extirpation of some species (Kilgore 1971, Pryah and Jorgensen 1974, Bunnell and Alleye-Chan 1984, Raphael and White 1984, Petit et al. 1985). The response of any bird community to habitat alteration will vary among species in accordance with the magnitude of the structural alteration (Szaro and Balda 1979, Freedman et al. 1981, Medin 1985).

The structural changes that thinning can produce have been used as tools to both enhance and discourage wildlife. Initially, thinning was used as a means to produce browse for ungulates (Bunnell and Eastman 1976, Rochelle and Bunnell 1978, Whitmer *et al.* 1985; Severson and Uresk 1988) and as a tool to reduce damage to crop trees by forest pests (Sullivan and Moser 1986; Sullivan and Sullivan 1988; George and Young 1989; Murua and Rodriguez 1989). Recently, the changes in vegetation structure that thinning can produce, especially those regarding the enhancement of a suppressed understory, have been seen as a means to manipulate, or affect biological diversity (Hunter 1990).

The effects of thinning on forest bird communities are largely dependent on the specific silvicultural prescription. Increased light reaching the forest floor and decreased competition for water from heavy thinning can lead to an increase in understory growth (McDonnell and Smith 1965, 1970, Agee and Biswell 1970, Halls 1973, Dien and Zeveloff 1980, Doerr and Sandburg 1986, Severson and Uresk 1988, Hilt et al., 1989), increased fruit yields from understory shrubs (Zeedyk and Evans 1975), and increased seed production from grasses and legumes (Mann and Lohrey 1972).

Increases in understory vegetation that occur as a result of thinning have been associated with increases in the numbers of ground and shrub nesting/foraging birds (Hagar 1990, Stribling et al. 1990, DeGraff et al. 1992). Heavy thinning also can alter the distribution patterns and behaviour of some species because of increased understory vegetation (Zwank et al. 1988) or decreased canopy cover (Rolstad and Wegge 1989). More moderate thinning on the other hand, creates smaller canopy gaps that can close before having any significant affect on understory growth, bird species abundance and diversity (Welsh et al. 1992), or bird behaviour (Rolstad 1989).

Thinning can also potentially limit both nesting and foraging opportunities for some forest bird species. Any prescription that removes dead standing trees (snags) or dying trees, may limit the ability of that stand to support cavity-nesting birds (Welsh et al. 1992). Initial decreases in stand density and hence canopy volume, can also affect the density of some bird species (Morse 1967, Balda 1969, Franzreb and Ohmart 1978). The eventual increased growth in the width and length of tree crowns, however, can create a greater feeding area for crown-foraging, insectivorous birds (Dien and Zeveloff 1980).

In this study, I examined the effects of the Thompson/Nicola Mule Deer Forage and Slashing Project on the forest bird community in dry interior Douglas-fir (scientific names for all plant species appear in Appendix I) forests.

This thinning project had three main objectives (Kurta 1991):

1) to provide habitat diversity for wildlife,
2) to improve the snow interception ability of overstory trees as larger crowns develop, and
3) to improve understory growth.

To meet these objectives, the Ministry of Environment Lands and Parks (Kamloops) sponsored a large-scale thinning project. The projected stocking densities for these thinning trials were 200-600 stems/ha lower than British Columbia Ministry of Forests standards (Lloyd et al. 1990). Given that these sites were in the hot, dry interior Douglas-fir biogeoclimatic sub-zones (IDF xh1 and IDF xh2; Lloyd et al. 1990), and that a large portion of the Douglas-firs in the lower forest strata was removed, it was predicted that the structure of these stands would change in at least four ways. First, herb and shrub cover would increase. Second, individual tree canopy volume would increase. Third, canopy volume on a per hectare basis would decrease. Fourth, down and dead woody material would increase.

In light of the potential consequences of thinning on the plant community, three predictions were made regarding the impact of this treatment on the forest bird community.

1) An increase in the percent cover of shrubs and herbs, would lead to an increase in the abundance of ground/shrub-foraging birds.
2) Removal of a portion of the overstory would limit potential foraging habitat and produce a decrease in abundance of foliage-foraging birds.
3) Because thinned material was left on site, woodpecker use of thinned areas for foraging would increase.

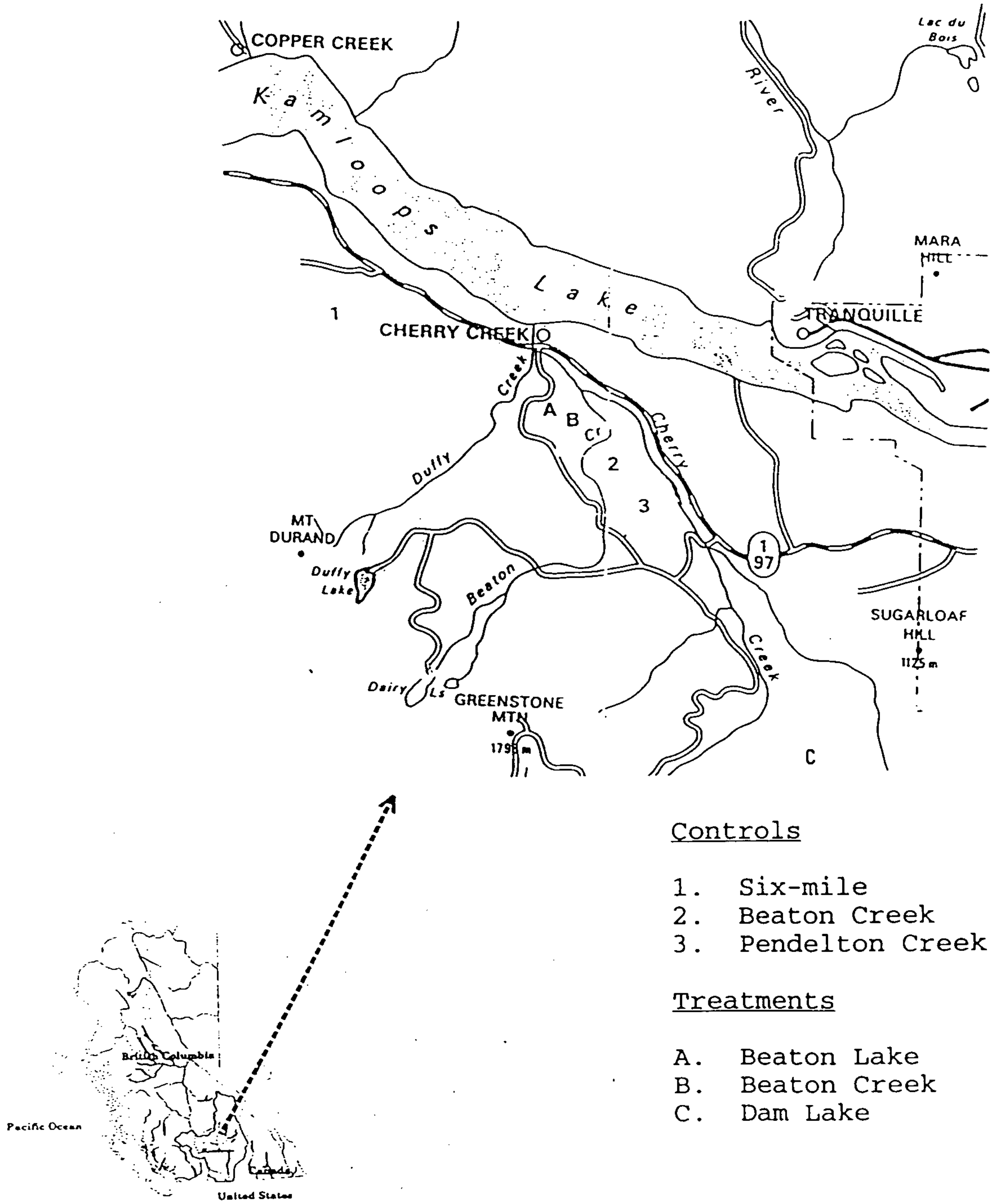
My approach to evaluating how the predicted changes in forest structure affected the forest bird community had three steps. First, I examined community structure in thinned and unthinned areas. Second, I examined how the relative abundance of individual species and three species groups (ground/shrub-feeding species, foliage-feeding species, and woodpeckers) were affected by thinning. Third, because the treatment altered the quantity and spatial distribution of some habitat types, I examined the within-site associations of individual bird species with different spatial distributions of habitat types.

2. Study sites

The Thompson/Nicola Mule Deer Forage and Slashing project was conducted in IDF xh1 and IDF xh2. These two forest types exist as a band along a elevational gradient, from 400-1200 m, in the Thompson, and Coldwater River valleys and in the Okanagan valley in the Kamloops Forest Region. Together they encompass 604,741 ha or 7.5% of the Kamloops Forest Region of B.C. (Lloyd et al. 1990).

I chose six study sites that were south and east of Cherry Creek, B. C. (Figure 1). There were three study sites in thinned areas and three study sites in unthinned, control areas. Each study site was 25 ha in size. Each of my six sites was classified as being of poor to low forest productivity. Thinning occurred three years prior to 1990. All stands had been harvested over the past several decades using a variety of methods (tie, cordwood, portable mill, and/or high diameter limit). The most recent harvesting activity took place prior to the 1970's (Kurta 1991). Firewood cutting is permitted as long as stems are less than 30 cm dbh. Flagrant contraventions of these guidelines occurred during both of my field seasons.

Douglas-fir, and Ponderosa pine were the most abundant tree species in the overstory in these stands. Both species also exist in the understory. Douglas-fir was more common (stems/ha) than Ponderosa pine in each stratum. There were a variety of both coniferous and deciduous shrubs including Saskatoon berry, spirea, common juniper,



Controls

1. Six-mile
2. Beaton Creek
3. Pendelton Creek

Treatments

A. Beaton Lake
B. Beaton Creek
C. Dam Lake

Figure 1. Location of study sites

rocky mountain juniper, rose, kinnikinik, sheperdia, and snowberry. The forb layer was made up of both grasses and herbs. Grasses included: pinegrass, bluebunch wheatgrass, rough fescue, and Kentucky bluegrass. Common herbs were: lemonweed, pussy toes, yarrow, fireweed, balsamroot, heart-leafed arnica, and strawberry. In riparian areas, trembling aspen, red alder, and paper birch were the most common trees. Red-oiser dogwood, Douglas maple, willow, and were common understory shrubs. Common herbs were twinflower, star Solomon's seal, northern bedstraw, and horsetail.

Continuous fire suppression had dramatically altered the forest structure in these stands. A natural fire return interval of approximately seven years (Dave Low, senior biologist, B.C. Ministry of Environment, Lands and Parks, Kamloops Region, pers. com.) once formerly maintained these stands in an open parkland structure. Without frequent ground fires to maintain this open parkland structure, Douglas-fir had re-seeded in the understory and now occurs in very dense patches of trees 2-5 m tall and 2.5-15 cm diameter of outside bark measured at 1.3 m above ground (dbh). These patches are juxtaposed with openings from past logging operations (e.g., old roads, skid trails and landings), small meadows, and open forests that did not re-seed to dense Douglas-fir.

The goal of the Thompson/Nicola Mule Deer Forage and Slashing Project was to achieve an average stocking density of 450 live stems/ha. This prescription required the thinning contractor to save Douglas-fir trees with full, healthy canopies and remove competing stems within 1 to 4 m. Ponderosa pines of all sizes were retained wherever possible (Kurta 1991). The degree of thinning on each site varied from 172 to 1100 trees per ha. Variability in stocking density was related to topographic position. North-facing or wetter slopes, representing better potential growing sites, were thinned to between 400-600 stems/ha. Ridge tops and south-facing or hotter, dryer slopes were thinned to between 250-300 stems/ha. Any stems > 15 cm dbh were not felled unless they were very poor quality. Snags greater than 50 cm dbh were retained, provided that workers did not fall trees within one tree height of the overstory trees plus the height of the material that was being thinned. Snags, therefore, occur in pockets of unthinned areas.

3. Effects of thinning on the plant community

The Thompson/Nicola Mule Deer Forage and Thinning Project was designed to increase habitat diversity by promoting understory growth and to improve the snow interception ability of overstory trees. This study assessed the efficacy of this thinning trial to produce desired changes. It was predicted that herb and shrub production would increase, individual tree canopy volume would increase, canopy volume on a per hectare basis would decrease, and down and dead woody material would increase.

Methods

Vegetation plots

Vegetation structure was sampled at 20 systematically located plots in each of the six study sites in July and August of 1991. Despite the thinning treatment varying according to slope position, vegetation sampling was not stratified by slope position for three reasons. First, ridge tops were often very small in size making it difficult to sample them effectively. Second, it was often difficult to determine where ridge tops began and where north or south slopes ended. Third, ridge tops, north slopes and south slopes were not present in all study sites.

Variable-radius plots were used to sample trees and snags greater than 10 cm dbh. I used a wide-field relaskope with a basal-area factor of 1. Variable radius plots were chosen because they are often more time efficient than fixed area plots, and are effective in the sampling of large trees, especially when these occur at low densities. Species, dbh, and height were recorded for each tree that was deemed "in" the plot. All trees greater than 2.5 cm dbh and less than 10 cm dbh were tallied according to species and status (live or dead) within a 10-m (0.03 ha), fixed radius plot.

Percent cover of tree, shrub, herb, grass, down and dead woody material, bare ground, litter (primarily pine needles), and moss plus lichen were estimated in the same 10-m radius (0.03 ha) fixed area plot as in Walmsley et al. (1980). Canopy cover above 4 m from the ground was estimated using a 'moose horn' (Bunnell and Vales 1989). In each vegetation plot, canopy cover was measured 17 times at 2.5-m intervals along two 20-m long, perpendicular transects that intersected at the centre of the plot. The 17 canopy cover measurements were then averaged to determine an overall estimate of canopy cover. The initial transect direction was a compass bearing generated from a random number table.

Canopy volume

Canopy volume measurements were taken after vegetation sampling was completed and were recorded as in Sturnam (1968). All volume measurements were recorded on trees encountered in earlier vegetation surveys. The numbers of trees for each species and in each diameter class were sampled proportional to occurrence. Canopy volume of trees < 10 cm dbh were recorded on trees tallied in fixed area plots. I sampled up to 60 trees both greater than and less than 10 cm dbh at each study site (Table 1). Of the trees ≥ 10 cm dbh that I previously sampled, I randomly selected trees from each diameter class. Two Douglas-fir trees of diameter class 1 (10-20 cm dbh) and one from diameter class 2 (21-30 cm dbh) were sampled at each vegetation plot until quotas were filled. Trees in the larger diameter classes were sampled opportunistically as they occurred in each vegetation plot until quotas were filled.

