Bird Communities in Interior Douglas-fir Forests in the Cariboo Forest Region

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INTRODUCTION

A large landscape- and stand-level biodiversity study was initiated in 1993 in the Interior Douglas-fir Dry Cool (IDFdk3) Biogeoclimatic Subzone in the Cariboo Forest Region (Dawson and Steen 1993). In the first phase, a modelled natural reference landscape was compared to the current landscape for important characteristics, such as the area of old and interior forest, the type of riparian forest, the amount of forest edge, and patch size by combinations of age, crown closure, and dominant tree species. A GIS analysis of forest cover maps was done over a 1800-km^2 study area. For the second phase, 79 stands were quantified for attributes thought important to biodiversity, such as dead trees, coarse woody debris, stand structure and composition, vertical diversity, and understorey vegetation. The stands were stratified according to harvesting and thinning history, and dominant overstorey species. Since conservation of local biodiversity is an important management objective, the project's third phase assessed how wildlife communities shift in response to forest management activities and tried to anticipate cumulative effects as whole landscapes change. Breeding birds were selected for study because they are easy to identify, occur in large enough populations for statistical analyses, and use many habitat attributes considered important for wildlife.

The Cariboo Forest Region has 895 331 ha of IDFdk3 forest. Because of its habitat attributes and proximity to population centres this forest has high value for timber, wildlife, range, and recreation. The area of old Douglas-fir forest has been substantially reduced on the landscape (Dawson, these proceedings, p. 9) because of forest harvesting. A range of harvesting techniques have been used. Starting in the late 1950s and continuing to the mid-1980s, harvesting was based on cutting all trees over a set diameter limit (i.e., a simulated shelterwood silvicultural system). This practice was replaced in the mid-1980s with partial cutting within a single-tree selection silvicultural system, which retained greater volume across all size classes of trees (Vyse et al. 1990). On mule deer winter ranges, a single-tree selection silvicultural system based on a low-volume partial cutting and an extended cutting cycle was developed to maintain important old forest habitat attributes (Armleder et al. 1986). Some of the diameter-limit and standard single-tree selection stands have been spaced within the last 10 years.

Douglas-fir is the dominant forest type on the landscape. Less common forest types, such as lodgepole pine, spruce, and aspen, may have different bird communities. Currently, lodgepole pine is the lead tree species on about 34% of the IDFdk3 landscape (Dawson, these proceedings). Stands leading in either white spruce or trembling aspen occupy a small area of the IDFdk3 (2%) and are small in size (Dawson, these proceedings). At maturity, these forest types are typically clearcut; sometimes, however, the Douglas-fir component within lodgepole pine stands is reserved from harvest. Another important habitat for birds are Douglas-fir forests adjacent to wetlands and lakes.

No previous work in the IDF forests of Cariboo Forest Region has documented bird species use of various forest types or their response to forest management practices. Furthermore, there is little documentation of the response of bird communities to harvesting and thinning practices in dry Douglas-fir forests in North America. Schwab and Sinclair (1994) compared bird communities found in young seral and mature stands in the IDF forest near Cranbrook, B.C. In the dry interior Douglas-fir/ponderosa pine forests near Princeton and Merritt, B.C., Morgan et al. (1989) compared bird communities in six stands of varying harvesting history over several years. This was followed by a detailed study on the foraging behaviour of several species in the same stands (Morgan et al. 1991). Hejl and Woods (1990) compared old Douglas-fir/ponderosa pine forests to rotational age stands of similar composition in the Northern Rockies; Mannan and Meslow (1984) completed a similar study in northeastern Oregon. In a more detailed study in larch/Douglas-fir forest in northwestern Montana, Tobalske et al. (1991) compared clearcut, partially cut, and old forest bird communities. Some small, nonreplicated studies in the United States have compared old and partially cut forests (Franzreb and Ohmart 1978; Medin 1985; Medin and Booth 1989). Booth (1994) found pre-commercial thinning in Douglas-fir stands in British Columbia's Kamloops Forest Region to have little effect on breeding birds. Although several papers have documented bird species composition in extensive lodgepole pine, aspen, and spruce forests, the use of these forest types when they occur as patches in a matrix of Douglas-fir has not been reported. Little information exists about bird use of Douglas-fir forests adjacent to wetlands or lakes.

For the breeding bird study, 40 Douglas-fir blocks were selected according to harvesting history: old forest, diameter limit (high-volume removal), standard single-tree selection (moderate-volume removal), and mule deer winter range single-tree selection (low-volume removal). To compare forest types, three mature blocks each of lodgepole pine, spruce, and aspen were sampled. To examine species composition in riparian Douglas-fir stands, data were collected from four blocks of different harvesting history adjacent to lakes and wetlands. The specific objectives of this project were:

- to assess the response of breeding bird communities and species to harvesting and thinning practices in Douglas-fir forests;
- to determine species composition of breeding bird communities in aspen, spruce, lodgepole pine, and riparian Douglas-fir forests in the IDFdk3;
- to examine the relationships between habitat attributes and community variables and individual species density through correlation analyses; and
- to evaluate management practices as they affect habitat attributes and consequently bird communities.

The study area was located on the east side of the Fraser River between Williams Lake and 100 Mile House, in the central interior of British Columbia. Within the study area, 53 blocks were situated between 51°39′ and 52°03′N, and 121°0′ and 121°58′W. The topography varied from flat to gently rolling hills with large lakes, and numerous small lakes, marshes, meadows, fens, and grasslands. The elevation ranges between 750 and 1150 m.

The whole study area is within the dry, cool Interior Douglas-fir biogeoclimatic subzone – Fraser variant (IDFdk3) (Steen and Coupé 1997) This is the northern extent of Interior Douglas-fir zone where uneven-aged Douglas-fir exists as the climax forest type. In mesic, old Douglas-fir forests, large trees (> 42.5 cm diameter at breast height), occurring either singly or in small groups (two to six trees), form the main canopy layer, while veterans (> 90 cm dbh and > 250 yr) are scattered throughout the stand. Poles (12.5–42.5 cm) and regeneration (< 12.5 cm) occur in patches. In some cases, regeneration exceeds 10 000 stems per hectare. Aspen (*Populus tremuloides*) and lodgepole pine (*Pinus contorta*) are often scattered throughout Douglas-fir stands.

Lodgepole pine stands collectively occupy 34% of the landscape, while aspen and spruce stands are small (mean size = 14 ha) and rare (2%) (Dawson, these proceedings). Pine stands are usually even-aged and often contain Douglas-fir veterans and regeneration. Hybrid white spruce (*Picea engelmannii* x *glauca*) sometimes mixed with Douglas-fir, is the climax species on the subhygric to hygric sites that commonly occur at the toe of a slope, in poorly drained depressions, or along creeks.

On mesic Douglas-fir and lodgepole pine sites, the shrub layer is sparse (< 10% gound cover), while the lush herb layer is dominated by pinegrass (*Calamagrostis rubescens*) and forbs such as twinflower (*Linnaea borealis*), kinnikinnick (*Arctostaphylos uva-ursi*), and showy aster (*Aster conspicuus*). Feathermosses, especially red-stemmed feathermoss (*Pleurozium schreberi*), are abundant, especially in old forests with high crown closure.

Various harvesting practices and silvicultural systems were used in Douglas-fir forests in the IDFdk3. Before the late 1950s, a "selective" harvesting method was used. The larger, high-quality trees were cut, essentially resulting in high-grading. Crown closure was substantially reduced; however, a scattering of large trees and numerous poles remained in the stand. Starting in the late 1950s and continuing until the mid-1980s, diameter-limit harvesting was formally applied to large areas of forest. Typically, all Douglas-fir trees over 33 cm dbh and other species over 15 cm dbh were cut. The crown closure in most stands was drastically reduced from 66–75% down to 14–35%. In the early 1980s, diameter limits were raised in some prescriptions to conserve some larger trees in the stand. Blocks logged by either of these methods are termed "high-volume removal" (HV) in this study.

From 1984 to present, the standard silvicultural system is single-tree selection using partial cutting. The objective of this system is to produce an all-aged stand with an even distribution of healthy, vigorous trees. Trees are cut from all merchantable size classes and most trees over 90 cm are removed resulting in a 50–60% reduction in volume. Blocks logged using this method are termed "moderate-volume removal" (MV).

A modified single-tree selection silvicultural system is used on mule deer winter ranges (Armleder et al. 1986). Partial cutting is based on the removal of small groups (two to six trees) through the range of diameter classes with emphasis on maintaining a large number of old, large trees because of their superior snow interception and forage value. This silvicultural system results in about a 20% volume reduction per entry and a cutting cycle long enough to replace the volume taken. Blocks logged using this method are termed "low-volume removal" (LV).

The old ("OLD") forest stands used in the study have no harvesting history. They are uneven-aged Douglas-fir forests that contain attributes of old forest such as numerous large trees, snags, and large pieces of coarse woody debris.

Within the Douglas-fir forest type, we selected four riparian blocks with different harvesting histories:

1. an OLD block bordering Big Lake (200 ha);
2. an MV block bordering several small marshy ponds (1–2 ha);
3. an HV block, with a scattering of Douglas-fir veterans, that bordered a small lake (3 ha) with a 10-m buffer; and
4. an HV block adjacent to a small marshy lake (2–3 ha).

Pre-commercial thinning has occurred in eight high-volume and two moderate-volume removal blocks. Blocks were typically thinned to 1800 stems per hectare for Douglas-fir and 1500 stems per hectare for lodgepole pine, for those trees over 1 m in height and below 12 cm dbh. All deciduous over 2 m in height was cut.

METHODS

Experimental Design and Data Analysis

A completely randomized design was used to compare breeding bird community variables and individual species between three harvesting treatments and unlogged controls in Douglas-fir forests. We randomly selected 11 old unlogged blocks, 6 low-volume removal, single-tree selection blocks, 10 moderate-volume removal, single-tree selection blocks, and 13 high-volume removal, diameter-limit or selectively logged blocks. For community variables and 13 normally distributed individual species, one-way analysis of variance and *a priori* contrasts were used to compare treatments. Results were considered significant at $\alpha = 0.05$ and all nonsignificant results were tested for power. The four *a priori* contrasts were:

1. The high-volume and moderate-volume removal treatments are used significantly differently from the low-volume removal and old treatments;
2. The old treatment is used differently from the other three treatments;
3. A linear pattern exists in the treatment means where richness, diversity, and number of observations are highest in the high-volume removal treatment, lower in the moderate-volume removal treatment, second lowest in the low-volume removal treatment, and lowest in the old treatments; and
4. A nonlinear pattern exists in the treatment means where richness, diversity, and number of observations are highest in the high-volume removal treatment, lower in the moderate-volume removal treatment, and equally low in the low-volume removal and old treatments.

Student *t*-tests were used to compare use of spaced and unspaced blocks within the moderate- and high-volume removal treatments ($\alpha = 0.05$). Nonparametric one-way analysis of variance was used to compare data for eight species that were not normally distributed. For another 14 species with 5–16 observations, chi-square tests were used to compare the observed and expected values for each treatment. Sixteen species had fewer than five observations so statistical analyses were not performed.

To assess the importance of other forest types and riparian Douglas-fir forests to breeding birds, we selected three blocks each of mature aspen (> 80 years), mature spruce (> 140 years), and mature pine (> 80 years), as well as four riparian Douglas-fir blocks. In the riparian blocks, the point-count stations were placed so that the 75-m plot perimeter abutted the edge of either a small marsh, pond, or lake. The bird communities and individual species abundance in pine, aspen, spruce, and riparian Douglas-fir forests were compared using 95% confidence intervals.

The relationships between 11 habitat attributes (eight stand and three landscape) and 25 bird variables (relative density of 21 bird species and 4 community variables) were assessed through Pearson correlation analyses. These analyses were based on the 40 upland blocks. Four riparian blocks were included only for correlations with distance to wetland/open water and distance to open water. Bird species with fewer than 14 observations were not included in these analyses.

Data Collection

Bird Surveys Point-count surveys (Bond 1957) were used to sample bird communities in 1994. Two point-count stations (75 m radius) were set up in each block. Every block was 25 ha or larger, and located within 100 m of an accessible road. There was a minimum distance of 200 m between station centres and the outer perimeter of each station was at least 50 m from treatment edges. These were surveyed three times between May 9 and May 27 between dawn and 9:00 am. Survey times and observers were changed for each survey of each block. The observer waited 2 minutes before recording all visual and aural observations of birds within a 12-minute interval. Distance to and type of behaviour were recorded for each observation within and beyond the 75 m radius plot.

Species richness (number of species), total number of territorial observations (singing, drumming, feeding young, and sitting on nests), total number of observations (based on all behaviours except moving high above the habitat), and diversity (Shannon Index) were calculated for each block. Diversity was based on the mean number of observations of three surveys (all behaviours within a 75 m radius plot) for each species. Observations for pine siskins, white-winged crossbills, and red crossbills were not included when calculating diversity and total number of observations because large flocks, observed periodically, caused the data to be abnormally distributed. Relative density of individual species was based on the mean number of observations of three survey dates for each block (two 75 m radius plots pooled).

Habitat Attributes The following habitat attributes were measured in each block:

- crown closure for each of three diameter classes;
- basal area and diameter of live trees;
- density of aspen saplings;
- density and decay class of snags;
- volume and decay class of coarse woody debris; and
- percent cover in each vegetation layer.

In almost all blocks, the variables were measured within the 75 m radius point-count stations. In addition, three landscape-level variables were measured:

1. percentage of forest within a 1 km radius of the stand;
2. distance to open water; and
3. distance to a wetland or open water.

The crown-closure data were used to calculate a canopy diversity index (CDI). This index is a measure of live canopy volume, which incorporates both vertical and horizontal forest structure. The method used was similar to Spies and Cohen's (1992) except that the proportion of canopy closure in three canopy layers was measured directly rather than calculating crown closure from diameter-crown width relationships. The three canopy layers used to calculate the index were saplings (dbh ≤ 12.5 cm), poles (dbh 12.6–41.5 cm), and large trees (dbh > 41.5 cm).

Basal area was measured by either of two methods: belt transects or fixed-area plots. In a detailed forest structure project, one 180-m transect was used per block. Transect width varied according to the three size classes of trees: 2 m for saplings, 4 m for poles, and 8 m for large trees. The dbh of all trees within the specified transect width were measured. Several stands that were surveyed for birds were not included in the detailed forest structure project. In these stands, basal area of poles and large trees were measured in six plots using a prism (BAF = 6 or 8). The basal area of saplings (dbh < 12.6 cm) was measured in six 2 m radius plots per stand.

The density of aspen saplings and snags of all species were recorded in either the belt transects or prism plots for each stand. Snags were divided into large tree and pole size classes for analysis. Snag decay was classified according to the B.C. Wildlife Tree Committee guidelines (Backhouse 1993).

The volume of coarse woody debris was measured using two 75-m equilateral triangles per stand (Luttmerding et al. 1990). Total volume included: all pieces over 10 cm diameter lying on the forest floor or standing at less than a 45° incline to the horizontal, and decay classes 1–5 (Backhouse 1993).

Vegetation was sampled in six 2-m radius plots per stand. Percent cover was estimated for five vegetation layers: moss (moss and terrestrial lichens), grass (grasses and sedges), forbs, and shrubs (height layer B1 < 2.1 m), and shrub (height layer B2 > 2.0 m).