For Douglas-fir trees less than 10 cm in diameter, three trees were selected within a 10-m radius from each plot center. I sampled the closest tree to a point in the plot determined by a randomly chosen compass bearing and distance from the plot center. In plots where less than 3 trees were encountered, I sampled more than 3 trees on denser plots to fill quotas. Small Ponderosa pine trees were sampled opportunistically as they occurred in each vegetation plot until quotas were filled.

Canopy volumes, expressed as cubic meters of foliage, were calculated for each tree species and diameter class. These volumes were then multiplied by the appropriate stocking density to yield canopy volume on a per hectare basis.

Table 1. Number of trees sampled for canopy volume measurements in each control and treatment plot by species and diameter class.

Tree species	Diameter class		Number sampled
Douglas-fir	0	(2.5-9.9 cm)	60
	1	(10-20 cm)	35
	2	(21-30 cm)	10
	3	(31-40 cm)	3
	4	(41-50 cm)	3
	5	(>50 cm)	3
Ponderosa pine	0	(2.5-9.9 cm)	3
	1	(10-20 cm)	3
	2	(21-30 cm)	3
	3	(31-40 cm)	3
	4	(41-50 cm)	3
	5	(>50 cm)	3

Data analysis

To examine the differences in stocking densities between treatments and controls of large (≥10 cm) live trees and snags for each dbh class and tree species (Douglas-fir and Ponderosa pine), I used a one-way analysis of variance in which vegetation plots were nested in condition (treatment and control; ANOVA table for vegetation analysis is presented in Appendix II). The same nested one-way analysis of variance was used to test for differences between treatments and controls on the stocking density of small (<10 cm dbh) live and dead Douglas-fir and Ponderosa pine, the total stocking densities for all small trees and small conifers, and the arcsine transformed cover data. Following Sokal and Rolf (1981), main effects were interpreted despite occasions where significant nested effects were detected. A one-way analysis of variance was used to test for differences between treatments and controls on the canopy volume of individual trees and the total canopy volume/ha for each dbh class and tree species (Douglas-fir and Ponderosa pine).

Results

Stocking density: Large trees

Densities of live trees (stems/ha) by species and diameter classes ≥10 cm dbh are summarized in Table 2. The densities of each species and in each diameter class did not differ significantly between treatment and controls

Table 2. Mean densities of live trees (stems/ha) ≥ 10 cm dbh in treatment and control sites by tree species and diameter class.

Species	DBH class		Control	Treatment	p-value
Douglas	1	(10-20 cm)	198.1	171.8	>0.25
-fir	2	(21-30 cm)	51.6	64.2	>0.25
	3	(31-40 cm)	17.4	16.6	>0.25
	4	(41-50 cm)	5.2	4.9	>0.25
	5	(>50 cm)	2.8	2.8	>0.25
Ponderosa	1	(10-20 cm)	13.7	6.6	>0.25
pine	2	(21-30 cm)	5.9	5.3	>0.25
	3	(31-40 cm)	3.5	3.7	>0.25
	4	(41-50 cm)	1.8	0.7	0.25<p<0.1
	5	(>50 cm)	0.6	1.0	>0.25

(α=0.05). The history of selective logging has resulted in a diameter distribution that favours younger/smaller trees. For both Douglas-fir and Ponderosa pine, younger/smaller trees were the most common age/size class. Stocking density decreased with increasing diameter class in both treatments and controls. Douglas-fir was the most common overstory tree species in treatment and control sites, and comprised 93% and 83% of overstory stem density in treatment and control sites, respectively.

Densities of snags by species and diameter classes are summarized in Table 3. Densities of each species and in each diameter class did not differ significantly between treatments and controls. Smaller Douglas-fir snags were the most common size class. Stocking density of snags decreased with increasing diameter class in both treatments and controls. Ponderosa pine snags in the smallest size classes were absent in both treatments and controls. The extremely low densities of larger diameter classes of snags reflects the logging history of these stands. The removal of larger live trees reduced recruitment of large live trees into the snag population. In addition, firewood cutting generally targets larger diameter snags, further skewing the diameter distribution in favour of smaller snags. Douglas-fir was the most common snag species in treatment and control sites, and comprised 95% of snags in both treatment and control sites.

Table 3. Mean densities of snags (stems/ha) ≥ 10 cm dbh in treatment and control sites by tree species and diameter class.

Species	DBH class		Control	Treatment	p-value
Douglas	1	(10-20 cm)	22.4	23.3	>0.25
-fir	2	(21-30 cm)	4.1	3.1	>0.25
	3	(31-40 cm)	1.5	1.2	>0.25
	4	(41-50 cm)	0.7	1.2	>0.25
	5	(>50 cm)	1.0	1.2	>0.25
Ponderosa	1	(10-20 cm)	0.0	0.0	n/a
pine	2	(21-30 cm)	0.3	0.8	>0.25
	3	(31-40 cm)	0.1	0.1	>0.25
	4	(41-50 cm)	0.7	0.3	0.25<p<0.1
	5	(>50 cm)	0.5	0.4	>0.25

Stocking density: Small trees

In both treatments and controls Douglas-fir was the common small tree (<10 cm dbh) in the understory (Table 4). Douglas-fir represented 98% and 99% of the understory stems in treatments and controls, respectively. In both treatment and control sites a substantial portion of the Douglas-fir were dead (21% and 30%, respectively). The density of live Douglas-fir was significantly higher in controls than in treatments ($0.025 < p < 0.01$), while the density of dead Douglas-fir did not differ significantly between treatments and controls. Together, there were significantly more Douglas-fir (both live and dead) in control plots than in treatments ($0.025 < p < 0.01$). The densities of live and dead Ponderosa pines and 'other' (rocky mountain juniper, birch, trembling aspen) species did not differ significantly between controls and treatments. There were significantly more conifers ($0.025 < p < 0.01$), and small trees of all species in control sites than in treatment sites ($0.025 < p < 0.01$). The large standard errors associated with estimates of the densities of Douglas-fir in both control and treatment sites were due to the extreme patchiness (variability) of individual, 0.03-ha vegetation plots. Estimates of density of Douglas-fir based on individual 0.03-ha vegetation plots ranged from 0 to 6000 trees/ha.

Table 4. Stocking density (trees per ha) of small trees (<10 cm dbh) from fixed area plots in controls and treatments.

Stocking variable	Control	Treatment	p-value
Total Douglas-fir dead	534.2	399.5	>0.25
Total Douglas-fir live	887.0	106.7	$0.025<p<0.01$
Total Douglas-fir	1421.2	506.1	$0.05<p<0.025$
Total Ponderosa pine dead	5.9	1.1	$0.1<p<.0.05$
Total Ponderosa pine live	20.2	1.1	$0.1<p<.0.05$
Total Ponderosa pine	26.1	2.1	$0.1<p<.0.05$
Total other dead	2.3	4.8	>0.25
Total other live	31.5	10.6	>0.25
Total other[a]	33.8	15.4	>0.25
Total small	1478.7	523.6	$0.05<p<0.025$
Total conifer	1447.2	508.2	$0.05<p<0.025$
Proportion of Douglas-fir dead	0.30	0.21	

[a] = Other includes trembling aspen, paper birch, and Rocky mountain juniper.

Canopy volume: Large trees

There was no significant difference of individual tree canopy volumes for either Douglas-fir or Ponderosa pine between controls and treatments for each diameter class (Table 5). On a per hectare basis, canopy volumes were significantly higher in controls than treatments for Ponderosa pines 31-40 cm dbh ($0.025 < p < 0.01$). Conversely, canopy volumes/ha were significantly higher in treatments than in controls for Ponderosa pines >50 cm dbh ($0.05 < p < 0.025$; Table 6).

Canopy volume: Small trees

Individual tree canopy volumes for both Douglas-fir and Ponderosa pine were not significantly higher in treatment sites than in controls as was predicted (Table 7). Because of the significantly higher density of live Douglas-fir in control plots (Table 4), the total canopy volumes on a per hectare basis were significantly higher for Douglas-fir in control plots ($0.01 < p < 0.005$; Table 8). The total canopy volume of Ponderosa pine did not differ significantly different between treatment and control plots.

Cover measurements

The percent cover of grass was not significantly greater in treatment sites than in control sites (Table 9). Herb and low shrubs (< 1 m) cover also showed no response

Table 5. Mean canopy volume (m^3) of individual Douglas-fir and Ponderosa pine trees $\geq$ 10 cm dbh by diameter class in control and treatment sites.

Species	dbh class		Control	Treatment	p-value
Douglas	1	(10-20 cm)	8.1	5.7	$0.25<p<0.1$
-fir	2	(21-30 cm)	26.9	15.1	$0.25<p<0.1$
	3	(31-40 cm)	61.5	43.1	>0.25
	4	(41-50 cm)	95.5	80.3	>0.25
	5	(>50 cm)	120.6	108.1	>0.25
Ponderosa	1	(10-20 cm)	9.6	7.8	>0.25
pine	2	(21-30 cm)	30.6	18.2	>0.25
	3	(31-40 cm)	50.4	37.5	$0.1<p<.0.05$
	4	(41-50 cm)	61.5	100.1	>0.25
	5	(>50 cm)	144.6	184.5	$0.25<p<0.1$

Table 6. Total mean canopy volume (m^3/ha) for Douglas-fir and Ponderosa pine trees ≥ 10 cm dbh in control and treatment sites.

Species	dbh class		Control	Treatment	p-value
Douglas-fir	1	(10-20 cm)	1623.5	940.2	0.25<p<0.1
	2	(21-30 cm)	1398.6	976.5	0.25<p<0.1
	3	(31-40 cm)	1079.4	749.2	0.25<p<0.1
	4	(41-50 cm)	493.5	420.5	>0.25
	5	(>50 cm)	389.0	399.8	>0.25
Ponderosa pine	1	(10-20 cm)	171.3	54.1	0.25<p<0.1
	2	(21-30 cm)	190.6	101.3	>0.25
	3	(31-40 cm)	175.5	146.4	0.025<p<0.01
	4	(41-50 cm)	116.7	58.1	>0.25
	5	(>50 cm)	93.1	259.1	0.05<p<0.025

Table 7. Mean individual tree canopy volume (m^3) of trees $\leq$10 cm dbh in control and treatment sites.

Species	Control	Treatment	p-value
Douglas-fir	1.6	2.2	0.25<p<0.1
Ponderosa pine	2.3	3.3	>0.25

Table 8. Total mean canopy volumes of trees <10 cm dbh (m^3/ha) in control and treatment sites.

Species	Control	Treatment	p-value
Douglas-fir	1332.4	227.4	0.01<p<0.005
Ponderosa pine	66.2	11.3	>0.25

Table 9. Percent cover estimates on treatment and control sites.

Cover variable	Control	Treatment	p-value
Down and dead	12.6	23.0	0.01<p<0.005
Litter	11.3	9.4	>0.25
Grass	12.5	14.8	>0.25
Bare ground	12.9	10.5	0.25<p<0.1
Tree	5.5	3.4	0.01<p<0.005
Shrub <1 m	9.8	5.3	0.25<p<0.1
Shrub 1-4 m	2.7	0.10	0.05<p<0.025
Herbs	12.0	10.7	>0.25
Moss & lichen	22.8	22.7	>0.25
Canopy cover	38.3	25.9	>0.25

to the treatment. Percent cover tall shrubs (1-4 m), however, was significantly greater in control sites ($0.05 < p < 0.025$).

Because thinned material was left on site, the percent cover of down and dead was significantly higher in treatment sites than in control sites ($0.01 < p < 0.005$). Due to the higher stocking density of trees in control sites, the physical space that tree trunks occupied was significantly higher ($0.01 < p < 0.005$) in controls than in treatment sites. The comparatively low estimates of canopy closure recorded illustrates the patchiness of the overstory in both treatments and controls. In addition, where stocking density was high, a large proportion of the canopy was dead (Table 4), further contributing to low canopy cover estimates.

Discussion

The main goal of this thinning project was to alter forest structure for the benefit of wildlife. Thinning was designed specifically to promote the growth of what was believed to be a previously suppressed understory plant community and to improve the snow interception ability of the overstory trees (Kurta 1991). Results from vegetation sampling indicated that two of the four initial predictions regarding the effects of thinning occurred as expected. As was initially predicted, thinning resulted in a significant increase in the amount of down and dead woody material and a significant decrease in the canopy volume per hectare of Douglas-fir trees. In contrast, contrary to predictions, ground and shrub cover did not increase significantly, nor did the canopy volume of individual trees in treated stands. Several factors could have contributed to the negligible response of these two habitat variables after thinning.

Ground cover

Cattle grazing may have affected ground cover. Cattle grazing can change the structure of grassland vegetation by negatively affecting plant vigour, growth, and productivity (Stoddart et al. 1975) or by reducing vegetation density (Kosco and Bartolome 1983) thereby altering species composition of plant communities (Ryder 1980). Cattle trampling can also disturb the cryptogamic layer of grassland communities. This layer of lichens, bryophytes, and cyanophytes stabilizes the soil (Anderson and Rushforth 1982), enhances soil water retention (Brotherson and Rushworth 1983), and increases soil nitrogen content via fixation by cyanobacteria (Cameron and Fuller 1960). Trampling can reduce the extent of this layer which can lead to further alteration of the plant community.

Cattle have been grazed in the IDF xh1 and xh2 stands in the Kamloops area since the late 1800's (Dave Low, pers com.). During my two field seasons cattle were brought on to my study sites early in May. In 1990, they remained until late June, after which they were moved to higher elevations. They were then returned to my study sites in late August. In 1991, cattle remained on two sites (1 treatment and 1 control) throughout May to September. Having cattle remain in these lower elevation stands from May to September is unusual. On the treatment site that was subject to cattle grazing from May to September, there have been ongoing problems with poorly maintained fences which allowed cattle to return after they had been moved out (Mike Deedle, range technician, Kamloops Forest Service). On my four other sites cattle were rotated in a similar fashion to that in 1990. Cattle have been grazed on these sites in a similar manner for the last several decades. Their influence in these stands prior to, and since the time of thinning may have prevented the understory from responding as predicted.

Although cattle grazing may have been a factor other explanations could account for the lack of understory response. Leaving slash on site may have represented a physical barrier that prevented ground cover from responding as it might have if slash had been removed.

Insufficient time may have passed for the effects of thinning on understory to manifest themselves in an appreciable way. Most studies examining the effect of thinning on understory development were performed anywhere from 7 to 22 years after thinning (Agee and Biswell 1970, McDonnel and Smith 1970, Greleh et al. 1972, Sassaman et al. 1972, Doerr and Sandburg 1986, Alaback and Herman 1988, Hilt and Sonderman 1989, Hagar 1990). In each of these other studies, significant increases in the amount of understory vegetation were recorded at the time of examination. Studies that measured vegetation response shortly after thinning, recorded significant increases in understory vegetation after 1-2 years (Zwank et al. 1988, Stribling et al. 1990), 3 years (McDonnell and Smith 1965), 4 years (Barrett 1970, Austin and Urness 1982) and 5 years (Crouch 1986). Crouch (1986) also noted annual gains of total plant production on heavily thinned plots in the years immediately after thinning.