The three landscape variables were measured from the midpoint between the two point-count stations used for bird surveys in each stand. A dot grid superimposed on a 1:20 000 scale forest cover map was used to measure the percentage of area covered by old and low-volume partially cut Douglas-fir forest (> 140 yr) within a 1000 m radius (OLDPK). The same maps were used to measure distance to permanently open water (pond, lake, or creek).

Distance to water was broken into three categories of WET (1 = > 1000 m, 2 = 500–1000 m, 3 = < 500 m). The third variable, WETALL, is based on the same three categories of distance to either wetland or open water.

RESULTS

Bird Communities in Douglas-fir Forest Types

A total of 82 species were detected during this study in 1994. Of these, 51 species were located in the upland Douglas-fir forests within the 75 m radius point-count stations. A further 16 species were recorded only in point-count stations within riparian Douglas-fir, aspen, spruce, and pine forests. The remaining 15 species were recorded when the listening distance was extended as far as the observer could hear. Detections of special interest include one record of the red-listed western grebe and 32 records of the blue-listed sandhill crane.

Species richness, species diversity, total observations, and territorial observations were used to compare bird communities within the four treatment types (HV, MV, LV, and OLD). All one-way parametric analyses of variance established significant ($\alpha = 0.05$) differences between the treatment means (Table 1). Results from the contrast statements show that the high- and moderate-volume removal treatment types have significantly greater diversity, species richness, and number of observations than either of the low-volume removal and old treatment types. A nonlinear pattern is evident in the treatment means where richness, diversity, and number of observations are highest in the high-volume removal treatment, lower in the moderate-volume removal treatment, and lowest in either the low-volume removal or old treatments.

Community variables did not differ significantly ($\alpha = 0.05$) between pre-commercially thinned and unthinned blocks within either the high- and moderate-volume removal treatments using parametric *t*-tests. However, all the nonsignificant results lacked power; therefore, more extensive sampling is required to adequately measure the influence of pre-commercial thinning.

In upland old, and low- and moderate-volume removal stands, the most locally common birds in order of decreasing abundance were yellow-rumped warblers, red-breasted nuthatches, dark-eyed juncos, and solitary vireos. However, in high-volume removal stands the order shifted to yellow-rumped warblers, dark-eyed juncos, chipping sparrows, and then red-breasted nuthatches.

Table 2 summarizes the chi-square test results for low-density species in the four treatments in upland Douglas-fir forests. The critical value for chi-

TABLE 1 *Comparison of the bird community variables in the four treatments found in upland Douglas-fir forests (IDFdk3) using analysis of variance*

	Treatment (mean)							
Variable	**HV ($n = 13$)**	**MV ($n = 10$)**	**LV ($n = 6$)**	**OLD ($n = 11$)**	**Df**	**Sums of squares**	***F***	***p***
Species richness	18.9	17.8	13.8	14.8	3	54.84	6.27	.0016
Species diversity	2.52	2.47	2.15	2.14	3	1.23	6.38	.0014
Total observations	20.7	17.2	12.4	13.9	3	410.06	9.27	.0001
Territorial observations	17.8	15.2	13.8	10.8	3	364.68	11.23	.0001

TABLE 2 *Chi-square results for low-density species found in the four treatments in the upland Douglas-fir forests within the IDFdk3 subzone. Asterisks indicate species that show significant (α = 0.05) differences in relative abundance among the treatments.*

		Treatment						
Species	**Type of count**	**HV (*n* = 13)**	**MV (*n* = 10)**	**LV (*n* = 6)**	**OLD (*n* = 11)**	**Chi-square value**	***p***	**Power**
Solitary sandpiper	Obs.	0	2	0	3	4.75	.19	.42
	Exp.	1.6	1.3	0.8	1.4			
Hairy woodpecker	Obs.	2	1	1	2	0.25	.97	0
	Exp.	2.0	1.5	0.9	1.7			
Pileated woodpecker*	Obs.	0	1	0	6	12.27	.01	.85
	Exp.	2.3	1.8	1.1	1.9			
Olive-sided flycatcher*	Obs.	5	0	0	0	10.38	.02	.78
	Exp.	1.6	1.3	0.8	1.4			
Western wood peewee	Obs.	3	2	0	0	3.74	.29	.34
	Exp.	1.6	1.3	0.8	1.4			
Gray jay	Obs.	3	3	4	3	2.62	.45	.24
	Exp.	4.2	3.3	2.0	3.6			
Common raven	Obs.	4	3	4	1	4.29	.23	.38
	Exp.	3.9	3.0	1.8	1.3			
Swainson's thrush	Obs.	2	4	2	5	1.90	.59	.18
	Exp.	4.2	3.3	2.0	3.6			
Townsend's warbler	Obs.	2	4	2	2	1.75	.63	.17
	Exp.	3.3	2.5	1.5	2.8			
Wilson's warbler	Obs.	6	2	0	2	4.13	.25	.37
	Exp.	3.3	2.5	1.5	2.8			
Vesper sparrow*	Obs.	6	0	0	0	12.46	.01	.85
	Exp.	2.0	1.5	0.9	1.7			
White-winged crossbill*	Obs.	14	0	10	24	22.09	.01	.99
	Exp.	15.6	12.0	7.2	13.2			

square ($\alpha = 0.05$ and $df = 3$) was 7.81. Pileated woodpeckers were found more than expected in the old stands and less than expected in the high-volume removal stands. White-winged crossbills were also detected more frequently than expected in the old type; however, individuals were concentrated in small flocks. Both olive-sided flycatchers and vesper sparrows were recorded exclusively in high-volume removal treatment blocks. Comparisons were not significant for the following species: solitary sandpiper, hairy woodpecker, western wood peewee, gray jay, common raven, Swainson's thrush, Townsend's warbler, and Wilson's warbler. Power was low ($1 - \beta < 0.80$) for all nonsignificant results.

For the eight non-normally distributed species (red-naped sapsucker, northern flicker, black-capped chickadee, mountain chickadee, ruby-crowned kinglet, hermit thrush, western tanager, and brown-headed cowbird), the nonparametric analyses of variance were not significant ($\alpha = 0.05$ and $df = 3$) (Table 3).

Table 4 summarizes the results of the parametric analyses of variance for 13 species. We found that dusky flycatchers, Townsend's solitaires, orange-crowned warblers, and chipping sparrows used the high- and moderate-volume removal treatments significantly ($\alpha = 0.05$) more than low-volume

TABLE 3 *Comparison of relative densities of birds found in the four treatments in upland Douglas-fir forests in the IDFdk3 subzone, using a one-way nonparametric ANOVA. Asterisks indicate species that show significant (α = 0.05) differences in relative abundance among the treatments.*

Species	Mean score HV (*n* = 13)	MV (*n* = 10)	LV (*n* = 6)	OLD (*n* = 11)	Kruskal-Wallis (Chi-square approx)	*p*
Red-naped sapsucker	22.57	21.50	23.00	15.77	3.15	.37
Northern flicker*	25.46	21.05	15.00	17.14	7.53	.06
Black-capped chickadee*	18.03	27.05	14.08	20.95	6.76	.08
Mountain chickadee	23.92	18.85	16.33	20.23	2.34	.50
Ruby-crowned kinglet*	26.08	20.30	18.95	11.58	7.37	.06
Hermit thrush	19.23	24.05	20.00	19.04	1.33	.72
Western tanager	17.73	22.10	20.25	22.45	1.40	.71
Brown-headed cowbird*	25.46	22.55	16.67	14.86	7.12	.07

removal or old treatments. On the other hand, red-breasted nuthatches and golden-crowned kinglets were more frequently detected in old treatment stands than in the three other treatments. Power analysis was used to test for a 30% and a 50% maximum difference in the treatment means for all non-significant species. Power was low ($1 - \beta < 0.80$) for all the nonsignificant species except for yellow-rumped warblers. This indicates that a study should be conducted with larger sample sizes before the null hypothesis is accepted.

Although the analysis of variance models were nonsignificant for American robins and dark-eyed juncos, several *a priori* contrasts were significant for each of these species. American robins and dark-eyed juncos were significantly more abundant in stands that had been logged by any method compared to old stands. The relative abundance of dark-eyed juncos also increased in a linear way with increased volume removal. All the contrasts were significant for chipping sparrows and dusky flycatchers, which preferred

TABLE 4 *Comparison of relative densities of birds found in the four treatments in upland Douglas-fir forests in the IDFdk3 subzone, using a one-way parametric ANOVA. Asterisks indicate species that show significant (α = 0.05) differences in relative abundance among the treatments.*

Species	Treatment mean HV (n = 13)	MV (n = 10)	LV (n = 6)	OLD (n = 11)	Df	Sums of squares	F	p
Ruffed grouse	0.82	0.93	0.44	0.52	3	0.09	1.12	.3531
Dusky flycatcher*	0.85	0.63	0.11	0.03	3	5.00	5.49	.0033
Red-breasted nuthatch*	1.72	1.77	1.89	2.91	3	10.47	3.49	.0341
Golden-crowned kinglet*	0.05	0.10	0.06	0.36	3	0.71	3.55	.0239
Townsend's solitaire*	0.92	0.77	0.22	0.39	3	4.46	3.86	.0171
American robin	0.82	0.87	0.78	0.42	3	1.33	1.86	.1539
Solitary vireo	1.13	1.17	1.50	0.97	3	1.10	0.68	.5678
Orange-crowned warbler*	1.10	0.27	0.06	0.06	3	8.47	8.41	.0002
Yellow-rumped warbler	3.00	2.87	2.00	2.97	3	4.72	1.67	.1909
Chipping sparrow*	2.38	1.07	0.44	0.24	3	1.14	16.37	.0001
Dark-eyed junco	2.69	2.20	2.28	1.45	3	9.26	2.58	.0682
Red crossbill	5.64	3.07	9.53	3.15	3	0.68	1.79	.1665
Pine siskin	4.54	3.03	7.56	4.24	3	78.23	1.81	.1630

more open stands. Also, both Townsend's solitaires and orange-crowned warblers preferred more open stands, and relative density increased with increased volume removal. In contrast, the relative density of both red-breasted nuthatches and golden-crowned kinglets was greatest in old treatment stands compared to any of the harvesting treatments, and the treatment means decreased linearly with increased volume removal. Results of all the contrast tests were not significant ($\alpha = 0.05$) for red-crossbills, pine siskins, solitairy vireos, yellow-rumped warblers, and ruffed grouse.

Ruby-crowned kinglet was the only species that showed a significant ($t = 2.89$, $df = 11$, $p = 0.01$) preference for spaced stands compared to unspaced stands in the high-volume removal treatment.

Bird Communities in Other Forest Types

Riparian Douglas-fir In riparian Douglas-fir blocks, species diversity was significantly ($\alpha = 0.05$) greater than any of the upland lodgepole pine and Douglas-fir, but not different from either spruce or aspen types (Figure 1a). Species richness was significantly greater than that found in low-volume removal and old treatments, or pine stands (Figure 1b). Furthermore, both territorial observations and total observations were significantly higher than in moderate- and low-volume removal treatments, the old treatment, and pine type (Figure 1c). In the riparian type, ruby-crowned kinglets were most frequently detected followed by red-breasted nuthatches, yellow-rumped warblers, and dark-eyed juncos. Many species were unique to the riparian type including Canada goose, Barrow's golden eye, bufflehead, greater yellowlegs, spotted sandpiper, common snipe, clay-coloured sparrow, savannah sparrow, red-winged blackbird, and yellow-headed blackbird. Compared to upland Douglas-fir types, the riparian type generally had greater relative densities of ruffed grouse, red-naped sapsuckers, dusky flycatchers, olive-sided flycatchers, tree swallows, violet-green swallows, black-capped chickadees, marsh wrens, ruby-crowned kinglets, American robins, warbling vireos, rusty blackbirds, and song sparrows. Species that had noticeably lower detections in the riparian type include: red crossbills, pine siskins, solitary vireos, Townsend's solitaires, and mountain chickadees.

Lodgepole Pine Bird diversity, richness, and abundance in the lodgepole pine type were similar to that of low-volume removal and old Douglas-fir treatments, but significantly ($\alpha = 0.05$) lower than in the high- and moderate-volume removal treatments, and the riparian and spruce stands (Figure 1a, 1b, 1c). In lodgepole pine, we found that the most frequently observed species occurred in similar proportions to those found in low-volume removal and old Douglas-fir treatments; however, woodpeckers, black-capped chickadees, golden-crowned kinglets, Swainson's thrushes, and Townsend's warblers were not detected in this forest type and the numbers of red crossbills, pine siskins, solitary vireos, and western tanagers were quite low. The only observation of Clark's nutcracker occurred in this forest type.

Spruce Species richness, diversity, and territorial observations were significantly ($\alpha = 0.05$) greater within the spruce type than in the low-volume removal and old treatments (Figure 1a, 1b, 1c). Furthermore, total observations were significantly higher in spruce forests than within the moderate- and low-volume removal treatments and the old treatment Douglas-fir forests (Figure 1d). In the spruce type, ruby-crowned kinglets were most frequently detected followed by red-breasted nuthatches, yellow-rumped warblers, dark-eyed juncos, and golden-crowned kinglets. The winter wren

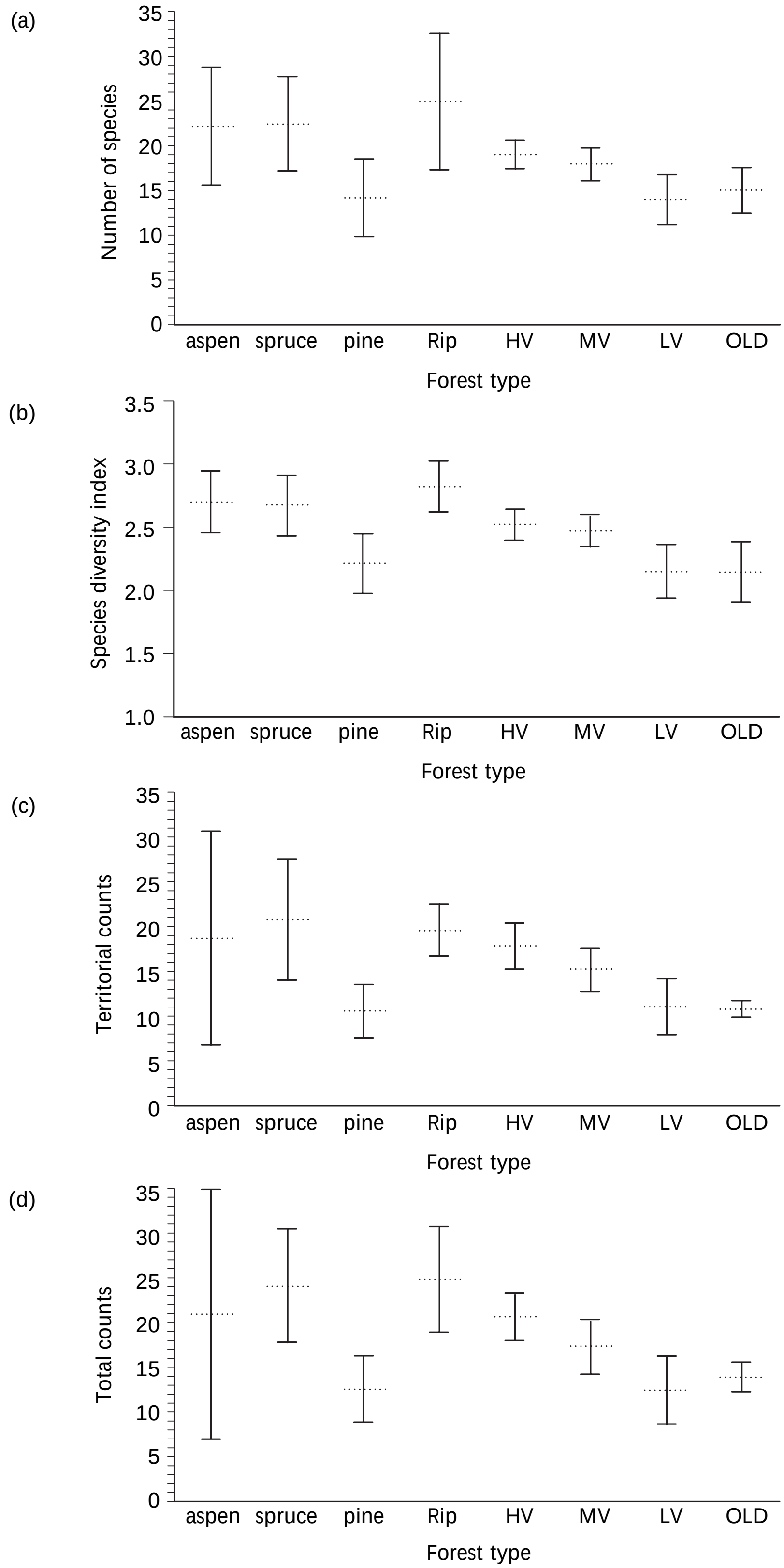

FIGURE 1 *Mean and 95% confidence interval for (a) species richness, (b) species diversity, (c) territorial observations, and (d) total observations of birds in aspen, spruce, lodgepole pine, and Douglas-fir types in the IDFdk3 subzone in the Cariboo Forest Region.*

was observed only in this forest type. Compared to the upland Douglas-fir forests, the following species were locally abundant: Lincoln's sparrow, Wilson's warbler, Townsend's warbler, ruby-crowned kinglet, golden-crowned kinglet, black-capped chickadee, Hammond's flycatcher, dusky flycatcher, three-toed woodpecker, red-naped sapsucker, and ruffed grouse. Species that were noticeably less frequently detected included red crossbills, pine siskins, brown-headed cowbirds, and hermit thrushes.

Aspen Species richness and diversity were significantly ($\alpha = 0.05$) greater within the aspen type than in the low-volume removal and old Douglas-fir treatments and in the pine type (Figure 1a, 1b). Although the mean values for territorial and total observation data were similar to riparian and spruce, they were highly variable. The aspen bird communities were distinct from the other forest types in species composition—the most abundant species were dusky flycatchers, ruby-crowned kinglets, orange-crowned warblers, and warbling vireos. Fox sparrows were found only in the aspen type. Compared to upland Douglas-fir forests, the following species were more frequently detected: ruffed grouse, rufous hummingbird, red-naped sapsucker, dusky flycatcher, Hammond's flycatcher, black-capped chickadee, ruby-crowned kinglet, American robin, warbling vireo, orange-crowned warbler, northern waterthrush, and Wilson's warbler. Species that were comparatively low in relative density include: red crossbills, pine siskins, solitary vireos, yellow-rumped warblers, and dark-eyed juncos.

Habitat Variables Eight stand variables were compared between the four Douglas-fir treatments (Table 5). Both the canopy diversity index (CDI) and basal area of large trees were highest in the old and low-volume removal treatments, low in the moderate-volume removal treatment, and extremely low in the high-volume removal treatment. Also, the basal area of large poles in old and low-volume removal treatments was almost double that in high- or moderate-volume removal treatments. The mean basal area of saplings was similar among the treatments, but highly variable within treatments. Spacing of the sapling layer within high- and moderate-volume removal treatments reduced the mean basal area from 8.9m^2/ha to 3.4m^2/ha. When aspen saplings were separated from the general sapling category, the density was very low in both low-volume removal and old treatments compared to high- and moderate-volume removal treatments. Within the high- and moderate-volume removal treatments, aspen density was greater in spaced blocks ($x = 480$ stems per hectare, SD = 581, $n = 10$) than in unspaced blocks ($x = 161$ stems per hectare, SD = 242, $n = 13$). The mean density of tree-sized snags did not vary greatly between treatments. However, the mean density of pole-sized snags tended to be greater and the 95% confidence interval much wider in old and low-volume removal treatments than in high- and moderate-volume removal treatments. The total volume of coarse woody debris did not differ significantly between the treatments. The percent cover of shrubs was consistently low between the treatments. Both grasses and forbs were more abundant in high- and moderate-volume removal treatments, while moss was more common in old and low-volume removal treatments.

Correlation and Regression of Habitat Attributes and Birds

Correlation coefficients were calculated for 21 bird species, and four bird community variables with eight stand attributes and three landscape attributes (Table 6). The habitat variables most frequently, positively or negatively, correlated with bird variables were basal area of large trees and percent of old

TABLE 5 *Summary of stand attributes measured in treatments surveyed for birds in the IDFdk3 subzone in 1994*

	HV (n = 13)		MV (n = 10)		LV (n = 6)		OLD (n = 11)	
Variable	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD	$\bar{x}$	SD
Canopy diversity index	1.06	0.43	1.43	0.35	2.33	0.42	2.38	0.51
Basal area (tree) (m^2/ha)	0.37	0.77	6.63	2.51	13.20	4.39	16.86	7.43
Basal area (pole) (m^2/ha)	10.83	5.02	9.51	3.05	16.31	3.49	16.27	7.91
Basal area (sapling) (m^2/ha)	5.41	4.42	7.93	3.82	7.87	5.95	8.99	3.66
Snags (tree) (stems per hectare)	6.84	10.55	4.81	7.34	7.31	7.56	10.44	9.80
Snags (poles) (stems per hectare)	14.60	13.41	9.72	13.18	27.93	32.65	29.94	27.71
Coarse woody debris (m^3/ha)	143.40	87.95	111.86	88.37	70.28	50.53	115.20	68.81
Aspen (saplings) (stems per hectare)	368.57	543.93	211.66	264.71	37.04	90.72	7.58	25.13

forest within a 1 km radius. The bird species were separated into seven groups based on their relationship with these two variables.

Red-breasted nuthatches and golden-crowned kinglets were the only species positively correlated with the basal area of large trees. At a landscape level, three species were positively correlated with quantity of old forest: red crossbills, pine siskins, and red-breasted nuthatches.

Dark-eyed juncos and brown-headed cowbirds were negatively correlated with basal area of large trees, while mountain chickadees were negatively correlated with the canopy diversity index. The four community variables and several species were negatively correlated with both basal area of large trees and the amount of old forest. These include: ruby-crowned kinglet, chipping sparrow, dusky flycatcher, Townsend's solitaire, and orange-crowned warbler. Although hermit thrushes, ruffed grouse, and yellow-rumped warblers were not correlated with basal area of large trees, they were negatively correlated with the amount of old forest.

We found no relationship between either American robins or black-capped chickadees and basal area or the amount of old forest, although they did correlate with other variables. Four species did not correlate with any of the habitat variables: northern flicker, red-naped sapsucker, solitary vireo, and western tananger.

Many species, including golden-crowned kinglet, ruby-crowned kinglet, dusky flycatcher, ruffed grouse, American robin, black-capped chickadee, and the four community variables were positively correlated to distance to water or wetland. The closest stands to water (< 500 m) had the greatest relative density of individuals by species and highest values for the community variables. No significant negative relationships existed between the bird variables and distance to water or wetland.

Density of aspen saplings was positively related to a number of species—ruby crowned kinglet, chipping sparrow, Townsend's solitaire, hermit thrush, ruffed grouse, yellow-rumped warbler—and the territorial observa-

TABLE 6 *Summary of p values for Pearson's correlation coefficients between habitat attributes and relative density of bird species and community variables in the IDFdk3 subzone (+/- = 0.1–0.05; ++/– – = 0.04–0.009, +++/– – – = < 0.009)*

Species	CDI $n = 40$	BA (tree) (m^2/ha) $n = 40$	BA (pole) (m^2/ha) $n = 40$	BA (sapling) (m^2/ha) $n = 40$	Aspen (stems/ha) $n = 40$	Snag (tree) (stems/ha)) $n = 40$	Snag (pole) (stems/ha)) $n = 40$	CWD (m^3/ha) $n = 40$	Wetall $n = 44$	Wet $n = 44$	Oldpk $n = 40$
Golden-crowned kinglet	++	+++							++		
Red-breasted nuthatch	++	++									+
Pine siskin											+++
Red crossbill											+++
Dark-eyed junco		---									
Mountain chickadee	-										
Brown-headed cowbird		--						++			
Ruby-crowned kinglet	--	--		--	++			++	+++	+++	--
Chipping sparrow	---	---		--	+++						--
Dusky flycatcher	---	---	-	--						+	---
Orange-crowned warbler	---	---									--
Townsend's solitaire	--	--	--		+	+		+++			--
Hermit thrush					++						-
Ruffed grouse			-		+++				+	+++	--
Yellow-rumped warbler					+						-
American robin										++	
Black-capped chickadee							+		++	+++	
Northern flicker											
Red-naped sapsucker											
Solitary vireo											
Western tanager											
Diversity	--	---	-					++	++	+++	---
Species richness	--	---	--					+++	+++	+++	---
Total observations	---	---	-		+++			++	++	+++	---
Territorial observations	---	---	-	-	+++					+++	---

tions and total observations. No significant negative relationships existed with this variable.

Basal area of poles was negatively related to density of dusky flycatchers, Townsend's solitaires, and ruffed grouse, as well as the four community variables. Ruby-crowned kinglets, chipping sparrows, dusky flycatchers, and territorial observations were negatively correlated to the basal area of saplings.

Coarse woody debris was positively correlated with diversity, richness, total observations, ruby-crowned kinglet, Townsend's solitaire, and brown-headed cowbird. Tree-sized snags were positively correlated only with Townsend's solitaires, while pole-sized snags were positively correlated only with black-capped chickadees.

DISCUSSION

As forest stand structure and attributes are manipulated through harvesting, changes occur in the breeding bird community. Low-volume selection harvesting caused the least change in the composition of the breeding bird community compared to old forest. Moderate- and high-volume removals resulted in increased richness, abundance, and diversity compared to old or lightly harvested sites. This is driven by 1) shifts in species abundance, and 2) new species being able to use the changed habitat. For example: 1) species that are common in old forest, such as the dark-eyed junco and Townsend's solitaire, are even more abundant in moderate- and high-volume removal stands; and 2) species such as the chipping sparrow, dusky flycatcher, and orange-crowned warbler, which are sparse in old or lightly harvested forests, are found in high numbers in the moderate-volume removal stands and in especially high numbers in high-volume removal stands. Similar patterns of increase occurred with less common species such as the northern flicker and brown-headed cowbird. Diameter-limit harvesting has caused the most radical change in forest structure from that found in old forests. This dramatic shift was also reflected in the number of species unique to the high-volume removal blocks: McGillivray's warbler, olive-sided flycatcher, tree sparrow, vesper sparrow, LeConte's sparrow, song sparrow, and Lincoln's sparrow. Tree sparrows and LeConte's sparrow are considered casual or rare, respectively, in the Cariboo Region (Roberts and Gebauer 1992); however, McGillivray's warbler, vesper sparrow, song sparrow, and Lincoln's sparrow are commonly associated with open or brushy habitat types. Although olive-sided flycatchers were recorded in high-volume removal blocks, they were usually located along hard edges between old and the diameter-limit cut forests. The juxtaposition of these two habitats may be ideal for this aerial insect feeder. In northeastern Oregon, Mannan and Meslow (1984) compared bird communities in 85-year-old thinned stands with 200-year-old forests. They found that increased numbers of birds occurred in the younger stands because of increases in chipping sparrow, dusky flycatcher, and ruby-crowned kinglet populations. Morgan et al. (1989) concluded that neither diameter-limit nor single-tree selection harvesting drastically harmed the bird community, although several mature forest species declined. This study, however, was poorly replicated in space.

The greatest management concern is for species that are unique to old forest or that respond negatively to harvesting. The only records for Cooper's hawk and brown creeper occurred in old Douglas-fir forests compared to the harvested stands. Cooper's hawks are uncommon throughout the Cariboo Region (Roberts and Gebauer 1992) and prefer open or patchy forests (Cannings et al. 1987). Cooper's hawk probably can use forest types other than old Douglas-fir. In this study, brown creeper was recorded once in old Douglas-fir and once in mature lodgepole pine forest. This species, often associated with old forest, is more of a management concern in the moister, higher-elevation forests in the Cariboo Region where it is found more frequently. Both red-breasted nuthatches and golden-crowned kinglets were significantly less abundant in the three harvesting treatments compared to unlogged controls. A literature comparison reported in Medin (1985) shows that both these species decrease in abundance following partial cutting. Medin and Booth (1989) report a significant decline in red-breasted nuthatches after a 29% volume reduction using a single-tree selection system. In our study, both of these species are positively correlated to the basal area of large trees and the canopy diversity index. The red-breasted nuthatch is a secondary cavity nester and forages by gleaning insects from the bark of trees. Morgan et al. (1991) report that 70% of nuthatch foraging was spent between 5 and 20 m above ground in the thickest, tallest trees. The golden-crown kinglet forages by gleaning insects from foliage while perched or hovering (Morgan et al. 1991). Both of these species may be responding to the decreased amount of foraging habitat resulting from harvesting. Furthermore, red-breasted nuthatch abundance was positively correlated with the amount of old forest within 1 km of each stand, which suggests a possible sensitivity to old forest fragmentation.

Pileated woodpeckers and white-winged crossbills show preference for old forests. A number of studies document pileated woodpeckers as preferring late successional stages of forests because they contain large-diameter trees used for foraging, roosting, and nesting, and large coarse woody debris used for foraging (Bull and Jackson 1995). Based on studies from the United States, home range size varies from 53 to 1056 ha. Given that the average old-forest stand size in the IDF in the Cariboo Forest Region is 30 ha (Dawson, these proceedings), woodpeckers are probably integrating several stand types into their home range. Currently, the number of dead, large trees and amount of coarse woody debris in harvested stands is similar to old stands; however, live, large trees (> 41.5 cm dbh) are half as abundant in moderate-volume removal stands and very infrequent in high-volume removal stands. Carpenter ants, a major forage species, are often found in live, large trees as well as snags in the IDF of the Cariboo Region.

During our bird surveys, conifer seed eaters, such as the white-winged crossbill, red crossbill, and pine siskin, were locally very abundant in May 1994. This is attributed to a huge cone crop across the Cariboo Region in the fall of 1993. At the stand level, small flocks of white-winged crossbills occurred most frequently in low-volume removal and old stands. This might correlate with the greater abundance of seed produced in stands with more larger, older trees (Benkman 1993). Although red crossbill and pine siskin relative abundance did not differ significantly between the harvesting treatments, at the landscape level both were positively correlated with the amount of old Douglas-fir within the area. These species may concentrate in areas where foraging success is higher. Benkman (1993) concluded that old forests support more red crossbills than young forests and as the proportion of old forest in the landscape declines, crossbills will decline disproportionately.