Research elsewhere has shown that three years was potentially long enough to expect a significant response in the production of the understory plant community. Most of the results reported above, however, were from moister forest types (e.g., Austin and Urness 1982, Crouch 1986, Zwank et al. 1988, Stribling et al. 1990). Increased herb and shrub cover after thinning in similar forest types occurred in the absence of grazing (McDonnell and Smith 1965, Barrett 1970). The grazing received by these stands over the last 50+ years may have prevented the understory vegetation from responding as predicted even if cattle had been excluded from these sites after thinning. Significant increases in understory vegetation may not occur with the continued presence of cattle. Previous research has shown that in similarly dry areas appreciable changes in understory vegetation may take a long time to occur even when cattle have been excluded (McLean and Tisdale 1972).

Finally, the extent of the thinning may not have been severe enough to generate a significant change in the percent cover of ground cover. However, this thinning trial did significantly reduce stocking density of smaller diameter classes of Douglas-firs, and did result in up to a 60% reduction of canopy volume/ha (Figure 2). Given the extent of the reduction of canopy volume/ha on these xeric sites, it is not unreasonable to expect an increase in the percent cover of ground vegetation.

Tree canopy volumes

There are two potential reasons why canopy volumes of individual trees did not respond as expected. First, release of trees after thinning is primarily governed by tree vigour (Barrett 1969, Helms and Standiford 1985). Any

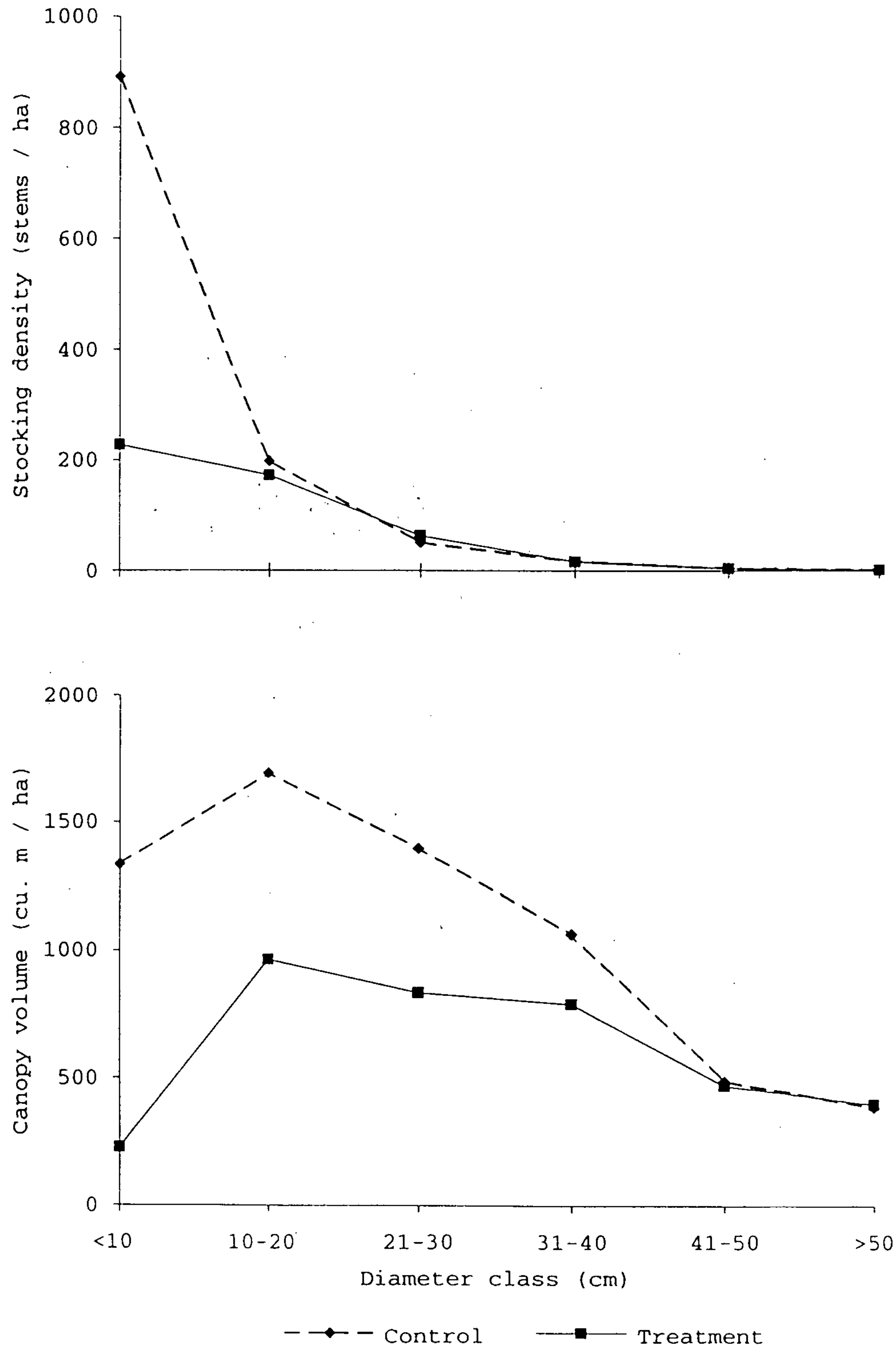


Figure 2. Comparison of stocking density and canopy volume / ha of Douglas-firs in control and treatment sites

factors that negatively affect tree vigour can delay potential benefits of thinning. At the time of study, Douglas-fir tussock moth (*Orgyia pseudotsugata*) and Western Spruce budworm (*Choristoneura fumiferana*) were at near epidemic levels in these stands(Lorraine Maclauchlan, Forest Entomologist, Kamloops Forest Service, pers. comm.). Given that these two forest insect pests can have negative affects on Douglas-fir vigour (Carlson et al. 1982, Schmidt et al. 1983), it is possible that their influence prevented the expected increase in canopy volume of individual trees.

Second as with ground cover, insufficient time may have passed for these stands to respond in the predicted direction. Previous studies have shown that it can take anywhere from 2-5 years to detect a marked increase of diameter growth and hence tree canopy growth after thinning (Herring and Ethridge 1976, Herring 1977, Gordon 1973, Helms and Standiford 1985). I examined my stands only three years after thinning. This amount of time may be insufficient in such dry forest types to produce a significant increase in canopy volume. A short post-treatment period, coupled with the negative effects of spruce budworm and Douglas-fir tussock moth on Douglas-fir vigour, likely led to the lack of increase in Douglas-fir canopy volume.

Conclusions

The vegetation on the three treatment sites did not respond as predicted. It is likely that the effects of this thinning have been confounded by a number of different factors. First, the effect of cattle grazing, both past and present, may have prevented the understory vegetation of these stands from increasing as expected. Second, leaving slash on site where it was felled may have inhibited the growth of understory vegetation. Third, the thinning trial itself may have not been severe enough to produce a significant change in understory vegetation. Fourth, spruce budworm and Douglas-fir tussock moth may have hindered the response of canopy volume to thinning. Finally, time since treatment may have been insufficient to permit response of either understory vegetation and canopy volume. Results of my study indicate that thinning was unable to generate the desired and predicted outcomes quickly. Clearly, these stands need to be re-surveyed to assess the ability of thinning to generate desired increases in understory and canopy volume. It would also be advisable to restrict cattle grazing in some areas to assess its potential impact. Continued grazing of these thinned stands without resting them may not benefit the wildlife for which the treatment was intended.

The absence of fire in these stands is also of concern. Without fire to prevent Douglas-fir from regenerating, the treated stands will soon revert to their previous structure. This will lead to a repetition of the cycle of stagnation, insect infestation, and the ultimate need for additional expensive treatments. However, social, environmental and political concerns regarding the use of fire as a management tool may make intervention of this kind difficult if not impossible (Bunnell 1991). It is important, however, that the use of fire as a management tool be explored.

4. Effects of thinning on the forest bird community

Habitat structure is widely acknowledged to effect forest bird communities. It was predicted that the Thompson/Nicola Mule Deer Forage and Slashing Project would alter four aspects of habitat structure in these forest stands: 1) herb and shrub cover would increase, 2) individual tree canopy volume would increase, 3) canopy volume on a per hectare basis would decrease, and 4) down and dead woody material would increase.

I examined potential changes in the forest bird community due to this thinning project in three ways. First, I examined bird community structure in thinned and unthinned sites. Second, I examined how the relative abundance of individual species and three species groups (ground- plus shrub-feeding species, foliage-feeding species, and woodpeckers) were affected by thinning. Third, because the treatment altered the quantity and spatial distribution of some habitat types within-sites, I examined the associations of individual species with the spatial distributions of four habitat types.

Methods

Censusing

Forest birds were sampled in 3 thinned and 3 unthinned sites. Because of time constraints, only 4 sites (2 in treatments and 2 in controls) were sampled in 1990. An additional treatment and control site were added for the 1991 season. All 6 sites were 25 ha in size. Within each study site, a grid was surveyed with station markers placed at 50-m intervals. Birds were surveyed using the spot-mapping technique as outlined by Verner (1985). Eight visits were made to each site in both 1990 and 1991. In 1990, censusing began on June 1 and ended on July 1. In 1991, censusing began on May 12 and ended on June 26. In each year 2 people performed bird censusing. One observer participated during both years. Censusing began one-half hour after sunrise and did not extend past 11:00 a.m. No censuses were conducted in rain, nor when wind exceeded 20 kph. On each day, 1 treatment and 1 control were surveyed. Censusing began at a different location in each grid for each census period. Observers walked along grid lines during censusing and marked the location of singing, calling, and visually-detected birds on maps of each study site. When possible, locations of simultaneous singing males and movements of singing males were recorded. After each census, raw data were transposed on to summary sheets for each species. Clusters of detections were then used to determine the number of territories for each species at the end of each season. At least 3 detections on 3 different censusing periods were required for a cluster of points to be considered a territory. Territories were summed to the nearest 0.5 territory within each study site. For a territory to be considered a full territory, all of the detections that identified the territory had to fall within the bounds of the study site. For a territory to be considered 0.5 territory, a minimum of 2 out of 3 detections, or at least one-half of the detections had to be located within borders of the study site.

Two estimates of the number of territories for each species were derived. The minimum number of territories was a conservative estimate that did not include detection clusters of questionable status (e.g., clusters of detections where no simultaneous singing detections were recorded). The maximum number of territories included the clusters of detections which were excluded in the estimate of minimum number of territories.

Composite vegetation maps

In each study site, forest bird grids were used to map coarse habitat attributes. I walked through each grid and mapped areas of dense-unthinned coniferous forest, open coniferous forest (those areas that would not require thinning) open meadows (openings >25 m^2) and areas with deciduous trees and shrubs >2-m tall such as willow, Douglas maple, and red-oiser dogwood (henceforth these habitat types will be referred to as dense forest, open forest, open and riparian, respectively). Percent cover of each habitat type was then estimated for each 50x50-m cell.

Because of the way habitat structure changed as a result of thinning, the resolution of the habitat maps differed between treatment and controls. In untreated sites, open habitat was clearly evident as it represented an obvious juxtaposition of habitat types. This juxtaposition of open habitat with dense forest, and open forest habitats was not as clear in treatment sites; that is, the simplification of the habitat in treatment sites made the detection of small open meadows less obvious. As a result, composite vegetation maps for treatment sites were dominated by open forest, whereas control plots were dominated by dense forest habitat. In addition, open habitat was easier to detect in controls and therefore comprised a larger proportion of the area than in treatments.

Data analysis: Community structure

To identify potential differences in overall community structure, I plotted dominance-diversity curves and examined the distributions of bird species detections in treatments and controls. I also calculated two similarity indices to examine the similarity of bird communities in treatment and control plots. I used Jaccard's similarity coefficient to test the degree of similarity between treatments and controls using simple presence-absence data from species lists, and I used Morisita's index of similarity to compare the degree of similarity based on log-transformed species detection data (Krebs 1989).

<u>Data analysis</u>: Relative abundance

I used two-way analysis of variance using condition (treatment vs. control) and year (1990, 1991) as 'treatment' variables, to examine the number of bird territories. Only 11 species occurred in sufficient density (number of territories) to warrant statistical analysis. I placed several species into groups of similar foraging strategies (ground- plus shrub-feeders and foliage-feeders) to test for general effects on species groupings predicted to change as a result of thinning. The ground- plus shrub-feeding group was comprised of Vesper sparrows, Dark-eyed juncos, American robins, Chipping sparrows, and Townsend's solitares (all scientific names for bird species are presented in Appendix II). Ground- plus shrub-feeding species were predicted to increase as a result of thinning because of the anticipated increase in herb and shrub cover. The foliage-feeding group was comprised of Mountain chickadees, Western tanagers, Red-breasted nuthatches, and Yellow-rumped warblers. Foliage-feeding species were expected to decrease in abundance because of the decrease in canopy volume and thus, abundance of foraging habitat.

I summed the number of detections from the 1991 data for each of 22 species over each study site as another means to assess relative abundance. I used all types of detections (songs, calls and visual) to avoid biasing results towards those detections where birds were most conspicuous (i.e., males singing from elevated posts in open areas). I excluded from this analysis those species that were transient in their behaviour (e.g., Pine siskins, Red crossbills, Evening grosbeaks), or occurred infrequently (e.g., Red-tailed hawk, Cooper's hawk, Sharp-shinned hawk).

One reason for this later analysis was to examine the frequency of occurrence of species that eluded analysis based on territories because of large territory size, or low numbers of detections. I used one-way analysis of variance to examine the number detections of each species between treatment and control sites.

I also used the two species groups noted above plus two additional groups (woodpeckers and woodpeckers without Red-naped sapsuckers) to analyze detection data. The woodpecker group included Black-backed woodpeckers, Three-toed woodpeckers, Hairy woodpeckers, Northern flickers, Pileated woodpeckers, and Red-naped sapsuckers. I removed Red-naped sapsuckers from the woodpecker group to restrict my analysis to only those woodpecker species that feed on dead and down material. I also used a one-way analysis of variance as that used for detections of individual species.

Data analysis: Bird/habitat relationships

I sub-sampled the 50x50-m cells of the composite habitat maps in treatments and controls to examine the distribution of detections in the extremes of each habitat type (e.g. those cells with a high percentage of dense forest, open forest, open, or riparian habitat). I assumed that examining cells with high levels of each respective habitat type best addressed the potential effects of that habitat type on bird species distribution. I chose cells that had >59% coverage for dense forest, open forest and open and cells >9% for riparian. This choice generated the greatest extremes of each habitat type while retaining enough cells for analysis. The numbers of cells in each habitat are summarized in Table 10. Once cut-off criteria were established, I summed the number of detections by species for each habitat type and rank ordered the number of detections. The results were plotted as dominance-diversity curves to examine community structure in each habitat type. The shapes of these curves were examined to determine if distinct communities occur in any of the four habitat types. I used presence-absence data from species lists to calculate Jaccard's similarity coefficient in order to test the degree of similarity between communities that were present in different habitat types within treatments and controls (Krebs 1989).