Cowbird brood parasitism and its influence on declines in interior forest bird populations is a concern raised in the literature (O'Conner and Faaborg

1992; Smith 1994). Although not statistically significant, brown-headed cowbirds were more frequently encountered in high- and moderate-volume removal stands, and a strong negative correlation existed between them and the basal area of large trees (Table 6). Morgan et al. (1989) also found more cowbirds on sites that were opened up by harvesting. Their greater abundance may be related to more open habitat structure, which facilitates foraging and finding host nests, or it may be a response to the greater abundance of hosts, and their nests. In either case, as the portion of old forest continues to decline and patch size decreases across the landscape, the negative effect of this species on others should be considered.

Within high-volume removal treatments, spacing reduced the basal area in the small size class of trees (< 12.5 cm dbh), but increased the density of aspen saplings. Similar to Booth (1994), this study found that spacing did not affect most bird species. Ruby-crowned kinglet was the only species that showed preference for spaced stands. Correlation analysis in this study found negative relationships between ruby-crowned kinglet density and basal area both of small trees and large trees, but positive relationships with aspen density and distance to water. Mannan and Meslow (1984) found that this species preferred more open, thinned forests compared to closed, old forests. Booth (1994) found that ruby-crowned kinglets were not affected by spacing, but responded strongly to the presence of riparian habitat within unspaced control areas.

Wetlands and lakes of varying sizes are numerous in the IDF of the Cariboo Region (Steen and Roberts 1988), with about 6% in wetlands and meadows, and 3% in open water in the IDFdk3 landscape (Dawson, these proceedings). Despite the limited sampling in Douglas-fir forest adjacent to wetlands or lakes in our study, we found many species unique to the riparian stands and a high level of biodiversity as indicated by species richness, diversity, and abundance indices compared to upland Douglas-fir forest.
The abundance of several species, including ruby-crowned kinglets, orange-crowned warblers, dusky flycatchers, golden-crowned kinglets, ruffed grouse, American robins, and black-capped chickadees, was strongly positively correlated with distance to a wetland or lake. The adjacent forests provide important habitat for cavity-nesting ducks: common goldeneye, Barrow's goldeneye, bufflehead, wood duck, common merganser, and hooded merganser. The blue-listed sandhill crane was heard frequently in this study, often near small wetlands and lakes. Adult sandhill cranes with young use the adjacent forested areas for hiding cover, and possibly for resting and feeding (Cooper 1996).

Correlation analyses assessed the relationship of certain habitat attributes to the breeding bird community variables and the relative density of individual species. The community variables were negatively correlated with basal area, canopy diversity index, and the amount of old forest. These relationships were largely driven by species that prefer more open habitat. Species that were more abundant in high- and moderate-volume removal treatments were the same ones that were negatively correlated with basal area of the various size classes of trees, canopy diversity index, and area of old forest. Strong positive correlations existed between the community variables and the amount of coarse woody debris, aspen density, and distance to water. Again, individual species preferences are driving these relationships. For some species, the reasons for the positive relationships are clear. For example, ruffed grouse prefer stands of aspen poles for nesting, the male-bud flowers on mature aspen as forage, and dense stands of saplings for wintering, brooding, and breeding (McCaffery et al. 1997). The causation of other positive relationships is obscure. For example, three species—brown-headed cowbird, ruby-crowned kinglet, and Townsend's solitaire—were positively related to coarse woody debris volume, but do not typically use this attribute directly. It is possible that coarse woody debris is correlated with other desirable habitat variables. Distance to water was related positively to species abundance; however, many causes may drive these relationships, such as forage abundance or habitat change.

Of the five cavity-nesting species that were detected frequently enough to correlate with density of tree- and pole-sized snags, only black-capped chickadees were positively correlated to the density of poles. The relative densities of mountain chickadees, red-breasted nuthatches, northern flickers, and red-naped sapsuckers were not related to snag density. The mean and variance for density of tree-sized snags were similar between the old forest and the harvesting treatments. The mean density of pole-sized snags was greater in old and low-volume removal stands than in the high-volume or moderate-volume removal stands, but a great deal of variability was evident. A sufficient abundance of snags for cavity-nesting species may now exist; however, repeated entries into forests for harvesting and spacing will reduce snag density. Another consideration is that species select habitat based on a combination of variables, not just one. Multiple linear regression (best combinations method) was attempted in this study, but the resulting r^2 values were quite low and difficult to interpret.

Within the IDFdk3 in the Cariboo Forest Region, stands dominated by lodgepole pine, aspen, or spruce occur. Lodgepole pine stands cover about 34% of the land area. These stands are usually large—65% of the pine areas consist of stands over 80 ha in size (Dawson, these proceedings). When lodgepole pine forests were compared to Douglas-fir forests, the breeding bird community variables were similar; however, several species occurred at lower abundance. Pure lodgepole pine forests found in the Montane Spruce and Sub-boreal Pine–Spruce biogeoclimatic zones in the Cariboo Forest Region have very low mean species diversity ($\overline{x} = 1.42$), richness ($\overline{x} = 7.13$), and abundance ($\overline{x} = 5.5$) compared to the pine-dominated forests in the IDFdk3 (Waterhouse 1995). The greater abundance and diversity in the IDFdk3 lodgepole pine forests might well be attributed to the presence of Douglas-fir veterans that are often scattered throughout stands and Douglas-fir regeneration in the understorey. Hein (1980) stated that species diversity is higher in forest types mixed with pine rather than in pure lodgepole pine.

Spruce and aspen stands occupy a small area of the IDF zone (about 2% each) and are small in size. Their high species diversity, richness, and abundance, as well as the unusual mixture of bird species, is a response to the different vegetation composition and stand structure. The assemblage of species is similar to those recorded for various seral stages found within the Engelmann Spruce–Subalpine Fir zone in the Cariboo Region (M.J. Waterhouse, unpublished data).

MANAGEMENT IMPLICATIONS

In the Cariboo Forest Region, only 2% of the IDFdk3 subzone has protected status (parks, ecological reserves). The rest of the forest falls into the enhanced resource management zone under the Cariboo-Chilcotin Land Use

Plan; therefore, most of the land base is available for forest harvesting. To date, it has been heavily affected by forest harvesting. The old Douglas-fir forest, which dominated the pre-industrial landscape, has been significantly reduced and fragmented. The current management practice in Douglas-fir forests is the single-tree selection silvicultural system. Partial cutting removes 50% of the initial stand volume and the subsequent volume that accrues over each 30-year period. About 16% of the IDFdk3 has been mapped as mule deer winter range, and the Douglas-fir portions of the winter ranges are managed through a low-volume single-tree selection system that maintains greater basal area on large trees through time.

While the density of some species either increases or stays the same, declines in the abundance of other species have occurred and species composition has shifted relative to old forests as a consequence of forest harvesting. The low-volume single-tree selection silvicultural system had the least influence on the structure and composition of the bird communities typical of old IDF forests, although both red-breasted nuthatches and golden-crowned kinglets declined. It should be considered a management option if the goal is to maintain natural levels of breeding bird diversity outside of mule deer winter range. However, the effect of repeated entries into stands using this silvicultural system or the standard single-tree selection silvicultural system has not been assessed. Simulated shelterwoods (based on low minimum diameter) or clearcutting silvicultural systems, especially when planted to pine, are poor options if the goal is to conserve bird communities similar to those found in older, uneven-aged Douglas-fir forests.

Pre-commercial thinning is required post-harvest for all blocks that exceed maximum density standards. Breeding birds did not respond to the spacing done in either the single-tree selection or diameter-limit cutblocks. In fact, spacing increased the density of aspen saplings, which was strongly positively correlated with the density of several species. In the long term, aspen will provide valuable habitat for cavity-nesting species, depending on size of the trees and degree of heartrot in them. This is especially important as the number of coniferous snags decline because of repeated harvesting and thinning entries. Before fire control was instituted, old stands were frequently burned and probably far less stocked in the regeneration layers than those on the landscape today. In the absence of fire, spacing can be used to create a more open regeneration layer. In the course of spacing, care should be taken to maintain wildlife trees, large deciduous trees, and unspaced patches or strips. Spacing can also be used as a tool to create stand structure in heavily cut stands and to retain the natural clumpy stand structure similar to that found in old Douglas-fir forests.

Currently, the number of wildlife trees and volume of coarse woody debris do not differ greatly between the various harvesting treatments and old forests. They have been cut only once and many were cut before the strict Workers' Compensation Board rules regulating the falling of dead trees were instituted and before rigorous utilization standards were in place. However, these attributes are expected to decrease over time as stands become intensively managed. Of the 82 bird species detected in this study, 18 are cavity nesters and at least another seven were known to occur IDFdk3. Many of these 25 species occur at low density or have large territories, making it difficult to directly measure their response to changes in habitat.

While spruce and aspen stands occupy a small portion of the landscape, they provide ideal habitat for some species and required habitat for other species. These stand types should be maintained through the landscape because of their high biodiversity value. Lodgepole pine forests occur extensively throughout the IDFdk3. The old Douglas-fir trees scattered through the stands as well as the Douglas-fir regeneration should be retained, since conversion to pure lodgepole pine would probably result in decreased bird use. Douglas-fir forests adjacent to wetlands and lakes are valuable habitat for many species. Sampling in riparian habitats was very limited in this study, so we could not directly compare the effect of different silvicultural systems on breeding birds. However, silviculture prescriptions for forests near wetlands and lakes should emphasize maintenance of wildlife trees and forest cover.

Old forests in the current IDFdk3 landscape are already quite fragmented compared to the pre-industrial landscape. Red crossbills, pine siskins, and red-breasted nuthatches appear to prefer areas where a high percent of old forest is present. Low-density species, such as barred owls and pileated woodpeckers, have been or could be affected as landscape becomes more fragmented. It is important to meet the seral-stage and patch-size distributions recommended in the *Biodiversity Guidebook* (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1995), where a minimum amount of the landscape unit should be left as old forest, depending on biodiversity emphasis. It is also important that the landscape be managed for some large areas of old forest. Some potential management options include: aggregation of blocks, higher-volume removals on smaller areas, and greater use of low-volume selection silvicultural systems.

Results from this study strongly support the recommendations for stand structure and species composition for Natural Disturbance Type 4 (NDT4) as described in the *Biodiversity Guidebook* (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1995). Specifically, the guidebook addresses the maintenance of rare ecosystems, uneven-aged stand structure in Douglas-fir forests, Douglas-fir in pine forests, and a broadleaf component. Furthermore, establishing wildlife tree patches in stands under selection management by designating leave trees or group reserves is important. Group reserves are necessary in most cases to maintain snags in the stand, especially since uneven-aged managed stands will be entered frequently for harvesting and thinning. Group reserves (wildlife tree patches) could be shifted at each stand entry if the wildlife trees have fallen and ones of equal value are available. Recruitment of equivalent wildlife trees will likely become more difficult after the first few entries unless the time between entries is quite long. Wildlife tree patches placed near wetlands or lakes accrue extra benefits to wildlife. This is especially important for small wetlands and lakes that do not have riparian reserve or management zones. Large, live trees could be maintained as wildlife trees by specifying a number per hectare to be left or a fixed maximum diameter limit for harvesting in the silviculture prescription.

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Armleder, H.M., R.J. Dawson, and R.N. Thomson. 1986. Handbook for timber and mule deer management coordination on mule deer winter ranges in the Cariboo Forest Region. B.C. Ministry of Forests, Victoria, B.C. Land Management Handbook No. 13.

Backhouse, F. 1993. Wildlife tree management in British Columbia. Co-published by: Workers' Compensation Board, B.C. Ministry of Forests Silviculture Branch, and Canada-British Columbia Partnership Agreement on Forest Resource Development, Victoria, B.C.

Benkman, C.W. 1993. Logging, conifers, and the conservation of crossbills. Conservation Biology 3:473–9.

Bond, R.R. 1957. Ecological distribution of breeding birds in upland forests of southern Wisconsin. Ecological Monograph 27:351–84.

Booth, B. 1994. The effects of thinning on forest bird communities in dry interior Douglas-fir forests. MSc thesis. University of British Columbia, Vancouver, B.C.

British Columbia Ministry of Forests and B.C. Ministry of Environment, Lands and Parks. 1995. Biodiversity guidebook. B.C. Forest Practices Code. Victoria, B.C.

Bull, E.L. and J.A. Jackson. 1995. Pileated woodpecker *Dryocopus pileatus. In* The Birds of North America, No. 148. A. Poole and F. Gill (editors). The Academy of Natural Sciences, Philadelphia, Penn., and The American Ornithologists' Union, Washington, D.C.

Cannings, R.A, R.J. Cannings, and S.G. Cannings. 1987. Birds of the Okanagan Valley, British Columbia. Royal British Columbia Museum, Victoria, B.C.

Catts, D.J. 1991. Bird communities and forest succession in the subalpine zone of Kootenay National Park, British Columbia. MSc thesis. Simon Fraser University, Vancouver, B.C.

Cooper, J.M. 1996. Status of the sandhill crane in British Columbia. B.C. Ministry of Environment, Lands and Parks, Wildlife Branch, Victoria, B.C. Wildlife Bulletin No. B-83.

Dawson, R. and O. Steen. 1993. Natural and managed disturbance patterns and their implications for biodiversity management in the IDF zone. B.C. Ministry of Forests, Cariboo Forest Region, Research Section, Draft Working Plan.

Franzreb, K.E. and R.D. Ohmart. 1978. The effects of timber harvesting on breeding birds in a mixed conifer forest. Condor 80:431–41.

Hein, D. 1980. Management of lodgepole pine for birds. *In* Management of western forests and grasslands for non-game birds. Workshop Proceedings, Feb. 11–14, 1980. Salt Lake City, Utah. U.S. Department of Agriculture Forest Service, Ogden, Utah. Technical Report INT-86.

Hejl, S.J. and R.E. Woods. 1990. Bird assemblages in old-growth and rotation-aged Douglas-fir/ponderosa pine stands in the northern Rocky Mountains: a preliminary assessment. *In* Interior Douglas-fir: the species and its management. Symposium in Spokane, Wash., Feb. 27–Mar. 1, 1990. Washington State University, Pullman, Wash.

Luttmerding, H.A., D.A. DeMarchi, E.C. Lea, D.V. Meidinger, and T. Vold. 1990. Describing ecosystems in the field. 2nd edition. B.C. Ministry Environment and Ministry of Forests, Victoria, B.C. B.C. Ministry of Environment Manual No. 11.

McCaffery, K.R., J.E. Ashbrenner, W.A. Creed, and B.E. Kohn. 1997. The aspen-ruffed grouse connection. Journal of Forestry 95(8): 16–20.

Mannan, R.W. and E.C. Meslow. 1984. Bird populations and vegetation characteristics in managed and old-growth forests, Northeastern Oregon. Journal of Wildlife Management 48:1219–38.

Medin, D.E. 1985. Breeding bird response to diameter-cut logging in west-central Idaho. U.S. Department of Agriculture Forest Service, Ogden, Utah. Research Paper INT-355.

Medin, D.E. and G.D. Booth. 1989. Responses of birds and small mammals to single tree selection logging in Idaho. U.S. Department of Agriculture Forest Service, Ogden, Utah. Research Paper INT-408.