I used a uni-variate and a multi-variate analysis to examine the effects of within-site vegetation variability on bird distributions. In this analysis, I summed the number of detections in each 50x50-m cell at each study site. I then used the composite vegetation maps to

Table 10. Number of cells in each habitat type

Habitat type	Condition	% coverage in each cell	# of cells	% of cells
Riparian	Control	>9	15	5
	Treatment	>9	7	2
Open	Control	>59	18	6
	Treatment	>59	6	2
Open forest	Control	>59	50	16
	Treatment	>59	271	90
Dense forest	Control	>59	126	41
	Treatment	>59	14	5

determine relationships of bird detections to cell composition. I used a Chi-square analysis to test if bird species were associated with any habitat type (dense forest, open forest, open, riparian) more than would be expected by chance. For this analysis, each cell was assigned one of five cover classes for each habitat type such that the sum of all habitat classes totaled 100% for each cell. Cover classes were: 0 = 0%, 1 = 1-25%, 2 = 26-50%, 3 = 51-75%, 4 = >75%. Because the number of cells in each habitat category was different (spatial heterogeneity), I weighted expected number of detections according to the proportion in which that habitat type occurred in each study site (e.g., if cells containing 1-25% dense cover made up 50% of the land base, then 50% of the detections should occur in that category class). If there was no disproportionate use of any habitat type, the number of detections in each habitat type would be similar to the proportion of the land base that each habitat type represented. I performed a Chi-square goodness of fit test for each species, in each of the 4 habitat types (dense forest, open forest, open, riparian). This analysis was done for treatments and controls separately.

The Chi-square analysis yielded several significant results. Most of these, however, were derived from intermediate percent cover classes (i.e., classes 1 and 2). This finding suggests that species may have been associated with cells which had a particular combination of habitat attributes. Therefore, I used a one-group direct discriminant function analysis using SPSSx (Nie 1983) in an attempt to classify cells in which bird species were detected (presence) compared to those where no detections were recorded (absence) based on the habitat types of each cell. The single discriminant function derived was evaluated by the significance of the Chi-square statistic based on Wilks' lambda. I used classification techniques to assess the ability of the function to separate groups effectively. The proportion of cases correctly classified according to actual group membership indicated the success of the between-group classification. Analysis was conducted on treatments and controls separately. Box M scores were calculated to test for the homogeneity of the variance/covariance matrix.

Results

Community structure

Dominance-diversity curves for controls and treatments were very similar (Figure 3). Both communities were dominated by Chipping sparrows, the most abundant species. There was a precipitous drop in abundance from the most abundant to the next most abundant species. After this steep drop both curves declined less steeply and plateaued after the ninth most abundant species. The composition of this group of eight species is the same in control and treatment sites, although the rank order in abundance of

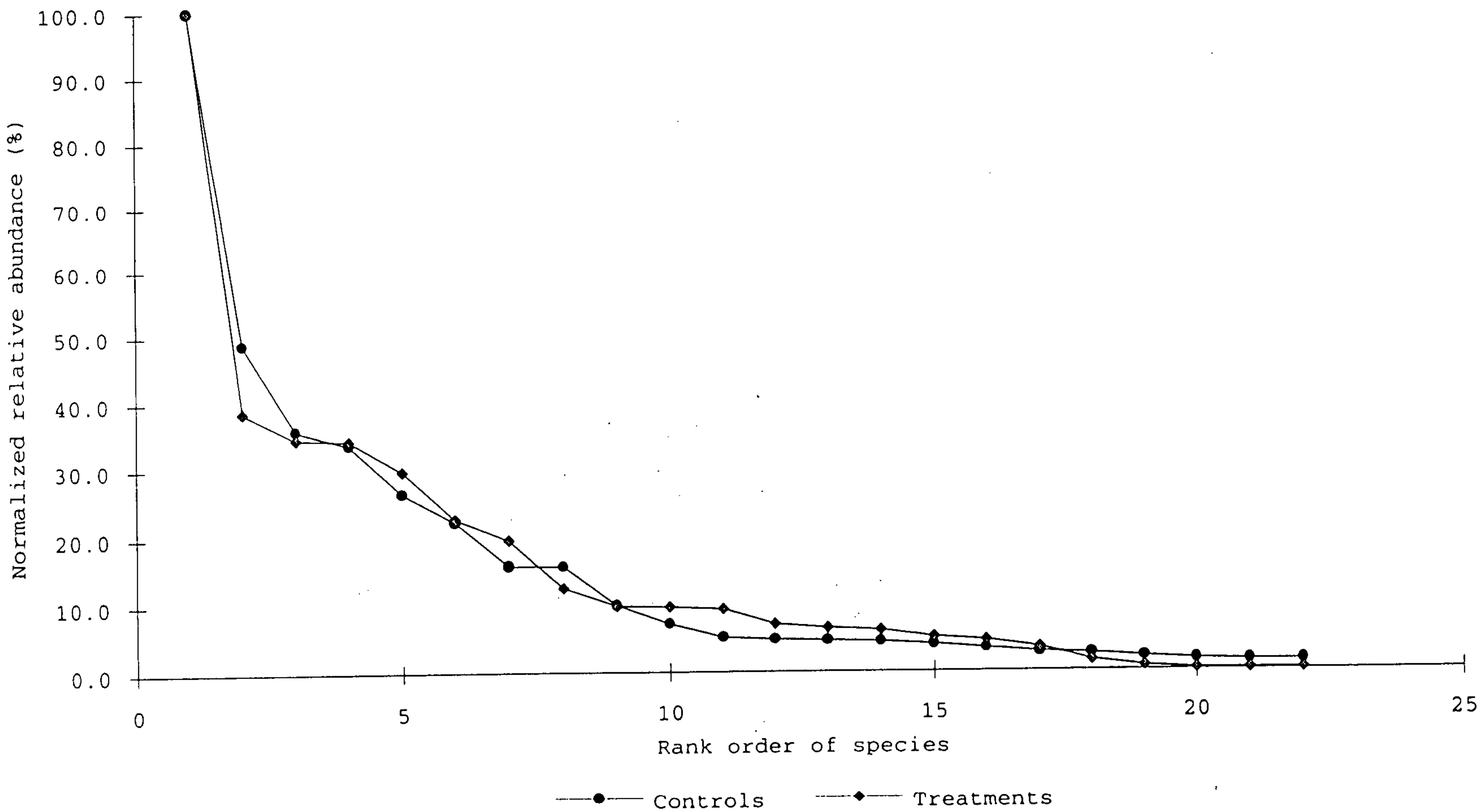


Figure 3. Dominance diversity curves for controls and treatments

these species is somewhat different (Table 11). After the ninth most abundant species, both communities are comprised of many rare species.

The two measurements of community similarity also indicate that the communities present in treatment and control plots were very similar. Using presence/absence data, Jaccard's index of similarity was 0.92. Using log-transformed detection data, Morisita's index of similarity was 0.96. These similarity indices can range from 0 (no similarity) to 1 (complete similarity).

<u>Bird abundance</u>: Number of territories

Of the more than 50 species recorded (Appendix III) only 11 species occurred in high enough densities (i.e., had small enough territories) and occurred frequently enough to test for differences between treatment and control plots. There were no significant differences in the number of territories between the two years with the following exceptions. In 1991 there were significantly more Mountain chickadees (maximum density estimate), Chipping sparrows (minimum and maximum density estimate) and Vesper sparrows (minimum density estimate) than in 1990 (Table 12). The differences in abundance between years for these species was likely because of the early start of censusing in 1991.

There were no significant differences in the number of territories for either maximum or minimum density estimates

Table 11. Rank order of the number of detections in both treatments and controls

Rank Order	Control		Treatment	
	Species	Number of detections	Species	Number of detections
1	Chipping sparrow	675	Chipping sparrow	653
2	Red-breasted nuthatch	329	Mountain chickadee	252
3	Dark-eyed junco	242	Yellow-rumped warbler	226
4	Yellow-rumped warbler	227	Dark-eyed junco	224
5	Mountain chickadee	178	Red-breasted nuthatch	194
6	American robin	150	American robin	148
7	Dusky flycatcher	107	Townsend's solitare	128
8	Townsend's solitare	106	Vesper sparrow	82
9	Vesper sparrow	67	Dusky flycatcher	64
10	Red-naped sapsucker	48	Red-naped sapsucker	61
11	Western tanager	34	Hairy woodpecker	46
12	Hairy woodpecker	32	Northern flicker	42
13	Northern flicker	31	Black-backed woodpecker	40
14	Solitary vireo	27	Solitary vireo	33
15	Orange-crowned warbler	23	Pileated woodpecker	30
16	Black-backed woodpecker	19	Three-toed woodpecker	23
17	Wilson's warbler	17	Western tanager	10
18	Ruby-crowned kinglet	14	Orange-crowned warbler	4
19	Three-toed woodpecker	11	Ruby-crowned kinglet	1
20	Pileated woodpecker	10	Swainson's thrush	0
21	Swainson's thrush	10	Warbling vireo	0
22	Warbling vireo	3	Wilson's warbler	0

Table 12. Mean number of territories for minimum and maximum density estimates in controls and treatments in 1990 and 1991 by bird species.

Species	Minimum density			Maximum density		
	1990	1991	p-value	1990	1991	p-value
American robin	2.6	2.6	>0.25	3.0	3.4	p>0.25
Chipping sparrow	12.1	18.8	0.05<p<0.025	13.9	21.2	0.05<p<0.025
Dark-eyed junco	7.9	7.4	>0.25	9.8	9.3	>0.25
Dusky flycatcher	2.6	2.8	0.25<p<0.1	3.9	4.2	p>0.25
Mountain chickadee	2.6	3.8	0.1<p<0.05	3.4	4.7	0.05<p<0.025
Red-breasted nuthatch	3.3	4.3	>0.25	4.3	5.2	>0.25
Solitary vireo	1.5	1.2	>0.25	2.3	1.8	>0.25
Townsend's solitare	2.5	2.7	>0.25	2.8	3.1	>0.25
Vesper sparrow	0.9	2.8	0.05<p<0.025	1.5	3.3	0.1<p<.0.05
Western tanager	1.4	1.2	>0.25	1.6	1.3	>0.25
Yellow-rumped warbler	9.1	7.1	>0.25	10.6	8.5	>0.25

between thinned and unthinned sites for any of the 11 species examined (Table 13). Furthermore, there was little difference in the number of territories in treatment or control sites of the species that were expected to be most affected by this treatment. Densities for the ground-feeding-species, and for the entire ground-feeding group, were almost identical in treatment and control sites (Table 14). Similarly, there was no significant difference in the number of territories of the foliage-feeding group between control and treatment sites (Table 15).

When all species were considered together there no significant difference between the number of territories in control sites than in treated sites for either density estimate ($\bar{x}_{\text{minimum number of territories control}} = 58.5$, $\bar{x}_{\text{minimum number of territories treatment}} = 52.3$, $p > 0.25$; $\bar{x}_{\text{maximum number of territories control}} = 70.5$, $\bar{x}_{\text{maximum number of territories treatment}} = 68.4$, $p > 0.25$). There was also no significant differences in the total number of territories of the 11 most abundant species between control and treatment sites for either density estimate ($\bar{x}_{\text{minimum number of territories control}} = 51.8$, $\bar{x}_{\text{minimum number of territories treatment}} = 47.2$, $0.25 < p < 0.1$; $\bar{x}_{\text{maximum number of territories control}} = 63.1$, $\bar{x}_{\text{maximum number of territories treatment}} = 57.6$, $0.25 < p < 0.1$).

Table 13. Mean number of territories for minimum and maximum density estimates in controls and treatments by bird species.

	Minimum density			Maximum density		
Species	Control	Treatment	p-value	Control	Treatment	p-value
American robin	2.7	2.5	$p>0.25$	3.3	3.2	$p>0.25$
Chipping sparrow	17.0	15.3	$p>0.25$	9.0	17.5	$p>0.25$
Dark-eyed junco	7.4	7.8	$p>0.25$	9.3	9.6	$p>0.25$
Dusky flycatcher	3.3	2.1	$0.25<p<0.1$	4.3	3.8	$p>0.25$
Mountain chickadee	3.1	3.5	$p>0.25$	3.9	4.4	$p>0.25$
Red-breasted nuthatch	4.4	3.3	$0.25<p<0.1$	5.6	4.0	$0.25<p<0.1$
Solitary vireo	1.4	1.2	$p>0.25$	1.9	2.0	$p>0.25$
Townsend's solitare	2.6	2.6	$p>0.25$	2.9	3.0	$p>0.25$
Vesper sparrow	1.9	2.1	$p>0.25$	2.5	2.7	$p>0.25$
Western tanager	1.6	0.9	$0.25<p<0.1$	1.9	1.0	$0.25<p<0.1$
Yellow-rumped warbler	8.7	7.1	$p>0.25$	0.4	8.3	$p>0.25$
All	4.9	4.4	$p>0.25$	5.9	5.4	$p>0.25$

Table 14. Mean number of territories of ground feeders for both minimum and maximum density estimates for control and treatment sites.

	Minimum density			Maximum density		
Species	Control	Treatment	p-value	Control	Treatment	p-value
American robin	2.7	2.5	$p > 0.25$	3.3	3.2	$p > 0.25$
Chipping sparrow	17.0	15.3	$p > 0.25$	19.0	17.5	$p > 0.25$
Dark-eyed junco	7.4	7.8	$p > 0.25$	9.3	9.6	$p > 0.25$
Townsend's solitare	2.6	2.6	$p > 0.25$	2.9	3.0	$p > 0.25$
Vesper sparrow	1.9	2.1	$p > 0.25$	2.5	2.7	$p > 0.25$
All	6.3	6.1	$0.25 < p < 0.1$	7.4	7.2	$p > 0.25$

Table 15 Density of foliage-feeding species for both minimum and maximum density estimates for control and treatment sites.

	Minimum density			Maximum density		
Species	Control	Treatment	p-value	Control	Treatment	p-value
Mountain chickadee	3.1	3.5	$p > 0.25$	3.9	4.4	$p > 0.25$
Red-breasted nuthatch	4.4	3.3	$0.25 < p < 0.1$	5.6	4.0	$0.25 < p < 0.1$
Western tanager	1.6	0.9	$0.25 < p < 0.1$	1.9	1.0	$0.25 < p < 0.1$
Yellow-rumped warbler	8.7	7.1	$p > 0.25$	10.4	8.3	$p > 0.25$

Bird abundance: Number of detections

A significant difference in the number of detections was found for only one species (Table 16). Mountain chickadees were detected more often in treatment sites than in controls ($0.05 < p < 0.025$). Mountain chickadees were the only species of the foliage-feeding group to be detected more often in treated sites. The other species of this group, and the group as a whole were not detected significantly more often in control sites than in treatment sites (Table 16 and 17).