Morgan, K.H., J.P. Savard, and S.P. Wetmore. 1991. Foraging behaviour of forest birds of the dry interior Douglas-fir, ponderosa pine forests of British Columbia. Canadian Wildlife Service, Pacific and Yukon Region, Victoria, B.C. Technical Report Series No. 149.

Morgan, K.H., S.P. Wetmore, G.E.J. Smith, and R.E. Keller. 1989. Relationships between logging methods, habitat structure, and bird communities of dry interior Douglas-fir, ponderosa pine forests of British Columbia. Canadian Wildlife Service, Pacific and Yukon Region, Victoria, B.C. Technical Report Series No. 71.

O'Conner, R.J. and J. Faaborg. 1992. The relative abundance of the brown-headed cowbird (*Molothrus ater*) in relation to exterior and interior edges in forests of Missouri. Transactions, Missouri Academy of Science 26:1–9.

Roberts, A. and M. Gebauer. 1992. Checklist of Cariboo Birds. Williams Lake Field Naturalists, Williams Lake, B.C.

Schwab, F.E. and A.R.E. Sinclair. 1994. Biodiversity of diurnal breeding bird communities related to succession in the dry Douglas-fir forests of southeastern British Columbia. Canadian Journal of Forestry Research 24:2034–40.

Smith, J. 1994. Conservation villains or convenient scapegoats? Birding 26(4): 257–9.

Spies, T. and W.B. Cohen. 1992. An index of canopy height diversity. Corvallis, Oreg. C.O.P.E. Report 5(3).

Steen, O.A. and R.A. Coupé. 1997. A field guide to forest site identification and interpretation for the Cariboo Forest Region. B.C. Ministry of Forests, Victoria, B.C. Land Management Handbook No. 39.

Steen, O. and A. Roberts. 1988. Guide to wetland ecosystems of the very dry Montane Interior Douglas-fir subzone (IDFb2) in the Cariboo Forest Region, British Columbia. B.C. Ministry of Forests, Victoria, B.C. Land Management Report No. 55.

Tobalske, B.W., R.C. Shearer, and R.L. Hutto. 1991. Bird populations in logged and unlogged western larch/Douglas-fir forest in Northwestern Montana. U.S. Department of Agriculture Forest Service, Ogden, Utah. Research Paper INT-442.

Vyse, A., R.A. Smith, and B.G. Bondar. 1990. Management of interior Douglas-fir stands in British Columbia: past, present and future. *In* Interior Douglas-fir: the species and its management. Symposium in Spokane, Wash, Feb. 27–Mar. 1, 1990. Washington State University, Pullman, Wash.

Waterhouse, M.J. 1995. The breeding bird community in mature lodgepole pine forests in the Msxv and SBPSxc in the Cariboo Forest Region: a summary report. B.C. Ministry of Forests, Cariboo Forest Region, Research Section, Williams Lake, B.C. Unpublished draft.

The Ecological Role of Fire for Cavity-nesting Birds in Dry Forest Ecosystems

MARLENE MACHMER

INTRODUCTION

Dry, low-elevation forests in the Rocky Mountain Trench were historically characterized by high-frequency, low-intensity fires. Estimated fire return intervals of 10–20 years resulted in open stands of large ponderosa pine, Douglas-fir, and western larch in association with fire-adapted plant communities and wildlife species. Long-term fire suppression has resulted in forest ingrowth and associated changes to stand structure, plant and animal species composition, successional processes, and forest health.

The effects of reduced fire frequency on fire-adapted wildlife species have not been well studied (Kremsater et al. 1994). The proportion of cavity-nesting species is comparatively high in fire-maintained (Natural Disturbance Type 4) forests (Bunnell 1995). Anecdotal evidence suggests that bark-foraging birds, and woodpeckers (e.g., black-backed, three-toed, and hairy woodpeckers) in particular, are closely associated with recently burned forest (Hutto 1995). These species require standing dead or defective live trees for cavity nesting and roosting, as well as for feeding (Backhouse and Lousier 1991; Steeger et al. 1995). Bark and wood boring beetles, which are highly attracted to trees scorched or killed by fire, form a large component of the diet of some cavity-nesting species (see review in Machmer and Steeger 1995). The temporal and spatial occurrence of burns is therefore expected to have a strong influence on the population dynamics of this wildlife guild.

The quantitative relationship between forest burns, insectivorous cavity-nesting birds, their habitat, and their prey is currently being investigated by Pandion Ecological Research Ltd. Funding is being provided through the Forest Renewal BC Research Program (FRBC no. KB96092-RE). This 5-year study (1996–2000) is being conducted in the Tata range unit of the Cranbrook Forest District. Harvesting/burning trials planned for the site in conjunction with the East Kootenay Trench Ecosystem Restoration Program permit collection of pre- and post-treatment data under several operational treatment scenarios.

Four treatments are being tested:

1. burn only (one replicate),
2. harvest only (one replicate),
3. harvest and burn (one replicate), and
4. control (two replicates).

Harvesting treatments (uniform shelterwood with reserves/final overstorey removal silvicultural system) were completed in February 1997 and burning is scheduled for spring of 1998.

Pre- and post-treatment response variables being monitored in 20-ha core sampling areas within each treatment unit include:

- the relative abundance of bark and wood boring beetles;
- the nesting density and relative abundance of cavity-nesting species;
- the feeding intensity of bark-foraging birds; and
- the characteristics, availability, and use of wildlife trees.

Relative Abundance of Bark and Woodboring Beetles

Standard forest insect/disease surveys combined with insect trapping (funnel/sticky) using standard woodborer bait are being conducted in all treatment units. Pre-treatment data collection indicates that all forest health factors are of low to moderate severity. Insect trap results are pending.

Nesting Density and Relative Abundance of Cavity-nesting Species

Systematic searches for cavity and open nests, and fixed radius (75 m) point-count surveys are being used to measure the nesting density and relative abundance of all bird species, respectively. In each 20-ha treatment unit, six nest searches and six point-count surveys (five point-count stations per treatment unit) were conducted during the bird breeding season (May–June).

A total of 59 active nests (31 cavity and 28 open nests) of 13 species were found during intensive nest searches. The number and density of nests per treatment is summarized in Table 1. Overall, pre-treatment nesting densities were of similar magnitude in the treatments, ranging from 0.4 to 0.7 active nests per hectare. Nesting species richness was also comparable (i.e., a range of 3–4 cavity-nesting species and 2–5 open-nesting species per treatment unit; Table 2). Based on point-count data, the relative abundance, richness, and diversity of species were also highly comparable among treatment units.

TABLE 1 *Pre-treatment cavity and open nest densities per hectare by treatment unit (the total number of nests found per core sampling area is shown in parentheses)*

Treatment unit	Cavity nests	Open nests	All nests
Harvest and burn	0.15 (3)	0.35 (7)	0.50 (10)
Harvest only	0.40 (8)	0.30 (6)	0.70 (14)
Burn only	0.25 (5)	0.40 (8)	0.65 (13)
Miller control	0.50 (10)	0.20 (4)	0.70 (14)
Lost Dog control	0.25 (5)	0.15 (3)	0.40 (8)
All sites	0.31 (31)	0.28 (28)	0.59 (59)

TABLE 2 *Cavity- and open-nesting species richness (n) detected in treatment units*

Treatment unit	Cavity (n)	Open (n)	Species code[a]
Harvest and burn	3	3	1, 2, 4, 9, 10, 12
Harvest only	4	5	2, 3, 4, 7, 9, 10, 11, 12, 13
Burn only	3	2	2, 4, 7, 9, 10
Miller control	3	3	2, 4, 7, 8, 11, 12
Lost Dog control	4	3	2, 5, 6, 7, 9, 11, 12

a Cavity nesters: 1 = northern flicker; 2 = red-breasted nuthatch; 3 = white-breasted nuthatch; 4 = mountain chickadee; 5 = black-capped chickadee; 6 = hairy woodpecker; 7 = red squirrel.
Open nesters: 8 = red-tailed hawk; 9 = chipping sparrow; 10 = dusky flycatcher; 11 = solitary vireo; 12 = western tanager; 13 = yellow-rumped warbler.

Feeding Intensity of Bark-foraging Birds

Fresh woodpecker foraging sign (e.g., foraging excavations, scaling, and sapsucking) is being assessed and quantified on trees according to Steeger and Machmer (1995). Observations of actively foraging woodpeckers are being conducted opportunistically in each treatment unit to supplement foraging sign data.

Of 991 trees assessed in random plots, only 64 (6.5%) trees had fresh woodpecker foraging sign in the form of scaling (78.1%), excavations (20.3%), or sapsucking (1.6%). This relatively low level of feeding intensity is attributed to the apparent scarcity of insect host trees within the treatment units. Observations were gathered on six woodpecker species actively feeding in treatments.

Characteristics, Availability, and Use of Wildlife Trees

Trees measuring greater than 10 cm dbh in 11.28 m circular plots spaced at 50 m intervals (one plot per hectare) along a uniform grid within core sampling areas are being assessed for:

- number, species, diameter, height, and decay class;
- presence and number of fresh/old nesting and roosting cavities and foraging sign; and
- presence of tree defects and mortality/disturbance agents (e.g., fire scars, insects, and root, stem, and foliage diseases).

Wildlife trees (i.e., defective live trees or standing dead trees) comprised 171 of 991 (17.2%) trees in the random plots. Pre-treatment wildlife tree densities averaged 42.5 trees per hectare with the following breakdown by species:

- Yellow pine: 46.8%
- Douglas-fir: 25.1%
- Lodgepole pine: 19.9%
- Western larch: 5.9%
- Trembling aspen: 2.3%

Mean diameter and height of wildlife trees averaged 22.6 cm and 11.6 m, respectively, with a median decay class of 2 (B.C. Wildlife Tree Committee 1997). Scarring (fire, lightning, or wildlife damage: total 59%) and insects (wood borers, pitch moths, turpentine beetles, bark beetles: total 23%) were the most common forest health agents affecting wildlife trees.

Pre-burn treatment and post-harvest treatment data collection continued during summer 1997.

REFERENCES

Backhouse, F. and J.D. Lousier. 1991. Silviculture systems research: wildlife tree problem analysis. B.C. Ministry of Forests, B.C. Ministry of Environment, Lands and Parks, Wildlife Tree Committee, Victoria, B.C.

British Columbia Wildlife Tree Committee. 1997. Wildlife/danger tree assessor's course workbook. B.C. Ministry of Forests, B.C. Ministry of Environment, Lands and Parks, and Workers' Compensation Board, Victoria, B.C.

Bunnell, F.R. 1995. Forest-dwelling vertebrate faunas and natural fire regimes in British Columbia: patterns and implications for conservation. Conservation Biology 9:636–44.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountains (U.S.A.) conifer forests. Conservation Biology 9:1041–58.

Kremsater, L., W. Klenner, and D. O'Brien 1994. The effects of fire and fire suppression in the forest types of the Kamloops Region. B.C. Ministry of Forests, Kamloops, B.C.

Machmer, M. and C. Steeger. 1995. The ecological roles of wildlife tree users in forest ecosystems. B.C. Ministry of Forests, Victoria, B.C. Land Management Handbook No. 35.

Steeger, C. and M. Machmer. 1995. Wildlife trees and their use by cavity nesters in selected stands of the Nelson Forest Region. B.C. Ministry of Forests, Nelson, B.C. Technical Report No. TR–010.

Steeger, C., M. Machmer, and E. Walters. 1995. Ecology and management of woodpeckers and wildlife trees in British Columbia. Environment Canada, Delta, B.C. B.C. Wildlife Fact Sheet.

Ecology and Management of Douglas-fir at the Northern Limit of Its Range: Problem Analysis and Interim Management Strategy

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WINNIFRED KESSLER, AND DAN LOUSIER

STATUS OF THE RESOURCE

Douglas-fir is a species requiring specific attention to ensure that it is maintained as an ecosystem component in the Prince George Timber Supply Area (TSA). Douglas-fir forest types comprise approximately 8% of the harvestable land base and Douglas-fir makes up approximately 4% of the cut in the Prince George TSA (Jull 1997). Douglas-fir is widely scattered over the landscape, and is the dominant species adjacent to many of the larger lakes and in areas with significant topographic relief and rich soil types. The age class distribution of Douglas-fir is relatively well balanced from 41 to 250 years of age (Table 1).

A mid-aged cohort (41–120 years) currently exists and, if properly managed, will develop all the necessary attributes of old-growth stands. However, in the long term, insufficient immature and regenerating stands (0–41 years) are present to maintain natural levels of this species or its functional role in the landscape. This lack of regenerating and immature stands is likely a result of both fire suppression and stand conversion from Douglas-fir to other species after sites are logged. Some instances of stand conversion have resulted

TABLE 2 *Prince George TSA Douglas-fir inventory using 1989–1994 data* [a]

	Douglas-fir forest cover (ha)[b]							
Age class	**Vanderhoof**		**Fort St. James**[c]		**Prince George**[c]		**All districts**	
All veterans	1346	(4.8)	167	(0.4)	4842	(2.2)	6354	(2.2)
Leading mature (7+)	3564	(12.8)	11270	(26.2)	20868	(9.6)	35702	(12.4)
Leading immature (3–6)	2006	(7.2)	6411	(14.9)	11611	(5.4)	20028	(7.0)
Leading seral (1,2)	518	(1.9)	173	(0.4)	4740	(2.2)	5431	(1.9)
Second-growth mature (7+)	7621	(27.4)	11430	(26.6)	93773	(43.3)	11282	(39.3)
Second-growth immature (3–6)	10752	(38.7)	9622	(22.4)	56213	(26.0)	76587	(26.7)
Second-growth seral (1,2)	1967	(7.1)	3878	(9.0)	24527	(11.3)	30372	(10.6)
Total area of Fdi (ha)	27772		42950		216575		287298	
Percent of total Fdi by district		(9.7)		(14.9)		(75.3)		—

a The data for this table were derived from the interior Douglas-fir study conducted in 1994 for the Prince George TSA. The inventory data used for the 1994 study were updated between 1989 and 1994.
b Percentages appear in parentheses.
c Does not include TFL 42 (FSJ) or TFL 30 (PG).

in changes in the dominance patterns of the stands; for example, in stands where Douglas-fir was previously the dominant species, second-growth stands now exist that contain Douglas-fir as a secondary or tertiary species.

Values and Significance of Douglas-fir and Douglas-fir Ecosystems

Douglas-fir has a key ecological role in these northern ecosystems. Relative to other northern species, Douglas-fir can grow to tremendous sizes, lives a very long time, and has a very slow rate of decay, both while standing and as coarse woody debris. These key attributes help to maintain aboveground biodiversity of animal species and contribute significant amounts of soil wood which is necessary for long-term soil productivity and health (Franklin et al. 1981; Franklin and Spies 1991). Douglas-fir's role in enhancing soil productivity is perhaps its most significant contribution to the landscape because a rich, healthy soil produces forest components that contribute to a rich biological diversity and high-value timber products (R. Graham, Research Forester, U.S. Department of Agriculture Forest Service, Ogden, Utah, pers. comm., 1997).

Douglas-fir has been identified as a species to retain and maintain on the landscape in all higher-level plans that govern the TSA. Douglas-fir is often associated with critical species habitats in this area, including mule deer, bats, garter snakes, and bushy-tailed woodrats (J. Vinnedge, Forest Ecosystem Specialist, B.C. Ministry of Forests, Fort St. James, pers. comm., 1996). Wildlife habitat was inventoried and measurements were gathered for coarse woody debris, the number and size distribution of snags, and usage levels of various wildlife species in an attempt to quantify these relationships. Douglas-fir is also a major component of a blue-listed plant community found in the Sub-boreal Spruce Wet Cool (SBSwk3) subzone.

Douglas-fir has a high profile in local land-use planning initiatives because of its high recreation, scenic, and ecological values. Within the scientific community, the Douglas-fir in this area commands a high level of interest because it is at the edge of its range and therefore has a high degree of genetic plasticity. Economically, Douglas-fir contributes approximately $12 million per year in stumpage revenues in the Prince George TSA, in addition to the revenues generated by industry in selling the timber. Partial-cutting strategies, which are used to maintain non-timber values in Douglas-fir stands, result in additional costs (L. Huffman, Woodlands Manager, Apollo Forest Products, pers. comm., 1996). The stumpage system does not respond to these costs, markets for Douglas-fir tend to be highly variable, and many mills do not use Douglas-fir. For these reasons, it is not a preferred species in some parts of the TSA. With past management practices, the additional costs from plantation failure and the extensive time periods required to reach free growing were seen as disincentives to establishing Douglas-fir.

Plantation data gathered on sites ranging from 5 to 22 years old indicate that Douglas-fir often grows more slowly than lodgepole pine (*Pinus contorta* var. *latifolia*) in the first 15 years. Figure 1 illustrates site productivity and height growth data for Douglas-fir gathered as part of the problem analysis. The data indicate that on sites that have been broadcast burned, Douglas-fir can reach the 3-m green-up height at 11 years, while taking 13 years on mechanically prepared sites (generally disc trenched), and 19 years on unimproved sites. Improved nursery culture for Douglas-fir combined with the appropriate choice of regeneration strategies can result in thriving Douglas-fir regeneration and reduced time to green-up.

Published site index species conversions for lodgepole pine to Douglas-fir indicate that Douglas-fir has a lower site index at 50 years than pine across all

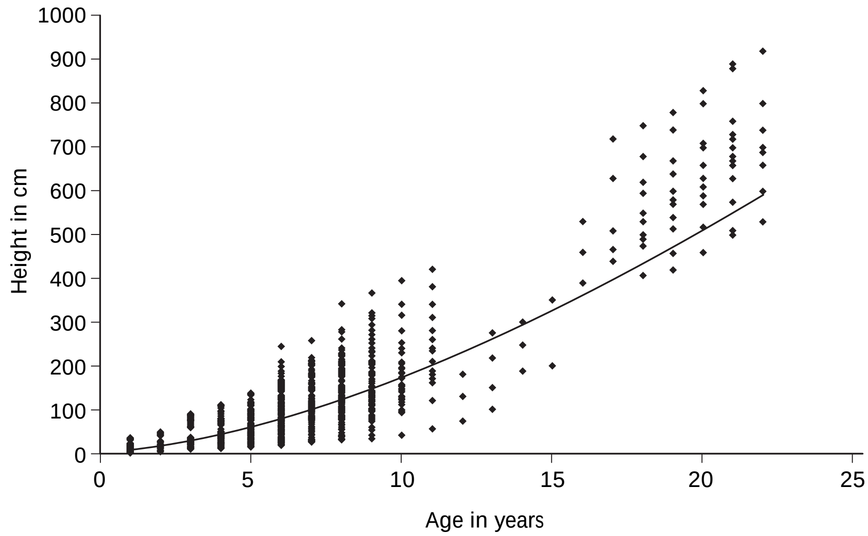

FIGURE 1 *Douglas-fir growth from plantation establishment to free growing (1–22 years).*

site indices (Nigh 1994). However, only three samples in this study are from subzones found in the study area, with none coming from the wettest of the subzones (P. Martin, Stand Development Forester, B.C. Ministry of Forests, pers. comm., 1996). Our data (Figure 2) show a trend toward an increasing site index at 50 years by site series as we move north into the wetter subzones. This trend suggests that the published correlations do not reflect the productivity potential for Douglas-fir in these northern subzones. Indeed, Figure 3 shows the reverse of what is expected for species conversion of lodgepole pine to Douglas-fir. More data are required to confirm these trends in these northern ecosystems.

Growth and yield curves for Douglas-fir indicate that this species is more responsive to site variables than either pine or spruce (*Picea glauca* x *engel-*

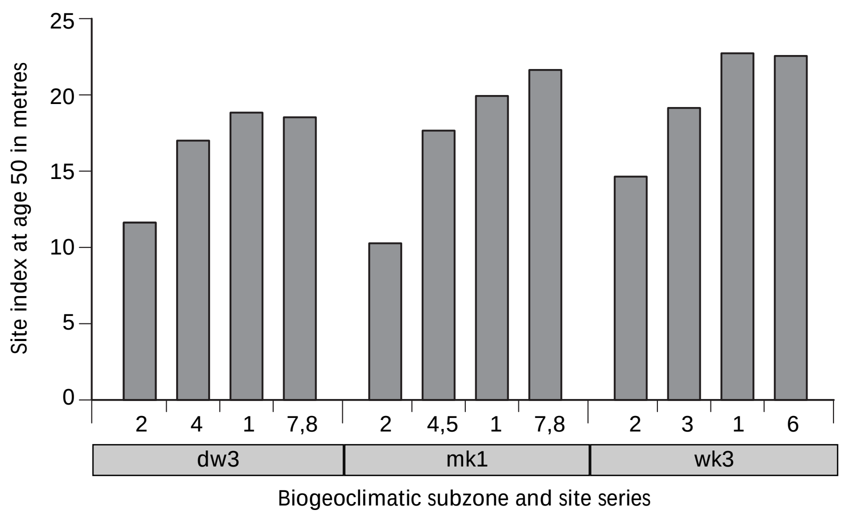

FIGURE 2 *Comparison of site index of Douglas-fir across the SBSdw3, SBSmk1, and SBSwk3.*

mannii). On sites where the Douglas-fir site index at 50 years is greater than 15 m, Douglas-fir exceeds the growth potential of pine at all rotation ages greater than 50 years, and for spruce at all rotation ages greater than 80 years (Thrower et al. 1991). This is illustrated graphically in Figure 3. Where longer rotations are required to meet non-timber goals, a significant benefit exists when establishing Douglas-fir, either in pure or mixed stands. Local data (Figure 4) show that Douglas-fir growth exceeds that of pine before the average 80-year rotation predicted for these areas. Douglas-fir also maintains a consistently high mean annual increment from age 70 to well over 100 years (Curtis 1993).

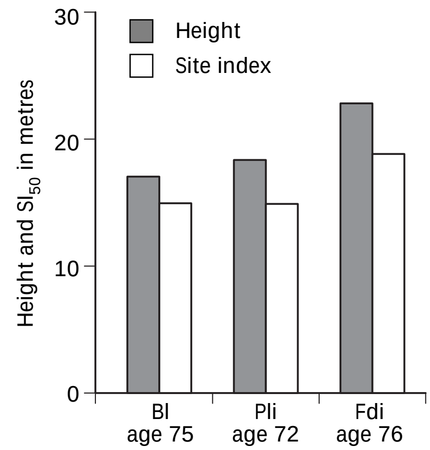

FIGURE 3 *The height growth and derived site index of the three major species on a naturally established second-growth stand in the SBSmk1 (Jumping Lake).*

Douglas-fir plantations can greatly increase the potential for wood production on these subzones while providing for other values, if the Douglas-fir component is retained longer than 80 years. This, in concert with Douglas-fir's links to higher site productivity through its contribution to soil wood, makes it a desirable species to manage over longer rotations.

Current Management Practices

The Douglas-fir bark beetle epidemic in the Fort St. James area has focused the attention of managers on the difficulty and expense of maintaining mature stands when such stands are out of balance with the natural ecological regime of the area. The natural ecological regime evolved in response to relatively frequent fire events. Because of fire exclusion in the past 60 years, many of the Douglas-fir stands in the area now have extremely high stocking.

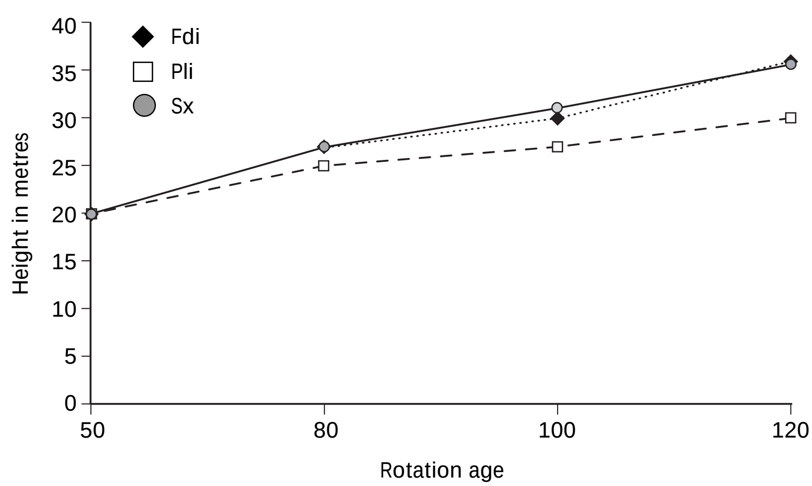

FIGURE 4 *Douglas-fir (Fdi), lodgepole pine (Pli), and spruce (Sx) growth trends at site index at 50 years of 20.*

High stocking in combination with a recently ended 10-year growing season drought has stressed these mature stands, increasing their susceptibility to bark beetle infestation. Current management practices focus on either the minimal harvest of Douglas-fir in some areas or the clearcut harvest of beetle-infested stands. Management regimes either call for thinning from above or clearcutting to remove the high-risk stems from Douglas-fir stands. These regimes are exactly contrary to the natural regime of thinning from below that results from a frequent fire history.

These management practices also have numerous inherent risk factors. Deferring harvest or logging without a management strategy allows bark beetles to infest areas more easily, limits our silvicultural options, and increases the gaps in age-class distribution. Consequently, higher-level objectives can not be realized, and in the long term, might result in a loss of biodiversity.

Goals and Strategic Directions

The key concepts, rationale, and recommendations for managing Douglas-fir in these northern ecosystems are outlined in Table 2. These concepts flow from two basic theses. First, that Douglas-fir and Douglas-fir ecosystems are sufficiently special to demand additional management attention, and second, that Douglas-fir stands must be managed within the context of landscape processes.

Goals and strategic directions for Douglas-fir management must start with a landscape-level plan, preferably by resource management zone (RMZ). Landscape-level planning steps include determining:

- extent of Douglas-fir within the landscape unit, chart, or RMZ;
- reference conditions for the area; and
- management objectives, which include both a spatial component and a temporal component.

Once goals for the unit are set, areas should be ranked for risk factors such as beetle infestation and windthrow, additional steps should be outlined to maintain special sites, and suitable silviculture systems should be chosen to meet short- and long-term goals for the areas.

Silviculture systems other than clearcutting are recommended to meet the multiplicity of objectives and maintain the values associated with Douglas-fir stands. Particularly important is the need to "beetle-proof" high-risk stands by reducing stocking while maintaining the large trees and coarse woody debris thought to be key attributes necessary for wildlife habitat and biodiversity. Beetle-proofing activities are especially critical along the southern aspects of many of the larger lakes in the Fort St. James area. These areas are identified as key wildlife habitat areas and contain significant scenic value. A focus on regenerating more Douglas-fir than is harvested is recommended, at least in the short term, to make up for the shortfall of regeneration in the past two to three decades. Regenerating more Douglas-fir than is harvested will meet the Land and Resource Management Plan (LRMP) goal of maintaining natural levels of Douglas-fir on the landscape. The additional regeneration effort should be concentrated in areas that contain many of the attributes associated with presently occurring Douglas-fir stands.

Meeting this goal requires some administrative changes. Most importantly, the biogeoclimatic ecosystem classification for the SBSdw3, SBSmk1, and SBSwk3 should be revised to include Douglas-fir in a number of site series where it is not currently recognized. Concurrently, Douglas-fir should be included as an acceptable species for these subzones in the *Establishment to*

TABLE 2 *Key concepts for the management of Douglas-fir (Fdi) in the Prince George Timber Supply Area*

Key concepts	Rationale	Recommendations
Landscape-level planning	Stands managed in isolation without the benefit of a succinct landscape-level plan will not produce a pattern suitable for the myriad of users of Fdi and Fdi ecosystems.	Determine target levels of Fdi for the RMZ to set the stage for stand-level management. Plan for active management intervention in specific areas to ensure long-term retention of old-growth attributes. Intervention will take the form of beetle-proofing high-risk stands.
Ecological role	Fdi is unique because it grows so large, lives so long, and takes so long to decay relative to other tree species in our landscapes. These factors make Fdi key in maintaining belowground soil function and aboveground biodiversity.	Maintain a Fdi component in all areas where it currently exists. Plan to retain a component of the largest Fdi in all stands for wildlife trees, coarse woody debris (CWD), and ultimately soil wood.
Productivity	On medium to good sites and rotation ages > 50 years, Fdi is the most productive species found in these northern ecosystems. Fdi enhances site productivity through its function as coarse woody debris and soil wood.	Promote the development of mixed and pure Fdi stands for maximum timber yields. Retain a percentage of the Fdi component past early rotation age for a higher-value product mix. Retain an Fdi component into future rotations to serve as snags, CWD, and soil wood.
Silviculture systems	As Fdi plays a number of significant roles in these northern ecosystems, it is as valuable standing as it is for timber production. Natural disturbance regimes produced a legacy of two-aged stands, which are suitable for a number of partial-cutting regimes.	Focus on partial-cutting regimes that mimic natural disturbance patterns. For optimum results, retained stems should be chosen on the basis of tree *and* site characteristics. Focus on developing stand structures that contain large remnant stems and are resistant to bark beetle infestation.
Site requirements	Fdi excels on rich well-aerated sites, but is prone to frost damage. Establishing Fdi on sites that meet the recommended criteria will result in successful regeneration within time frames that are comparable to that of pine and spruce.	In areas of bark beetle risk, reducing beetle brood habitat by promptly disposing of slash is paramount. Focus on leaving soil wood and humus layers intact during all management operations. Ensure good soil aeration. Plan to minimize frost damage through choice of site and/or use of nurse crops of pine and birch.

Free Growing Guidebook (B.C. Ministry of Forests 1995). When Douglas-fir is associated with birch, definitions of free growing should be revised because of the considerable evidence that the two species are mutually beneficial. Issues around seed planning should also be reviewed to determine precise seed requirements and to develop a suitable cone-collection strategy to meet the needs of all stakeholders in the area.

Regeneration, Recruitment, and Restoration

A number of critical factors must be considered when regenerating Douglas-fir. First, a future stand structure should be determined that meets the majority of landscape-level goals, and then a silviculture system should be chosen to best achieve that stand structure. Partial-cutting regimes are recommended because they can meet a number of landscape-level objectives while still providing timber values.

When retaining overstorey trees in partial-cutting regimes, consider both the site variables and the tree characteristics most likely to result in minimal losses to windthrow and bark beetles after harvest. When prescribing partial-cutting regimes for Douglas-fir harvest in areas of high beetle risk, stands should be beetle-proofed to ensure that the desired number of retained stems will remain. Beetle-proofing activities include choosing the most vigorous trees for leave trees and promptly disposing of slash. Using prescribed fire as both a slash disposal tool and to beetle-proof stands is highly recommended.