Individual ground-feeding species were detected equally in treatment and control sites (Table 16). Furthermore, the ground-feeding species were detected equally in treatment and control sites (Table 17). Finally, each of the species of woodpeckers was detected equally in treatments and controls. When taken together as a group these six species were also detected equally in treatments and controls (Table 17). However, when Red-naped sapsuckers were removed from this group and the analysis was redone, the woodpecker group was detected significantly more often (but only marginally so) in treatment sites ($0.1 < p < 0.05$; Table 17).

Bird/habitat relationships: Composite maps

In controls, the most common habitat was dense forest followed by open forest, open and riparian. In treatments, open forest was the most common habitat type, followed by

Table 16. Mean number of detections for 22 species in control and treatment sites.

Species	Control	Treatment	p-value
Black-backed woodpecker	7.3	13.3	p >0.25
Three-toed woodpecker	2.7	7.7	p >0.25
Hairy woodpecker	10.7	15.3	p >0.25
Northern flicker	10.3	14.0	p >0.25
Pileated woodpecker	3.3	10.0	0.25<p<0.1
Red-naped sapsucker	16.0	22.3	p >0.25
Mountain chickadee	59.3	84.0	0.05<p<0.025
Red-breasted nuthatch	109.7	64.7	0.25<p<0.1
Yellow-rumped warbler	75.7	75.3	p >0.25
Western tanager	11.3	3.3	0.25<p<0.1
Chipping sparrow	225.0	217.7	p >0.25
Dark-eyed junco	80.7	74.7	p >0.25
Vesper sparrow	22.3	27.3	p >0.25
American robin	50.0	49.3	p >0.25
Townsend's solitare	35.3	42.7	p >0.25
Dusky flycatcher	35.7	21.3	0.25<p<0.1
Swainson's thrush	3.3	0.0	p >0.25
Ruby-crowned kinglet	4.7	1.0	p >0.25
Warbling vireo	1.0	1.7	p >0.25
Solitary vireo	9.0	12.0	p >0.25
Orange-crowned warbler	7.7	1.3	p >0.25
Wilson's warbler	5.7	0.0	0.25<p<0.1

Table 17. Number of detections of four species groups in control and treatment sites.

Species	Control	Treatment	p-value
Foliage-feeders	64.0	56.8	$p > 0.25$
Ground-feeders	82.7	82.3	$p > 0.25$
Woodpeckers	8.4	13.8	$0.25 < p < 0.1$
Woodpeckers minus sapsuckers	6.9	12.1	$0.1 < p < .0.05$

dense forest, open, and riparian (Table 18). Dense forest habitat was patchy in both treatment and control sites, but more so in controls. Open habitat was less common in treatment sites than expected because of my inability to detect discrete openings in treatment sites. Riparian habitat was extremely rare in both treatments and controls. The main portion of riparian habitat occurred in one control plot. Smaller, isolated areas were found in one other control plot and one treatment plot. The large standard deviation of all estimates illustrates the heterogeneity of these sites, especially when combined with one another.

<u>Bird/habitat relationships</u>: Bird communities

The dominance-diversity curves for each habitat type in control sites are presented in Figure 4. Communities in dense forest, open forest and open habitats show a similar pattern to overall dominance-diversity curves (Figure 3). Each community is dominated by one species (Chipping sparrow). There also is a sharp drop from the most abundant to the next most abundant species. This drop is followed by a more gentle decline from the second to the ninth most abundant species. After the ninth most abundant species the curves for all three communities level off and are comprised of many rare species (Tables 19-21). Open habitat deviates somewhat from this pattern. The initial decline from the most abundant species is steeper, and the

Table 18. Average percent coverage of each habitat type in each 0.25 ha cell for controls and treatments.

	Control		Treatment	
Habitat	Mean	(SE)	Mean	(SE)
Dense forest	49.9	(14.5)	9.3	(11.4)
Open forest	27.5	(14.5)	86.9	(13.5)
Open	21.3	(10.7)	3.5	(7.1)
Riparian	1.3	(3.1)	0.3	(1.4)

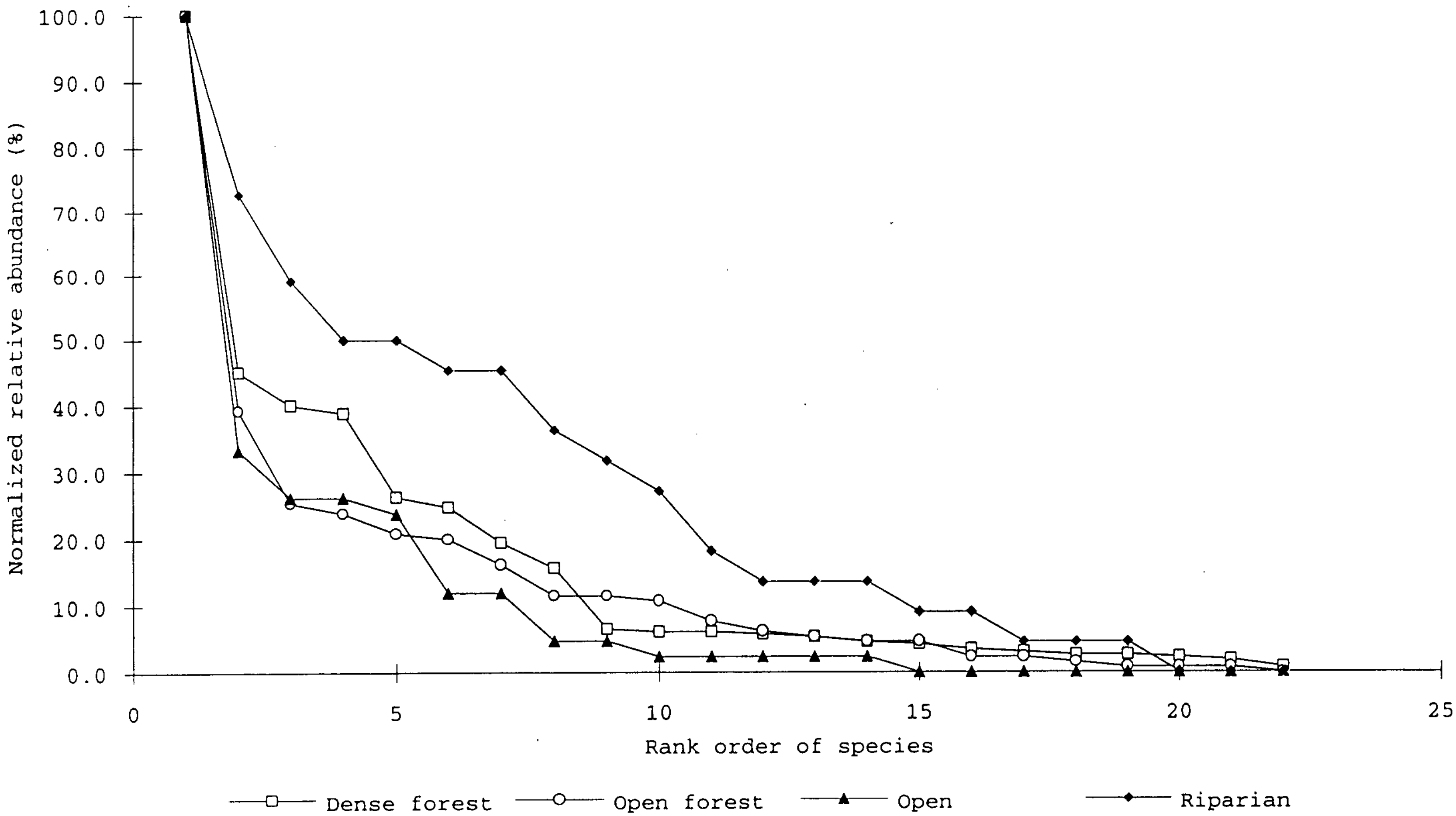


Figure 4. Dominance diversity curve for control plots in each of four habitat types.

Table 19. Rank order, number of detections and percentage of the total number of detections recorded that occurred in dense forest habitat in both control and treatment sites.

	Control (41 % of cells)				Treatment (5 % of cells)			
Rank	Species	Number	%	Total	Species	Number	%	Total
1	Chipping sparrow	105	15.6	675	Chipping sparrow	10	1.5	653
2	Red-breasted nuthatch	63	19.1	329	Yellow-rumped warbler	9	4.0	226
3	Yellow-rumped warbler	44	19.4	227	Dark-eyed junco	7	3.1	224
4	Dark-eyed junco	38	15.7	242	Mountain chickadee	7	2.8	252
5	Mountain chickadee	28	15.7	178	Dusky flycatcher	6	9.4	64
6	American robin	26	17.3	150	American robin	5	3.4	148
7	Dusky flycatcher	23	21.5	107	Townsend's solitare	4	3.1	128
8	Townsend's solitare	17	16.0	106	Black-backed woodpecker	3	7.5	40
9	Red-naped sapsucker	11	22.9	48	Red-breasted nuthatch	3	1.5	194
10	Hairy woodpecker	9	28.1	32	Solitary vireo	3	9.1	33
11	Western tanager	7	20.6	34	Northern flicker	2	4.8	42
12	Wilson's warbler	5	29.4	17	Red-naped sapsucker	2	3.3	61
13	Northern flicker	4	12.9	31	Pileated woodpecker	1	3.3	30
14	Solitary vireo	4	14.8	27	Ruby-crowned kinglet	1	100.0	1
15	Black-backed woodpecker	3	15.8	19	Vesper sparrow	1	1.2	82
16	Vesper sparrow	2	3.0	67	Western tanager	1	10.0	10
17	Orange-crowned warbler	1	4.3	23	Hairy woodpecker	0	0.0	46
18	Swainson's thrush	1	10.0	10	Orange-crowned warbler	0	0.0	4
19	Three-toed woodpecker	1	9.1	11	Swainson's thrush	0	0.0	0
20	Warbling vireo	1	33.3	3	Three-toed woodpecker	0	0.0	23
21	Pileated woodpecker	0	0.0	10	Warbling vireo	0	0.0	0
22	Ruby-crowned kinglet	0	0.0	14	Wilson's warbler	0	0.0	0

Table 20. Rank order, number of detections and percentage of the total number of detections recorded that occurred in open forest habitat in both control and treatment sites.

	Control (16 % of cells)				Treatment (90 % of cells)			
Rank	Species	Number	%	Total	Species	Number	%	Total
1	Chipping sparrow	70	10.4	675	Chipping sparrow	337	51.6	653
2	Red-breasted nuthatch	33	10.0	329	Mountain chickadee	135	53.6	252
3	Dark-eyed junco	20	8.3	242	Dark-eyed junco	108	48.2	224
4	Vesper sparrow	15	22.4	67	Red-breasted nuthatch	103	53.1	194
5	American robin	14	9.3	150	Yellow-rumped warbler	99	43.8	226
6	Yellow-rumped warbler	14	6.2	227	Townsend's solitare	82	64.1	128
7	Mountain chickadee	12	6.7	178	American robin	73	49.3	148
8	Dusky flycatcher	11	10.3	107	Vesper sparrow	52	63.4	82
9	Northern flicker	10	32.3	31	Red-naped sapsucker	34	55.7	61
10	Townsend's solitare	8	7.5	106	Hairy woodpecker	33	71.7	46
11	Hairy woodpecker	6	18.8	32	Northern flicker	25	59.5	42
12	Red-naped sapsucker	5	10.4	48	Black-backed woodpecker	17	42.5	40
13	Three-toed woodpecker	4	36.4	11	Dusky flycatcher	15	23.4	64
14	Western tanager	4	11.8	34	Pileated woodpecker	15	50.0	30
15	Orange-crowned warbler	2	8.7	23	Solitary vireo	12	36.4	33
16	Wilson's warbler	2	11.8	17	Three-toed woodpecker	10	43.5	23
17	Black-backed woodpecker	0	0.0	19	Western tanager	5	50.0	10
18	Pileated woodpecker	0	0.0	10	Orange-crowned warbler	0	0.0	4
19	Ruby-crowned kinglet	0	0.0	14	Ruby-crowned kinglet	0	0.0	1
20	Solitary vireo	0	0.0	27	Swainson's thrush	0	0.0	0
21	Swainson's thrush	0	0.0	10	Warbling vireo	0	0.0	0
22	Warbling vireo	0	0.0	3	Wilson's warbler	0	0.0	0

Table 21. Rank order, number of detections and percentage of the total number of detections recorded that occurred in open habitat in both control and treatment sites.

	Control (6 % of cells)				Treatment (2 % of cells)			
Rank	Species	Number	%	Total	Species	Number	%	Total
1	Chipping sparrow	42	6.2	675	Chipping sparrow	9	1.4	653
2	Dark-eyed junco	14	5.8	242	Red-naped sapsucker	7	11.5	61
3	Red-breasted nuthatch	11	3.3	329	Vesper sparrow	5	6.1	82
4	Yellow-rumped warbler	11	4.8	227	American robin	3	2.0	148
5	American robin	10	6.7	150	Dusky flycatcher	3	4.7	64
6	Mountain chickadee	5	2.8	178	Mountain chickadee	3	1.2	252
7	Vesper sparrow	5	7.5	67	Townsend's solitare	3	2.3	128
8	Dusky flycatcher	2	1.9	107	Dark-eyed junco	2	0.9	224
9	Townsend's solitare	2	1.9	106	Pileated woodpecker	2	6.7	30
10	Black-backed woodpecker	1	5.3	19	Red-breasted nuthatch	2	1.0	194
11	Hairy woodpecker	1	3.1	32	Black-backed woodpecker	1	2.5	40
12	Northern flicker	1	3.2	31	Northern flicker	1	2.4	42
13	Red-naped sapsucker	1	2.1	48	Hairy woodpecker	0	0.0	46
14	Orange-crowned warbler	0	0.0	23	Orange-crowned warbler	0	0.0	4
15	Pileated woodpecker	0	0.0	10	Ruby-crowned kinglet	0	0.0	1
16	Ruby-crowned kinglet	0	0.0	14	Solitary vireo	0	0.0	33
17	Solitary vireo	0	0.0	27	Swainson's thrush	0	0.0	0
18	Swainson's thrush	0	0.0	10	Three-toed woodpecker	0	0.0	23
19	Three-toed woodpecker	0	0.0	11	Warbling vireo	0	0.0	0
20	Warbling vireo	0	0.0	3	Wilson's warbler	0	0.0	0
21	Western tanager	0	0.0	34	Western tanager	0	0.0	10
22	Wilson's warbler	0	0.0	17	Yellow-rumped warbler	0	0.0	226

Table 22. Rank order, number of detections and percentage of the total number of detections recorded that occurred in riparian habitat in both control and treatment sites.