Though partial-cutting regimes are preferred, all silviculture systems can successfully regenerate Douglas-fir in this area. Because of existing stand structures, some silviculture systems will require several entries before a particular goal can be met.

Regeneration strategies for Douglas-fir should focus on artificial regeneration because of administrative limitations such as time frames for free growing. Douglas-fir excels on rich, frost-free, well-drained sites. Aeration, both when growing in the nursery and when outplanted is one of the key requirements for successful regeneration. For Douglas-fir to excel, a rich soil substrate, which can take the form of rotten wood, humus, or base-rich mineral soils such as weathered limestone, is required. Therefore, site preparation strategies should ensure that aeration and soil wood and organic layers are retained. Mounding and broadcast burning are both suitable site preparation techniques. Machine-piling is not recommended as it can result in a significant loss of organic material. Douglas-fir is often associated with birch in stands and these two species may form a symbiotic relationship, sharing resources through their mycorrhizae. As well, Douglas-fir is more susceptible to frost damage than other tree species in this area. Because of frost issues and the mutually beneficial relationship between Douglas-fir and birch, brushing should be minimized on sites with birch competition as long as the Douglas-fir maintains adequate leader growth.

FURTHER RESEARCH

This problem analysis has emphasized a number of initiatives that are required to maintain Douglas-fir on our northern landscape. Key elements include the need to co-ordinate a timber harvest strategy, and to develop training packages for partial-cutting harvest operations, leave-tree selection and marking, and plantable spot selection for Douglas-fir.

Further research requirements include studying:

- Relationships between Douglas-fir ecosystems and wildlife species of concern;
- Douglas-fir/birch dynamics;
- Site index correlations with other tree species;
- Douglas-fir's role in maintaining biodiversity and soil productivity;
- Role of renewal agents (fire and bark beetles) in Douglas-fir ecosystems;
- Comparison of Douglas-fir growth dynamics in pure and mixed plantations;
- Requirements for successful natural regeneration; and
- Stand reconstruction studies and identification of traits that contribute to the survival of veteran Douglas-fir

REFERENCES

Curtis, R.O. 1993. Douglas-fir rotations—time for reappraisal? Western Journal of Applied Forestry. 8(3): 81–5.

Franklin, J.F., K. Cromack Jr., W. Denison, A. McKee, C. Maser, J. Sedell, F. Swanson, and G. Judau. 1981. Ecological characteristics of old growth Douglas-fir forests. United States Department of Agriculture, Forest Service, Pacific NW Forest and Range Experimental Station, Portland, Oreg. General Technical Report PNW-118.

Franklin, J.F. and T.A. Spies. 1991. Composition, function and structure of old growth Douglas-fir forests. *In* Wildlife and vegetation of unmanaged Douglas-fir forests. K.B. Aubrey, N. Brookes, J. Agee, R. Anthony, J. Franklin, B. Noon, M. Raphael, R. Storm, and J. Verner (editors). United States Department of Agriculture, Forest Service, Pacific NW Research Station, Portland, Oreg. General Technical Report PNW-GRT 285.

Jull, M. 1997. Douglas-fir silviculture: 'on the edge' silvicultural systems at the northern range of the species. *In* Ecology and management of Douglas-fir at the northern limits of its range. J.D. Lousier and W. Kessler (editors). University of Northern British Columbia Press. Prince George, B.C. In press.

Nigh, G.D. 1994. Site index conversion equations for mixed species stands. B.C. Ministry of Forests, Research Branch, Victoria, B.C. Research Report 01.

Thrower, J.S., A.F. Nussbaum, and C.M. Di Lucca. 1991. Site index curves and tables for British Columbia: interior species. Field Guide Insert 6. Forest Productivity and Decision Support Section, Forest Sciences Research Branch. B.C. Ministry of Forests, Victoria, B.C.

PARTCUTS: A Computerized Annotated Bibliography of Partial-cutting Methods in the Pacific Northwest (1995 Update)

PATRICK DAIGLE

INTRODUCTION

Forest resource professionals are challenged to develop biologically sound, economically feasible, and socially acceptable ways to manage public and private forest resources. Forest managers are facing growing social pressure against clearcutting practices.

The PARTCUTS bibliography provides a compilation of written materials (published between 1970 and 1995) that describe alternatives to clearcutting and seed-tree silvicultural methods in Pacific Northwest forest types that contain:

- coastal Douglas-fir
- Engelmann spruce
- interior Douglas-fir
- lodgepole pine
- ponderosa pine
- subalpine fir
- western hemlock
- western larch
- western redcedar

Silvicultural Practices Included

Terms used in the compiled literature include:

- commercial thinning
- diameter limit
- group selection
- improvement cut
- individual-tree selection
- overstorey removal
- partial cutting
- salvage cut
- sanitation cut
- selection cut
- shelterwood systems

How to Use the PARTCUTS Bibliography

PARTCUTS can be used on IBM-PC or compatible microcomputers. Users can browse references dealing with various aspects of partial-cutting practices. Citations, keywords, and authors' abstracts can be displayed on the screen, stored in a file, or printed. For instance, by searching the keyword terms "*Pinus contorta*" and "bark beetles," a person can locate 25 years of published information about beetle-proofing lodgepole pine stands.

Obtaining a Copy of PARTCUTS

The PARTCUTS bibliography can be obtained from the Internet at http://www.publications.gov.bc.ca, or by phoning 1-800-663-6105.

Uneven-aged Interior Douglas-fir Stand Dynamics at Pothole Creek (Abstract of poster)

CATHERINE BEALLE-STATLAND

ABSTRACT

Uneven-aged interior Douglas-fir stand structure and dynamics are complex as a result of irregular fire history, insect and disease outbreaks, and various forms of partial harvesting and range use. Stocking, crown closure, canopy layering, size-age relationships, and spatial distribution are highly variable from one stand to another and within any one stand. Many different factors interact to influence individual tree development.

To study tree regeneration and growth relationships within this important but poorly understood forest type, a 5-year project was initiated in 1996 at a research site within the Pothole Creek Demonstration Area near Merritt by the Forest Productivity and Decision Support group of the B.C. Ministry of Forests Research Branch[1]. The objectives of the project are to intensively study some fairly productive, pest-free, single-species multi-cohort stands to identify and describe the most important factors and interactions that determine individual tree growth. The ultimate goal is to develop functions that will enable us to model uneven-aged stand growth and yield.

In the first phase of the project, we concentrated on exploratory studies of basic stand structure and biology, developing field methods and identifying potentially fruitful areas for further, more formal study and experimentation. Subprojects are under way to study current and past stand structure, short-term growth estimation, light measurement and modelling, tree spatial distribution modelling, age and height growth of advanced regeneration clumps, coarse woody debris, seedling germination and survival, root distribution and growth, moisture availability, and the effects of light and moisture distribution on tree physiology. Ongoing analysis of the data collected so far will help determine future directions. During the 1997 field season, we continued the studies started in 1996 and located additional study sites to compare with the one already established.

1 Funding provided by Forest Renewal BC.

The Opax Mountain Silvicultural Systems Project: Evaluating Alternative Approaches to Managing Dry Douglas-fir Forests

WALT KLENNER AND ALAN VYSE

SILVICULTURAL PRACTICES IN DRY DOUGLAS-FIR STANDS

The dry Douglas-fir (*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco) forests of British Columbia's south-central interior (for descriptions see Lloyd et al. 1990; Meidinger and Pojar 1991; Vyse et al. 1991; for location see Figure 1) have been managed by stand-level partial cutting for 50 years. The first foresters in the area recognized that they were dealing with a forest of some structural complexity. Charged with the responsibility of bringing order to a previously uncontrolled exploitation regime and concerned about the need to sustain timber yields, they set about providing an economical supply of timber while reserving a vigorous residual stand with sufficient seed source to restock gaps (Benteli 1955). A system with frequent, light cutting cycles was proposed for sites that were generally easy to access and cheap to log in all seasons. This regime was implemented by field crews who marked stems either to cut or to leave, while striving to leave a well-distributed layer of thrifty stems for future harvest. When the demand for timber began to rise in the 1960s and more logging licences were approved, marking was eliminated in favour of easily administered diameter limits. The inevitable result was poor-quality residual stands, with many damaged small stems and irregular stocking.

After a decade of unsatisfactory results (Vyse et al. 1991) foresters again sought more control over the loggers, but this time encouraged tree fallers to leave a well-distributed layer of vigorous large stems and to plan skid trails to reduce logging damage (Chan 1987). As the value of timber has increased in the last decade, stem marking has been re-introduced, and ever more complicated schemes for stocking control proposed (B.C. Ministry of Forests 1992).

Today, most stands in the Southern Interior have been logged at least once and records show that some stands have been logged three times in the last 90 years (Chan 1987; Vyse et al. 1991). Interior Douglas-fir is still the dominant species and, as in the past (judged by old photographs), size distributions are still irregular with varying numbers of regeneration, poles, and canopy stems. The largest trees rarely exceed 30 m in height and 70 cm in diameter. Stand volumes range widely, but reach 250–300 m^3/ha in relatively undisturbed stands. Stand conditions vary widely at the local level, but the same pattern is repeated throughout the region.

How these stands are managed continues to be an issue. It is widely believed that these stands of dry Douglas-fir should be managed on an uneven-aged basis with a balanced distribution of age classes represented at a scale of about 0.1 ha (Vyse et al. 1991; B.C. Ministry of Forests 1992; Day 1996). However, there are several reasons why the extensive application of this common management prescription may not be appropriate without considering alternatives.

- It seems unlikely that one general prescription would be applicable to a highly variable forest type that extends over 1 million hectares of forest land, even though these lands support one dominant tree species. The variations in climate, soils, vegetation, regeneration, pest incidence, logging history, and management objectives seem likely to disrupt such a simplistic management approach.
- Although multi-age stand structures can be found within the current wide range of forest structures, they are only one of many possible sustainable structures. Current structures are the result of one to several cycles of cutting over several decades, superimposed on a complex and poorly understood wildfire history, and varying, but often high, levels of insect and disease incidence.
- A stand structure goal of several age or size classes intermixed at a fine scale is highly susceptible to western spruce budworm (*Christoneura occidentalis* Freeman), and at lower elevations to Douglas-fir tussock moth (*Orgyia pseudotsugata* McD.). In particular, the conifers in smaller size classes can suffer high levels of defoliation over several years, which often results in mortality, stem defects, and loss of growth. The risk of such attacks is very high in the southern portion of the Interior Douglas-fir zone, with a return interval of 10–20 years (Harris et al. 1985; Alfaro and Maclauchlan 1992).
- The uneven-aged management goal assumes that small seedlings are recruited and grow at a steady rate to replace seedlings in larger size classes as harvesting entries are made. However, little evidence supports this proposition in dry Douglas-fir forests. In contrast, operational staff in the Kamloops Forest District have recently expressed concern over poor regeneration performance following partial-cutting practices over the last 30 years (Mather 1997; P. Lishman, pers. comm., 1997).
- The proponents of uneven-aged management assume that the goal of balanced age structure will meet wildlife habitat needs and provide suitable summer range for cattle. Both assumptions are contentious. For example, frequent uniform partial cutting is very likely to reduce or eliminate large snags in the stand.
- The administrative, planning, and supervisory requirements for a balanced age class approach are high and have prevented industrial foresters in some parts of the zone from submitting cutting permits in this timber type. Logging costs are not an issue because contractors are accustomed to low-volume sales, and considerable competition exists for wood close to the local mills.

In response to the concerns about the widespread application of uniform partial cutting, and a more general need to test many long-held assumptions about appropriate silviculture systems in dry Douglas-fir forests, a long-term examination of silvicultural options in dry Douglas-fir forests was initiated 20 km northwest of Kamloops, B.C. at Opax Mountain (Figure 1). In co-operation with the Small Business Forest Enterprise Program in the Kamloops Forest District, initial planning began in 1992, and the site was logged in the winter of 1993–1994.

The treatments used did not represent operational prescriptions, but instead were focused on creating or removing specific stand structures (e.g., number of large-diameter trees > 35 cm) and creating a range of canopy gaps (e.g., 0.01–0.05 ha in the partial-cutting units, and 1–4 tree heights wide [0.09, 0.36, and 1.6 ha]) in the patch-cut units. The treatments were applied by harvesting 20-ha blocks in two different ways (small patch cuts versus uniform selection) at two levels of cutting intensity (20 and 50% of stand volume). An additional treatment removed timber by the uniform selection method at the higher level of intensity, but retained one-quarter of the total cutting block in uncut patches. These treatments, plus a control area of similar size, were applied in two different areas in a randomized complete-block design. One area has an elevation range of 950–1100 m in the IDFxh subzone and the other, in the IDFdk subzone, has a range of 1200–1370 m. The design is shown in Figure 2. A range of site preparation methods were also applied to the site, but on a very small scale.

A wide range of studies have been conducted on the site and include microclimate, stand fire history, soil chemistry, soil biology, stand growth and yield, conifer regeneration, all non-conifer vegetation including lichens and bryophytes, and wildlife responses to the cutting. Results from some of these studies are reported in this volume of workshop proceedings.