	Control (5 % of cells)				Treatment (2 % of cells)			
Rank	Species	Number	%	Total	Species	Number	%	Total
1	Dusky flycatcher	22	20.6	107	Chipping sparrow	16	2.5	653
2	Chipping sparrow	16	2.4	675	Dusky flycatcher	9	14.1	64
3	Dark-eyed junco	13	5.4	242	Red-breasted nuthatch	7	3.6	194
4	American robin	11	7.3	150	Red-naped sapsucker	7	11.5	61
5	Mountain chickadee	11	6.2	178	Dark-eyed junco	6	2.7	224
6	Orange-crowned warbler	10	43.5	23	Mountain chickadee	6	2.4	252
7	Ruby-crowned kinglet	10	71.4	14	Pileated woodpecker	3	10.0	30
8	Red-breasted nuthatch	8	2.4	329	Yellow-rumped warbler	3	1.3	226
9	Yellow-rumped warbler	7	3.1	227	American robin	2	1.4	148
10	Wilson's warbler	6	35.3	17	Hairy woodpecker	2	4.3	46
11	Townsend's solitare	4	3.8	106	Vesper sparrow	2	2.4	82
12	Hairy woodpecker	3	9.4	32	Northern flicker	1	2.4	42
13	Red-naped sapsucker	3	6.3	48	Black-backed woodpecker	0	0.0	40
14	Solitary vireo	3	11.1	27	Orange-crowned warbler	0	0.0	4
15	Swainson's thrush	2	20.0	10	Ruby-crowned kinglet	0	0.0	1
16	Western tanager	2	5.9	34	Solitary vireo	0	0.0	33
17	Three-toed woodpecker	1	9.1	11	Swainson's thrush	0	0.0	0
18	Warbling vireo	1	33.3	3	Townsend's solitare	0	0.0	128
19	Black-backed woodpecker	0	0.0	19	Three-toed woodpecker	0	0.0	23
20	Northern flicker	0	0.0	31	Warbling vireo	0	0.0	0
21	Pileated woodpecker	0	0.0	10	Western tanager	0	0.0	10
22	Vesper sparrow	0	0.0	67	Wilson's warbler	0	0.0	0

plateau in abundance occurs earlier. Further, open habitat supports fewer species than the other three habitats.

Riparian habitat in controls differs markedly from the other habitats. First, this habitat is not dominated by one species to the same extent as the other three habitats. Second, the rank order of abundance is different; Dusky flycatchers, not Chipping sparrows are the most abundant species. Species such as Orange-crowned warbler, Ruby-crowned kinglet, Wilson's warbler are more abundant in this habitat (Table 22). Finally, there is a more gradual decline in species abundance in riparian habitat.

In treatments, the community in the open forest habitat closely mimics that for the over all treatment dominance-diversity curve (Figure 5). This is not surprising given that those cells with >59 % open forest coverage make up 90% of the total number of cells (Table 10). As in controls, there is a sharp drop from the most abundant to the next most abundant species. This is followed by a more gentle decline from the second to the ninth most abundant species. After the ninth most abundant species the curve for open forest habitat levels off and are is made up of many rare species (Table 20). Communities in both dense forest and open habitat were not dominated by one species to the same extent that open forest communities were (Tables 19 and 21). The decline in species abundance was less pronounced in dense forest habitat than any of the other three habitat types.

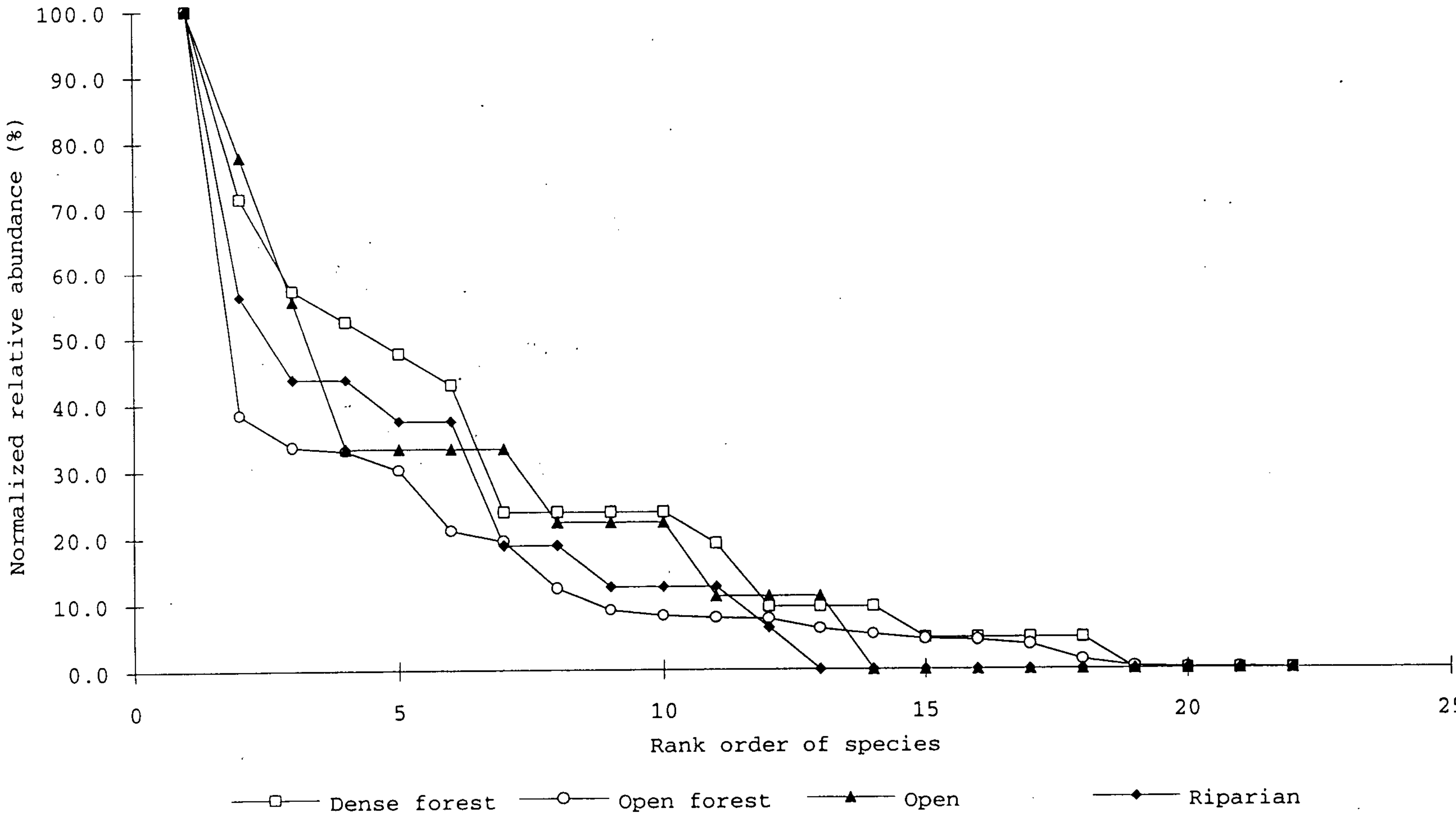


Figure 5. Dominance diversity curves for treatment plots in each of four habitat types

Riparian habitat and open habitat supported the least species (Table 22). There is less evidence of a riparian specific community in treatments than in controls, likely due to the scarcity of this habitat in treatments (Table 18).

Bird/habitat relationships: Within plot similarity analysis

In controls, communities in dense forest and riparian habitat were the most similar of all pairwise comparisons (Table 23). These two communities were the most similar because cells with riparian habitat usually contain a large amount of dense forest habitat. As a result, communities in dense forest habitat contain many of the species associated with riparian habitat. Communities in open forest and dense forest habitat were also quite similar because of the shared presence of some of the species associated with more open habitat (e.g. Northern flicker, and Vesper sparrow) and the shared absence of some of the riparian species (e.g. Ruby-crowned kinglet). The two most dissimilar pairings of communities were those between dense forest and open habitat, and open and riparian habitat. Dense forest and open habitat were dissimilar primarily because many of the species noted as occurring in riparian habitats were detected in dense forest habitat, but not in open habitats (e.g., Orange-crowned warbler, Wilson's warbler, Swainson's thrush, and Warbling vireo). Open and riparian habitats were the least similar communities

Table 23. Similarity indices for communities in each habitat type based on presence/absence data using Jaccard's similarity index.

	Controls			
	Dense forest	Open forest	Open	Riparian
Dense forest	1.00			
Open forest	0.80	1.00		
Open	0.65	0.71	1.00	
Riparian	0.81	0.70	0.48	1.00

	Treatments			
	Dense forest	Open forest	Open	Riparian
Dense forest	1.00			
Open forest	0.83	1.00		
Open	0.75	0.71	1.00	
Riparian	0.69	0.71	0.72	1.00

because all of the species primarily associated with riparian habitat were absent in open habitat, and those species associated with open habitat were absent in riparian habitat. Open forest and riparian habitat were relatively similar due to the spilling over of riparian species such as Wilson's and Orange-crowned warblers into cells with high amounts of open forest habitat. Open forest and open communities were not as similar as might have been expected for similar reasons. Open forest communities contained species that would usually be found in riparian habitat.

In treatment sites there was much less variability in similarity measures. Dense forest and open forest communities were the most similar (Table 23). Those communities in dense forest and open, dense forest and riparian, open forest and open, open forest and riparian, and open and riparian were equally similar to one another. This lack of variability was likely due to the lack of dense forest, open and riparian habitat in treatment sites (Table 10). The lack of riparian habitat led to the absence of three species associated with riparian habitat: Swainson's thrush, Wilson's warbler and Warbling vireo). This resulted in a much less diverse community in riparian habitats of treatment sites.

Bird/habitat relationships: Chi-square analysis

Of the 22 species and four species groups examined, 12 species and two species groups were found to have significant associations with one, or a combination of, habitat types. Those significant results are presented in Table 24. From this analysis two main points emerge. First, only five significant Chi-square values represented cells with extreme habitat classes (i.e., >50% coverage of that habitat). The remaining significant Chi-square values were derived from low and intermediate cover classes. Second, many of the significant Chi-square values were derived from a low percentage of the total number of detections.

These significant Chi-square values are difficult to interpret because it is hard to determine if a species was associated with that particular habitat type, avoiding another habitat type, or occurring in a cell with a combination of habitat types. Further problems in interpretation arise when a species is found to have a significant association for a particular extreme habitat while >50% of the total number of detections occur in other habitat types, or other classes of the same habitat type. Two species, Northern flicker and Vesper sparrow were detected cells with >75% coverage of open forest habitat in control sites. Dusky flycatchers were detected in sites with between 51% and 75% coverage of dense forest. Red-naped sapsuckers and woodpeckers were detected in sites

Table 24. Summary of significant Chi-square tests for each species in treatment and control sites.

Habitat	Cover Class[a]	Condition	Species	Obs	Exp	$\chi 2$	Detections %	Detections No.
O. forest	1	Treatment	American robin	11	4	9.7	7.4	148
O. forest	1	Treatment	Black-backed woodpecker	5	1	12.0	12.5	40
D. forest	1	Treatment	Chipping sparrow	195	150	13.4	29.9	675
Riparian	1	Control	Dusky flycatcher	24	7	36.4	22.4	107
Riparian	2	Control	Dusky flycatcher	10	2	40.1	9.3	107
D. forest	3	Treatment	Dusky flycatcher	5	1	10.8	7.8	64
O. forest	1	Treatment	Dusky flycatcher	7	2	13.4	10.9	64
Riparian	1	Treatment	Dusky flycatcher	9	1	46.6	14.1	64
D. forest	0	Control	Northern flicker	3	0	23.3	9.7	31
O. forest	4	Control	Northern flicker	7	2	10.6	22.6	31
Riparian	1	Control	Orange-crowned warbler	12	2	67.1	52.2	23
Riparian	0	Control	Ruby-crowned kinglet	1	12	10.8	7.1	14
Riparian	1	Control	Ruby-crowned kinglet	4	1	65.6	64.3	14
Riparian	2	Control	Ruby-crowned kinglet	9	0	63.7	28.6	14

[a] = Cover classes: 0 = 0%, 1 = 1-25%, 2 = 26-50%, 3 = 51-75%, 4 = >75%.
[b] = O. forest = Open forest, D. forest = Dense forest

Table 24 continued. Summary of significant Chi-square tests for each species in treatment and control sites.

Habitat	Cover Class[a]	Condition	Species	Obs	Exp	$\chi 2$	Detections %	Detections No.
O. forest[b]	1	Treatment	Red-naped sapsucker	8	2	20.8	13.1	61
Open	3	Treatment	Red-naped sapsucker	7	1	34.3	11.5	61
Riparian	1	Treatment	Red-naped sapsucker	7	1	27.4	11.5	61
O. forest	2	Control	Townsend's solitare	37	21	11.8	34.9	106
D. forest	1	Control	Vesper sparrow	29	15	13.8	43.3	67
O. forest	4	Control	Vesper sparrow	12	5	11.4	17.9	67
Open	3	Control	Vesper sparrow	13	5	10.9	19.4	67
Riparian	1	Control	Wilson's warbler	7	1	28.4	41.2	17
D. forest	1	Control	Yellow-rumped warbler	28	50	9.6	12.3	227
D. forest	0	Treatment	Yellow-rumped warbler	95	147	18.3	42	226
D. forest	1	Treatment	Yellow-rumped warbler	89	52	26.4	39.4	226
D. forest	1	Treatment	Foliage-feeding species	199	157	11.3	29.2	682
O. forest	1	Treatment	Woodpeckers	6	7	10.5	6.6	242
Open	3	Treatment	Woodpeckers	12	4	15.1	5.0	242
Riparian	1	Treatment	Woodpeckers	12	5	10.6	5.0	242

[a] = Cover classes: 0 = 0%, 1 = 1-25%, 2 = 26-50%, 3 = 51-75%, 4 = >75%.
[b] = O. forest = Open forest, D. forest = Dense forest

with between 51% and 75% coverage of open habitat. In each case, however, these species and species groups were only detected in each of these habitat types less than 25% of the time. In addition, except for Orange-crowned warbler, and Ruby-crowned kinglet in control sites, the number of detections that led to significant Chi-square values did not exceed 50% of the total number of detections.

Those species that had both significant Chi-square values and had >50% of their detections in that habitat class appeared to be closely associated with riparian habitat. In control sites, Ruby-crowned kinglets avoided sites without riparian habitat more than expected, and were detected in cells with both low and high amounts of riparian more than expected. Also in controls, Orange-crowned warblers were detected in sites with low amounts of riparian habitat more than expected.

Bird/habitat relationships: Discriminant function analysis

Discriminant function analysis was conducted with 22 bird species using four habitat types (open forest, dense forest, open, and riparian habitat) as predictors of presence or absence of bird detections in 50x50-m cells. These analyses yielded five significant discriminant functions. All significant discriminant functions failed Box's M test for homogeneity of the variance/covariance matrices, suggesting caution in interpreting the functions. However, discriminant function analysis is robust to violations of the homogeneity of the variance/covariance matrix with equally sized or large samples (greater than 30 observations in each group; Freese 1964). This criterion was met by all 5 significant discriminant functions. Chi-square values, p-values, number of cases in each group, and % correct classification of each function are presented in Table 25. The standardized canonical discriminant function coefficients and loading matrices of the pooled within group correlations between predictor variables and each discriminant function are presented in Table 26. Following Comrey (1973), loadings of less than 0.45 were not interpreted. Group means for each discriminant function are presented in Table 27.

In control sites, three variables contributed to the discrimination of cells with and without Vesper sparrow detections. The loading matrix of correlations between predictor variables (Table 26) suggest that percent coverage of dense forest was the primary variable distinguishing occupied and unoccupied cells. Vesper sparrows occurred in cells with lower amounts of dense forest ($\bar{x}=_{presence}=39.8\%$) than those cells where Vesper sparrows were absent ($\bar{x}_{absence}= 51.7\%$). Also contributing to the separation of these two groups was percent cover of open forest habitat. Vesper sparrows were found in cells with a higher amount of open forest cover ($\bar{x}_{presence}=35.1\%$) than in cells were birds were not detected ($\bar{x}_{absence}=26.1\%$). Finally, Vesper sparrows were

Table 25. Summary of discriminant function analysis.

	$\chi 2$	p	Number absent	Number present	% classified correctly	Box M
Treatments						
Dusky flycatcher	24.5	<0.001	255	47	76.8	<0.01
Yellow-rumped warbler	23.2	<0.001	160	142	64.9	<0.01
Controls						
Chipping sparrow	13.0	<0.01	53	255	69.8	<0.01
Dusky flycatcher	17.9	<0.01	236	72	78.9	<0.01
Vesper sparrow	12.8	<0.01	259	49	58.4	<0.01

Table 26. Discriminant function coefficients and pooled within group correlations for direct discriminant function analysis for treatments and controls.

	Standardized Canonical Discriminant Function Coefficients		Pooled Within Group Correlations	
Treatments				
Dusky flycatcher	Open forest	4.52	Open forest	0.90
	Dense forest	2.97	Dense forest	-0.83
	Open	2.20	Riparian	-0.42
			Open	-0.27
Yellow-rumped warbler	Open forest	1.09	Dense forest	0.76
	Dense forest	1.68	Open	-0.61
	Open	-0.06	Open forest	-0.29
			Riparian	-0.24
Controls				
Chipping sparrow	Open forest	4.35	Riparian	-0.87
	Dense forest	4.01	Open forest	0.50
	Open	2.71	Open	-0.34
			Dense forest	-0.06
Dusky flycatcher	Open forest	4.75	Riparian	-0.97
	Dense forest	4.52	Open forest	0.28
	Open	3.48	Dense forest	-0.14
			Open	0.08
Vesper sparrow	Open forest	2.42	Dense forest	-0.85
	Dense forest	1.54	Open forest	0.63
	Open	1.88	Riparian	-0.45
			Open	0.41

Table 27. Mean percent coverage of all four habitat types in 50x50-m cells (group means) where each species was either absent or present for each of the significant discriminant functions.

	Habitat type							
	Open forest		Dense forest		Open		Riparian	
Species	Abs	Pres	Abs	Pres	Abs	Pres	Abs	Pres
Treatments								
American robin	89.1	81.9	7.8	12.3	2.6	5.5	0.4	0.3
Dusky flycatcher	89.4	73.0	7.3	20.2	3.1	5.8	0.2	1.0
Yellow-rumped warbler	88.7	84.8	5.3	13.7	5.5	1.3	0.5	0.2
Controls								
American robin	26.5	27.5	52.3	44.9	19.9	24.1	1.3	1.2
Chipping sparrow	21.7	28.8	50.6	49.7	24.2	20.7	3.4	0.8
Dusky flycatcher	28.5	24.3	49.4	51.5	21.5	20.6	0.6	3.6
Vesper sparrow	26.1	35.1	51.7	39.8	20.6	35.0	1.5	0.1

found in cells with lower amounts of riparian habitat ($\bar{x}_{presence}$= 0.1%) than in cells without Vesper sparrow detections ($\bar{x}_{absence}$= 1.5%).

One significant discriminant function was generated for Yellow-rumped warblers in treatment sites. The loading matrix in Table 26 indicates that two variables, dense forest and open habitat, distinguish cells with and without Yellow-rumped warbler detections. These birds were found in cells with a higher percent coverage of dense forest ($\bar{x}_{presence}$= 13.5%) compared with cells where Yellow-rumped warblers were not detected ($\bar{x}_{absence}$= 5.3%). In addition, cells with Yellow-rumped warblers detections tended to have a lower amount of open habitat ($\bar{x}_{presence}$=1.3%); cells without Yellow-rumped warbler detections had a higher amount of open habitat ($\bar{x}_{absence}$= 5.3%).

One significant discriminant function was generated for Chipping sparrows in control sites. The primary variable responsible for separating these groups was the amount of riparian habitat (Table 26). Chipping sparrows were found in cells with low percent coverage of riparian habitat ($\bar{x}_{presence}$=0.8%, vs. $\bar{x}_{absence}$=3.4%). In addition, the percent cover of open forest habitat also contributed to the separation of cells with and without detections. Chipping sparrows were detected in cells with a higher percent cover of open forest habitat ($\bar{x}_{presence}$= 28.8%) than in cells where they were not detected ($\bar{x}_{absence}$= 21.7%).

Dusky flycatchers were the only species to have significant discriminant functions in both treatments and controls. In controls the loading matrix (Table 26) suggests that riparian habitat was the only variable that contributed to the separation of cells with and without Dusky flycatcher detections. This species was found in cells with a higher amount of riparian habitat ($\bar{x}_{presence}$=3.6%) than in cells where they were absent ($\bar{x}_{absence}$=0.6%). In treatments, three variables played a role in predicting Dusky flycatcher presence (Table 26). Open forest was the variable primarily responsible for the separation. Dusky flycatchers were detected in cells with lower amounts of open forest habitat ($\bar{x}_{presence}$=73%, $\bar{x}_{absence}$=89%). The amount of dense forest was the next most influential variable. Dusky flycatchers were found in cells with higher amounts of dense habitat than in cells where they were absent ($\bar{x}_{presence}$=20.2%, $\bar{x}_{absence}$=7.3%). Finally, this species was found in cells with more riparian habitat than in cells where they were absent ($\bar{x}_{presence}$=1.0%, $\bar{x}_{absence}$=0.2%).

Discussion

Community structure

This thinning project appeared to have little affect on the structure of the forest bird community. Dominance diversity curves of communities in thinned and unthinned sites were very similar (Figure 3). Each community consisted of one very common species (Chipping sparrows), 7-8 relatively abundant species, and 14-15 relatively uncommon species. In both treatment and controls the 9 most abundant species were the same although their rank orders were somewhat different (Table 11). The high values of Jaccard's and Morisita's indices of similarity (0.92 and 0.96, respectively) further illustrate the minimal impact that thinning had on overall community structure.

Relative abundance

Thinning also seems to have had little affect on the relative abundance (either in terms of the number of territories or number of detections) of any of the species examined. Ground and shrub foragers such as Dark-eyed junco, Vesper sparrow, Chipping sparrow, Townsend's solitare, and American robin were expected to increase in abundance because of an anticipated increase in understory vegetation but did not. There is a general trend for the relative abundance of ground and shrub feeding/nesting species to increase after logging (Hagar 1960, Flack 1976, Franzreb and Omhart 1978), fire (Bock and Lynch 1970, Raphael et al. 1987) and thinning (Hagar 1990, Stribling et al. 1990, DeGraff et al. 1991) due to increases in ground and shrub cover. In my study the lack of response of the ground-feeding species of this bird community is consistent with the negligible response of the vegetation community. The only habitat variable that increased significantly due to thinning was down and dead woody material. There was no significant increase in the percent cover of herbs, grass and shrubs (Table 9). In studies where the increase of ground-feeding species was attributed to increases in the amount of logging slash, the slash was arranged in discrete piles (Franzreb and Omhart 1978). These slash piles afforded ground-feeding species, particularly Dark-eyed junco, increased foraging opportunities, escape cover and observation posts. Slash in this study was left where it was felled. This approach resulted in a more uniform distribution of slash (at least in the horizontal plane) than in other studies.

The thinning treatment did not have any affect on foliage-feeding species. Foliage-feeding species such as Yellow-rumped warbler, Mountain chickadee, Red-breasted nuthatch, and Western tanagers were expected to decrease in abundance because of a reduction in canopy volume on a per hectare basis. The amount of canopy volume has been related to the availability of food for foliage-feeding species (Sturnam 1968, Szaro and Balda 1979). In addition, canopy volume has also been associated with the relative abundance of several foliage-feeding species. Increases in the abundance of foliage-feeding species have been noted in instances where forest canopies develop and close after harvesting or fire (Flack 1976, Raphael et al. 1987, Szaro and Balda 1986). Conversely, decreases in the abundance of foliage-feeding species have been correlated with decreases in canopy volume after harvesting (Franzreb and Omhart 1978, Morgan et al. 1989). In my study there was significantly less canopy volume per hectare of Douglas-fir trees <10 cm dbh in treatment sites than in control sites (Table 7); however, there was no significant decrease in the relative abundance of any individual species for each density estimate. In fact, Mountain chickadees were detected significantly more in treatment sites (Tables 15). This was unexpected as it was believed that chickadees would be more abundant where foraging opportunities were greatest. However, the difference in the number of Mountain chickadees in treatment sites was largely due to the influence of one plot. Mountain chickadees were very abundant in the Dam Lake study site. I located 7 nests within this plot.

The lack of response of the bird community may be attributable to the following factors. First, the treatment and hence reduction of the canopy volume, may not have been severe enough to make food limiting. Second, an abundance of food (spruce budworm and Douglas-fir tussock moth) negated the loss of available foraging habitat.

The predicted decline in foliage-feeding species in treated sites was based on the assumption that food was limiting for this group of species because of reduced canopy volume. Thinning in this project was directed at a specific component of the forest stratum (trees <15 cm dbh). A significant proportion of this stratum was dead or dying, in both control and treatment sites at the time of study (30% and 21% respectively; Table 4) thereby representing habitat that was already unavailable to foliage-feeding species. Had thinning occurred prior to the extensive defoliation of these stands by spruce budworm and Douglas-fir tussock moth, the effect that this thinning trial had on limiting foraging habitat could have been greater. In addition, these stands are managed on an un-even aged basis. Because thinning was directed at the younger trees within this forest, a significant amount of foraging habitat still remained in the form of overstory Douglas-fir and Ponderosa pine foliage (Table 6). Therefore, the assumption that food was made limiting as a result of this treatment may have been erroneous. To make food limiting, and therefore to see the decrease in the number of foliage-feeding species, thinning would have had to have been directed at removing a greater number of healthier trees, and/or trees in larger size classes.

The preceding argument assumes that a reduction in canopy volume would have led to a corresponding reduction in the availability of food. However, spruce budworm and Douglas-fir tussock moth were both sufficiently abundant in these stands that the British Columbia Forest Service felt it necessary to spray some of my study sites with BTK (*Bacillus thurigensis*). Insect outbreaks, especially those associated with forest pests, and specifically spruce budworm, have been correlated with numerical responses of some species of birds (MacArthur 1958, Blais and Banks 1964, Morris et al. 1958, Mattson et al. 1968, Crawford and Jennings 1989). Therefore, elevated levels of spruce budworm may have led to increased population levels of some species in both controls and treatments thereby damping the potential negative effects of the habitat manipulation. Whether spruce budworm and Douglas-fir tussock moth populations led to a numerical response in any species or not, the presence of this abundant food source likely had a confounding affect in this manipulation. The presence of this superabundant food source could easily have made up for the reduced availability of foraging habitat that resulted from thinning. When budworm populations crash, differences in the abundance of foliage-feeding species may become apparent.

The modest significant difference in woodpecker detections favoring treatments over controls may have been a result of bias in detectability (Table 16). Woodpeckers may have been more easily detected in treatments than in controls. The openness of treatments did make it easier to detect birds visually. However, if visibility was the reason for differences in the number of detections, the bias should have been more systematic. In other words, more species, or groups of species, should have been detected more often in treated sites. This was not the case. Only Mountain chickadees were detected more often in treated sites. A more likely explanation for the difference in the number of detections is that woodpeckers were using treatment sites preferentially because of increased foraging opportunities. The practice of thinning and leaving cut material on site can contribute to increases in bark beetle (*Dendroctonus frontalis*) populations (Moser et al. 1987, Nebecker and Hodges 1985, Conner et al. 1991). Woodpeckers are major predators of bark beetles (Moeck and Safranyik 1984), and like passerines, have been known to show both numerical and functional responses to increased food abundance (Conner and Crawford 1974, Knight 1958, Koplin 1969, Otvos 1970). The abundant down and dead material in treatment sites and probable elevated beetle populations, likely represented a food source that drew woodpeckers into treatment sites. Observations of foraging woodpeckers supports this idea. During this study, all 5 species of woodpeckers encountered were seen foraging on the material felled during the thinning treatment.

Bird/habitat relationships

Dominance-diversity analysis for control sites illustrates two main points. First, riparian habitat appears to support the most distinct community of any of the four habitat types. This community was the only community not dominated by Chipping sparrows. In addition, many species such as Orange-crowned warblers, Ruby-crowned kinglets, Wilson's warblers, and Swainson's thrushes were more prominent in this community (Table 22) Second, open habitat supported the fewest species of all habitat types. This is not surprising given the limited extent of these areas (i.e., they do not constitute a true grassland habitat and as such, do not support a very diverse community).

Similarity indices (Table 23) also support the distinctness of the riparian and open communities. The riparian community is substantially different from the community found in open habitats. The reduced species richness of the open habitat community was primarily responsible for the low similarity between it and dense forest communities. No communities appeared to be distinct from the overall community structure in treatment sites. This lack of distinctness was likely due to the extreme scarcity of riparian, dense forest and open habitat types.

Thinning altered the forest landscape by changing the distribution of two of the four habitat types. In control sites, dense forest was the most abundant habitat type, whereas in treatment sites, open forest was the most abundant (Table 18). Thinning effectively created isolated islands of dense-unthinned forest surrounded by expansive areas of open canopy forest. It is then possible that bird/habitat associations may have changed in the following ways. If a species uses open canopy forest, this association should be evident in control sites because of the relative rarity of these areas in controls. This association should disappear, or become less apparent, in treatment sites as the availability of this habitat increases. Any species that uses dense-unthinned habitat may undergo similar changes. Because dense-unthinned is the most abundant habitat type in control sites, any strong association with this habitat type may not be apparent (i.e., species distribution would be more uniform). However, in treatment sites dense-unthinned habitat was less abundant. As a result, preference for this habitat type should become apparent there.

Chi-square and discriminant analysis were performed to determine whether this kind of association existed. The uni-variate Chi-square analysis of bird/habitat relationships yielded mostly ambiguous results. However, three species, Vesper sparrows, Northern flickers and Dusky flycatchers appeared to follow the expected pattern. Both Vesper sparrows and Northern flickers were detected in areas with high amounts of open forest in control sites more than was expected. These associations were not apparent in treatment sites where open forest was very abundant (Table 24). Dusky flycatchers, on the other hand, were detected in areas with high amounts of dense forest in treatment sites (Table 24). This association was not present in control sites where dense forest is far more abundant.

Results from discriminant analysis suggest that the changes in the distribution of habitats may have led to the changes in the spatial distribution of three species. In control sites, Chipping sparrows and Vesper sparrows appeared to be most strongly associated with areas that had more open forest characteristics (i.e., areas with little dense forest, and more abundant open forest habitat; Table 26). In treatment sites, however, this association was not noted. Conversely, Yellow-rumped warblers and Dusky flycatchers were found most often in areas with high amounts of dense forest habitat in treatments. This association was not noted in control sites where dense forest habitat was more abundant.

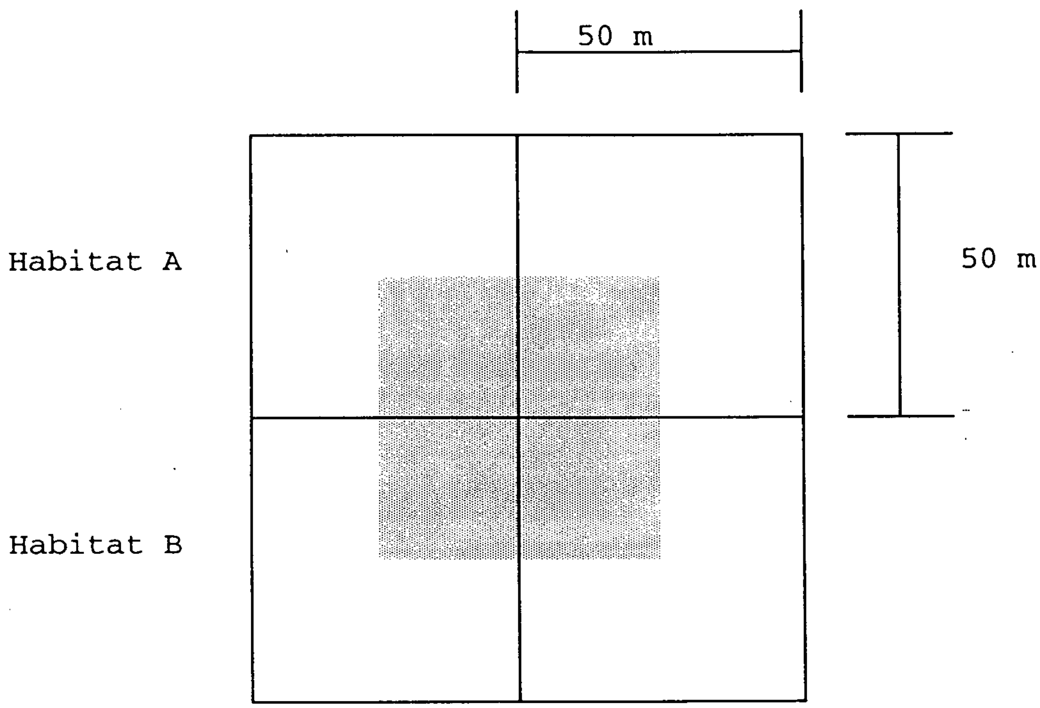
Results the Chi-square analysis further suggest that riparian habitat supported a somewhat distinct community and played a major role in the distribution of some species. Five species (Orange-crowned warbler, Ruby-crowned kinglet, Wilson's warbler, Red-naped sapsuckers and Dusky flycatchers) and one species group (woodpeckers) were detected in riparian habitat more than expected by chance (Table 24). Orange-crowned warblers, Wilson's warblers, and Ruby-crowned kinglets and to a lesser extent Dusky flycatchers are long distance migrant species that are known to be associated with riparian habitat (Brown 1985).

Red-naped sapsuckers and the woodpecker group were also detected more often than expected in riparian habitat (Table 24). Red-naped sapsuckers were often observed feeding on water birch trees that are found only in wetter areas. Red-naped sapsuckers, and the group of woodpeckers, particularly Hairy woodpeckers, and Pileated woodpeckers often nested in Trembling aspen trees (Black-backed and Three-toed woodpeckers were exceptions in this group) resulting in numerous detections in cells with riparian habitat. Despite riparian habitat being extremely rare in both treatments and controls, it appears that even a small amount of riparian habitat can have a positive affect on species diversity. This phenomenon has also been noted by Morrison and Meslow (1983), and Hagar (1990).

It is unclear why Red-naped sapsuckers and woodpeckers were detected in openings more than was expected by chance. Openings usually do not contain very much down and dead material that may have drawn some of the woodpecker species into open areas. What is more likely is that these species were detected in openings when en route to or from their nest sites.

One of the problems associated with the bird/habitat associations involved using 50x50-m cells as the unit of analysis. The use of 50x50-m cells as the unit of analysis contributed to both the low number of cells with a high percentage coverage of any habitat type, and the preponderance of significant Chi-square values in intermediate cover classes. The reason that using a 50x50-m cell as the unit of analysis was problematic is illustrated in Figure 6. In this instance habitat B would cover an entire 50x50-m cell, but because of it's position, is split between four cells each consisting of 25% habitat B and 75% habitat A. Any detections that might be restricted

Figure 6. The potential problem with using a 50x50 m cell as the unit of analysis in bird/habitat relations. Habitat B would cover an entire 50x50 m cell. However, because of the placement of the grid cell, each cell contains 75% of Habitat A and 25% of habitat B.



to habitat B would then be grouped with cells that are made up of low amounts of habitat B and high amounts of habitat A.

Conclusions

Many authors believe that thinning could prove to be a useful tool to manipulate bird species diversity (Dien and Zeveloff 1980, DeGraff et al. 1992, Hunter 1990). This belief is based mainly on the ability to increase microhabitat diversity and within-stand structural diversity. Those goals are achieved by enhancing understory vegetation and developing a shade tolerant, mid-story. Thinning has been an effective tool to increase wildlife diversity when opening the forest canopy has resulted in the development of an understory community that did not already exist (Hagar 1990). Increases in diversity have also been associated with the occurrence of one or more species that are detected infrequently (DeGraff et al. 1992). At present the stands in the IDF xh1 and xh2 are managed on an uneven-aged basis and, as a result, already have a shade tolerant understory. Increased herb and shrub production will likely benefit those species that presently use this resource.

Thinning as conducted in my study sites, unlike some other thinning trials, (e.g. DeGraff et al. 1992) will not likely result in the recruitment of new species into these forests. Long-billed curlews and Mountain bluebirds are two species that could be recruited into these stands. Both were seen on occasion in these stands over the two years of the study. However, both species would require the creation of very large openings. From a silvicultural perspective openings of the size needed for these species would be inappropriate in this forest type. From a wildlife perspective creating large openings for the benefit of these species is also inappropriate as these species are best managed in more open stands at lower elevations.

The main role that thinning will play in these stands will be to alter the relative abundance of species rather that affecting bird species diversity. For this to happen, thinning will likely have to be accompanied by some form of controlled burning and some restrictions or alterations of present grazing patterns. In addition, snag management as it is presently practiced should be continued, but should be accompanied by an effective extension program regarding the role of snags in forests and enforcement of firewood cutting restrictions. Douglas-fir and Ponderosa pine snags in larger diameter classes (those sizes generally used by woodpeckers for nesting and foraging) were rare in both treatments and controls (Table 3). Effective snag management is imperative if these stands are going to be able to support cavity-nesting species in the future.

Results from this study suggest that Vesper sparrows and Chipping sparrows are two species that will potentially benefit from the form of thinning applied. Both species appear to be associated with open canopy forests that are made more abundant as a result of this treatment.

Patches of unthinned forest should continue to be left in thinned sites even if snag management does not necessitate it. Patches of dense-unthinned Douglas-fir were likely always an intricate part of this forest system. Frequent wildfires would have maintained an open parkland forest, but would not completely eradicate all dense patches of Douglas-fir. In addition, results from this study suggest that at least Dusky flycatchers and Yellow-rumped warblers are associated with dense patches of Douglas-fir. In addition to providing an obvious structural diversity, they provide valuable security cover. On one occasion, I witnessed a Pileated woodpecker escaping from the pursuit of a Cooper's hawk by seeking cover in an unthinned patch that was left around a snag.

Slash could be piled, rather than left where it was felled. Piling of slash could benefit some of the ground nesting and feeding species, particularly Dark-eyed juncos.

Results from this study suggest that thinning did not have a significant affect on the forest bird community. Poor understory response to this thinning trial, either because of the affects of cattle grazing, leaving slash on site, ineffective thinning, or the passing of an insufficient period of time, or all four, may explain the negligible response of ground-plus shrub-feeding bird species. In addition, elevated levels of spruce budworm and

Douglas-fir tussock moth may have masked any significant differences in the abundance of foliage-feeding species. These results, however, should be considered as preliminary data and illustrate the need the for long-term monitoring of both vegetation and bird species. Long-term monitoring is necessary particularly if the Thompson/Nicola Mule Deer Foraging and Slashing Project becomes a model for thinning projects in the IDF xh1 and xh2 forest types.

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Appendix I. List of common and scientific names of plants used in text.

Trees

Douglas-fir	*Pseudotsuga menziesii*
Ponderosa pine	*Pinus ponderosa*
Trembling aspen	*Populus tremuloides*
Red alder	*Alnus rubra*
Paper birch	*Betula papyrifera*

Shrubs

red-oiser dogwood	*Cornus stolonifera*
common juniper	*Juniperus communis*
rocky mountain juniper	*Juniperus scopulorum*
rose	*Rosa spp.*
kinnikinik	*Arctostaphylos uva-ursi*
Saskatoon berry	*Amelanchier alnifolia*
sheperdia	*Sheperdia canadensis*
snowberry	*Symphoricarpos albus*
Douglas maple	*Acer glabrum*
spirea	*Spiraea betulifolia*
willow	*Salix spp.*

Herbs

pine grass	*Calamagrostis rubescens*
rough fescue	*Festuca scabrella*
Kentucky bluegrass	*Poa pratensis*
bluebunch wheatgrass	*Agropyron spicatum*
lemonweed	*Lithospermum ruderale*
pussy toes	*Antennaria spp.*
yarrow	*Achillea millefolium*
fireweed	*Epilobium angustifolium*
balsam root	*Balsamorhiza sagittata*
heart-leafed arnica	*Arnica cordifolia*
strawberry	*Fragaria virginiana*
star Solomon's seal	*Smilacina stellata*
twinflower	*Linnaea borealis*
northern bedstraw	*Galium borealis*
horsetail	*Equisetum spp.*

Appendix II. Anova table for vegetation analysis

Source of variation	DF	F-test
Cond	2-1=1	$\sigma^2{}_e + nk\sigma^2{}_r + nk\phi_s$
Plot(Cond)	2*(3-1)=4	$\sigma^2{}_e + nk\sigma^2{}_r$
Error	2*3(20-1)=114	$\sigma^2{}_e$
Total	120-1=119	

Appendix III. List of common and scientific names of birds observed in control and treatment sites.

Red-tailed hawk[a]	*Buteo jamaicensis*
Cooper's hawk[a]	*Accipter cooperii*
Sharpshinned hawk [a]	*Accipter striatus*
American kestrel[a]	*Falco sparverius*
Ruffed grouse[b]	*Bonasa umbellus*
Northern pygmy-owl[b]	*Glaucidium gnoma*
Great-horned owl[c]	*Bubo virginianus*
Common nighthawk[a]	*Chotdeiles minor*
Vaux's swift[a]	*Chaetura vauxi*
Pileated woodpecker[a]	*Dryocopus pileatus*
Black-backed woodpecker[a]	*Picoides arcticus*
Hairy woodpecker[a]	*Picoides villosus*
Northern flicker[a]	*Colaptes auratus*
Red-naped sapsucker[a]	*Sphyrapicus nuchalis*
Three-toed woodpecker[a]	*Picoides tridactylus*
Dusky flycatcher[a]	*Epidonax oberholseri*
Western woodpeewee[c]	*Contopus sordidulus*
Tree swallow[a]	*Tachycineta bicolor*
Gray jay[a]	*Perisoreus canadensis*
Common raven[a]	*Coruvs corax*
American crow[a]	*Coruvs brachyrhynchos*
Black-capped chickadee[a]	*Parus atricapillus*
Mountain chickadee [a]	*Parus gambeli*
Red-breasted nuthatch[a]	*Sitta canadensis*
White-breasted nuthatch[a]	*Sitta carolinensis*
Winter wren[b]	*Troglodytes troglodytes*
Ruby-crowned kinglet[a]	*Regulus calendula*
American robin[a]	*Turdus migratorius*
Swainson's thrush[a]	*Catharus ustulatus*

[a]= found in both controls and treatments

[b]= found only in controls

[c]= found only in treatments

Appendix III continued. List of common and scientific names of birds observed in control and treatment sites.

Mountain bluebird[a]	*Sialia currocoides*
Townsend's solitare[a]	*Myadestes townsendi*
Cedar waxwing[a]	*Bombycilla cedroum*
Solitary vireo[a]	*Vireo solitarius*
Warbling vireo[a]	*Vireo gilvus*
Orange-crowned warbler[a]	*Vermivora celata*
Wilson's warbler[a]	*Wilsonia pusilla*
Tennessee warbler[a]	*Vermivora peregrina*
Yellow-rumped warbler[a]	*Dendroica coronata*
Chipping sparrow[a]	*Spizella passerina*
Dark-eyed junco[a]	*Junco hyemalis*
Vesper sparrow[a]	*Pooecetes gramineus*
Brown-headed cowbird[a]	*Molothrus ater*
Western tanager[a]	*Piranga ludoviciana*
Red crossbill[a]	*Loxia curvirostra*
Pine siskin[a]	*Carduelis pinus*
Cassin's finch[a]	*Carpodacus cassinii*
Evening Grosbeak[a]	*Coccothtaustes Vespertinus*

[a]= found in both controls and treatments

[b]= found only in controls

[c]= found only in treatments