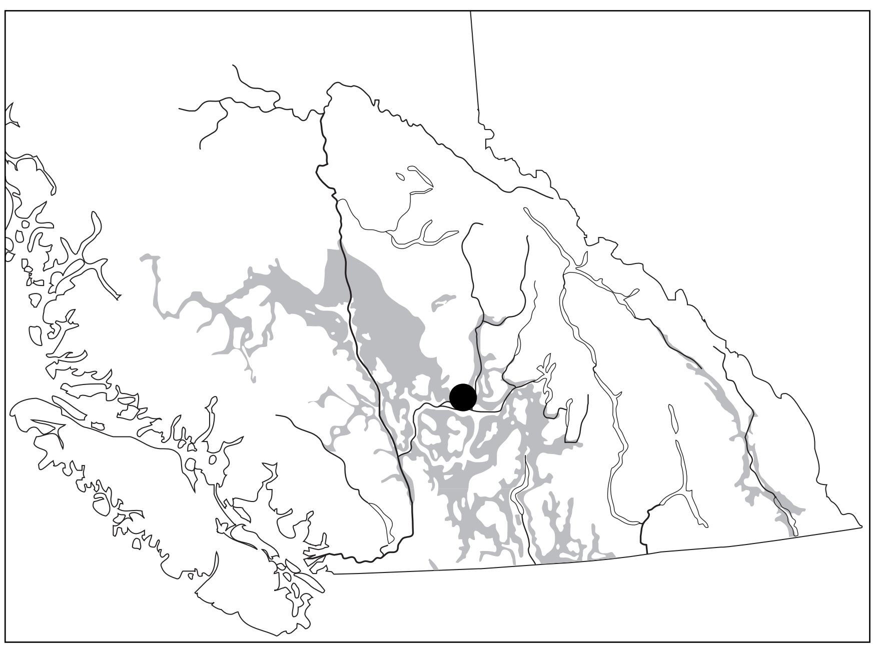

FIGURE 1 *Location of the Interior Douglas-fir zone in British Columbia and the location (●) of the Opax Mountain Research Project.*

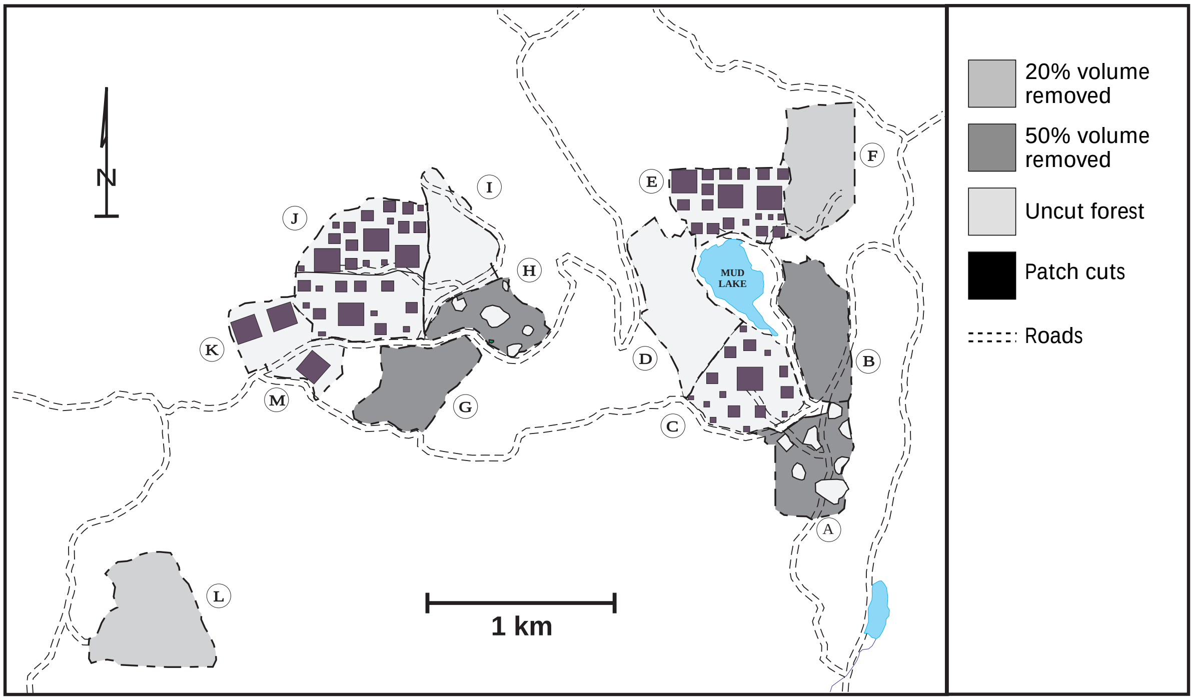

FIGURE 2 *Harvesting treatments at the Opax Mountain research site: 20% volume removal using individual-tree selection (units F, L); 35% volume removal, consisting of 75% of the treatment unit area harvested as 50% volume removal using individual-tree selection, and 25% of the treatment unit area retained as uncut reserves (units A, H); 50% volume removal using individual-tree selection (units B, G); patch cuts of 0.1, 0.4, and 1.6 ha on 20% of the treatment unit area (units C, K); patch cuts of 0.1, 0.4, and 1.6 ha on 50% of the treatment unit area (units E, J); uncut controls (units D, I).*

RESEARCH TEAM

The nucleus of the research team was established in 1992 when scientists from various disciplines were invited to join Forest Service team members. The team has since grown to include the scientists and studies shown in Table 1. Membership is not restricted and additional research scientists are welcome to participate. The team has met on an informal basis on several occasions to make basic decisions on project design and to identify project connections and gaps.

ACCOMPLISHMENTS

Much has already been accomplished and delivered in the project. The project has been designed, the site selected and prepared, the project team assembled, operational co-operation sought and received, experimental and operational plans prepared, pre-logging studies carried out, logging carried out and monitored, and post-logging impact studies begun. More detailed early results of this work are reported in the remainder of this volume.
A brief summary of the early results follows.

TABLE 1 *List of studies and investigators at the Opax Mountain Silvicultural Systems Project*

Study	Title	Principal investigators
1.	Growth and yield of interior Douglas-fir.	Catherine Bealle-Statland, Research Branch, B.C. Forest Service, Victoria, B.C.
2.	Effects of silvicultural systems at Opax Mountain on the structure of interior Douglas-fir songbird communities.	Tom Dickinson, Nancy Flood, and Ernest Leupin, University College of the Cariboo, Kamloops, B.C.
3.	Habitat and landscape relationships and the effects of silvicultural systems: squirrels; cavity nesters; small carnivores; ungulates; arthropods; and insectivores.	Walt Klenner and Dave Huggard, Kamloops Forest Region, B.C. Forest Service, Kamloops, B.C.
4.	Relationships between small mammals and downed wood.	Walt Klenner, see above. Tom Sullivan, Applied Mammal Research Institute, Summerland, B.C. Vanessa Craig, Faculty of Forestry, UBC
5.	Effects of alternative silvicultural systems on the habitat requirements of amphibians in IDF forests.	Walt Klenner, see above. Christine Ferguson, Faculty of Forestry, UBC.
6.	The potential of shrubs to act as refuge for ecto-mycorrhizal innoculum following logging in a dry-belt fir forest.	Dan Durall, Okanagan University College, Kelowna, B.C.
7.	Effects of silvicultural systems on soil productivity in the IDF zone; impacts on soil organic matter and soil nitrogen.	Graeme Hope, Kamloops Forest Region, B.C. Forest Service, Kamloops, B.C.
8.	Spatial and temporal response of vegetation to silvicultural practices in the IDF.	Dennis Lloyd and Andre Arsenault, Kamloops Forest Region, B.C. Forest Service, Kamloops, B.C.
9.	Advanced regeneration in IDF forests.	Pasi Puttonen, Research Branch, B.C. Forest Service, Victoria, B.C. Alan Vyse, Kamloops Forest Region, B.C. Forest Service, Kamloops, B.C.
10.	Effects of opening size on disease in IDF forests.	Hadrian Merler, Kamloops Forest Region, B.C. Forest Service, Kamloops, B.C.
11.	Effects of insects on health of trees and stands in the Opax Mountain project.	Lorraine Maclauchlan, B.C. Forest Service, Kamloops, B.C.
12.	Seedling establishment in openings of IDF forests managed with partial cuts.	Pasi Puttonen, see above. Alan Vyse, see above.
13.	Reconstruction of past fire events at Opax Mountain.	Andre Arsenault, see above.
14.	Seedfall in patch cuts and partial cuts at Opax Mountain.	Alan Vyse, see above.
15.	Microclimate at Opax Mountain.	Ralph Adams, Research Branch, B.C. Forest service, Kamloops, B.C.

- The logging contractors were able to operate efficiently in all treatments, with the exception of the 20% uniform selection cut. This demonstrates that few operational constraints exist to the choice of treatment methods.
- Herbaceous response to complete overstorey removal was rapid and very dramatic, presumably in response to increased moisture and light reaching the forest floor; changes in the quality of forage are still under investigation.
- Soil moisture levels in the cleared patches were similar to those in the uncut stands, presumably because the herbaceous vegetation captured and used the same amount of moisture as the former forest.
- Wildlife response to the treatments has been varied; the open conditions of the high-volume removals have favoured some species of small mammals, while the loss of snags in the uniform, partial-cut treatments has had a detrimental effect on woodpecker foraging habitat.
- Seedlings of Douglas-fir, ponderosa pine, and lodgepole pine planted in the patch-cut and partial-cut treatments following mechanical site preparation have survived well and performed above expectations for the dry Douglas-fir type.
- Windthrow losses were expected to accelerate following logging on the site, but after 3 years the occurrence of windthrow appears more related to site conditions than logging treatment.

Demonstration of the project results has also been a high priority. With the co-operation of the Kamloops District Recreation section, we built a trail through part of the project area to relay the story of silviculture in dry Douglas-fir forests to both forestry personnel and the general public. Numerous tours by local forest managers, Canadian and foreign students, school groups, visitors to the Kamloops area, and interested local residents have been guided around the 2-km loop in the last 2 years. A brochure is also available for self-guided tours. The trail is designed to demonstrate the natural features of the site and the effects of stand management alternatives.

FUTURE DIRECTIONS

The Opax Mountain project results show that alternatives to uniform partial cutting have some advantages that should be pursued more vigorously by both researchers and forest managers. We expect that future research results from Opax Mountain will contribute to this re-examination, along with results from other trials in other parts of the dry Douglas-fir zone. New research areas in this forest type should also be explored, including the role of cattle grazing, the use of controlled fire as a tool to create stand structures, and the application of the new uneven-aged dry forest management concepts (O'Hara 1996), which promote a broader, more flexible approach based on such factors as regeneration difficulties, structural features for non-timber resources, and allocating more growing space to older, more rapidly growing stems for increased timber growth.

ACKNOWLEDGEMENTS

The project would not have been possible without the dedication to improved forestry practices of the Kamloops Forest District, their staff in the Small Business program, and their logging contractor for Opax Mountain, who worked hard to ensure that the project was implemented as planned and within a reasonable time frame.

The project is one of many that was provided encouragement and support by the Provincial Silvicultural Systems Research Program. In the last 3 years, generous funding support has been provided by Forest Renewal BC.

REFERENCES

Alfaro, R.I. and L.E. Maclauchlan. 1992. A method to calculate the losses caused by the western spruce budworm in uneven-aged Douglas-fir forests of British Columbia. Forest Ecology and Management 55:295–313.

Benteli, S. 1955. Marking of Douglas-fir: a preliminary investigation of the problems restricted to interior dry types. B.C. Forest Service, Victoria, B.C. Internal Report.

British Columbia Ministry of Forests. 1992. Correlated guidelines for management of uneven-aged drybelt Douglas-fir stands in British Columbia. B.C. Ministry of Forests, Silviculture Branch, Victoria, B.C.

Chan, R. 1987. Dry belt fir timber management: a review for practitioners. Paper presented to the Dry Belt Fir Timber Management Workshop, Oct. 1987, Williams Lake, B.C. Northern and Southern Interior Silviculture Committees.

Day, K. 1996. Interior Douglas-fir and selection management. Faculty of Forestry, Department of Forest Sciences, University of British Columbia, Vancouver, B.C. Directed Study Report.

Harris, J.W.E., R.I. Alfaro, A.G. Dawson, and R.G. Brown. 1985. The western spruce budworm in British Columbia 1909–1983. Canadian Forest Service, Victoria, B.C. Information Report BC-X-257.

Lloyd, D., K. Angove, G. Hope, and C. Thompson. 1990. A guide to site identification and interpretation for the Kamloops Forest Region. B.C. Ministry of Forests, Victoria, B.C. Land Management Handbook No. 23.

Mather, J. 1997. Dry Douglas-fir regeneration following logging in the 1960's and 1970's in the Kamloops Forest Region. Poster presented to the Dry Douglas-fir Workshop, April 1997.

Meidinger D. and J. Pojar (editors). 1991. Ecosystems of British Columbia. B.C. Ministry of Forests, Victoria, B.C. Special Report Series No. 6.

O'Hara, K. L. 1996. Dynamics and stocking-level relationships of multi-aged ponderosa pine stands. Forest Science. Monograph No. 33.

Vyse, A., R.A. Smith, and B.G. Bondar. 1991. Management of Interior Douglas-fir stands in British Columbia: past, present and future. *In* Interior Douglas-fir: the species and its management. Symposium proceedings. D.M. Baumgartner (editor). Washington State University, Department of Natural Resource Sciences, Pullman, Wash.

Stand Structure and Growth Estimates for the Opax Mountain Silviculture Systems Trial

CATHERINE BEALLE-STATLAND

INTRODUCTION

Foresters have prescribed partial cutting in the British Columbia Interior Douglas-fir (IDF) dry-belt forest type for many years with little quantitative information to guide them and with mixed results. This silvicultural approach is increasingly relied on to combine timber harvesting with enhancing regeneration, minimizing visual impacts, and maintaining animal habitats. A better understanding of stand dynamics is imperative if partial cutting regimes are to meet these goals. Decisions about harvest cycle, residual stocking, and residual stand structure should not be made without some awareness of how these will influence subsequent natural regeneration, timber production, and stand attributes important for wildlife (e.g., snags, woody debris, vegetation, and canopy cover). These influences depend on the post-harvest tree growth and mortality rates, and on multi-cohort stand development patterns.

Interior dry-belt Douglas-fir stands are complex; therefore, assessing stand structure and growth is difficult. Irregular fires, insect and disease outbreaks, and past harvest entries have rendered crown closure, canopy layering, size-age relationships, and spatial distribution highly variable from one stand to another and within any one stand. Consequently, precise stand surveys are usually laborious and costly, and partial-cutting regimes are difficult to prescribe and administer. Quantifying both the results of the harvest operation and post-harvest growth is complicated. Of the many partially cut stands in the Kamloops Forest Region, few have been rigorously examined. The Opax Mountain trial provided an opportunity to thoroughly measure both pre- and post-treatment stand structure and to observe stand development over the long term. The work presented here describes an attempt to do this, with acceptance of the inevitable limitations. The objectives of this study were:

1. to document the volume, stem numbers, volume removed, and volume left standing in the Opax Mountain stands before and after treatment, comparing the results to the prescriptions; and
2. to install a system of permanent plots in the uniform stand treatments to measure tree growth and observe stand development over time.

This silviculture systems trial was designed mainly to test the effects of different partial-cutting prescriptions on wildlife habitat and habitat use and was not considered suitable for a formal growth and yield experiment. The replicated areas are not sufficiently similar (different biogeoclimatic subzone variants), nor sufficiently uniform within and among the treatment units. The assignment of treatments to the units was not entirely random and the treatments were defined by percent volume removal, with little control of the residual stocking. Hence, this study cannot report definitively on growth response to the different levels of treatment. Nevertheless, measuring the growth and regeneration of at least the uniform treatments is worthwhile, given the lack of good data for this forest type and the amount of pre- and post-treatment information available. It will also provide supporting information for the many other studies under way in this area.

SITE DESCRIPTION

The silviculture systems trial is about 16 km northwest of North Kamloops, B.C. and consists of two blocks of five partial-cut treatments and an untreated control. One block covers the southerly slopes of Opax Mountain at 1200–1370 m elevation. This area is classified as mainly the Interior Douglas-fir Dry Cool (IDFdk1) biogeoclimatic subzone variant, with the upper reaches in the Montane Spruce Dry Cool (MSdk1) biogeoclimatic subzone variant. The other block surrounds nearby Mud Lake at 950–1100 m elevation and is in the IDF Very Dry (IDFxh2) variant. The forest in both areas is open and patchy. This is likely the result of microsite variation and repeated small disturbances, such as fire, insects, isolated root rot infections, windthrow, cattle grazing, and irregular harvest entries.

The Mud Lake block (139 ha) is mainly Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) with some spruce (*Picea engelmannii* x *glauca*), lodgepole pine (*Pinus contorta*), trembling aspen (*Populus tremuloides*), and paper birch (*Betula papyrifera*). The block was partially harvested in 1956 and 1957, probably with a diameter-limit method of single-tree selection. Regeneration after this disturbance was sparse and slow growing, particularly on the rocky, exposed sites. Subdominant and intermediate trees have weak crowns and some top dieback from a combination of poor site quality, drought, insect attack (particularly western spruce budworm [*Choristoneura occidentalis*]), and possibly laminated root rot (*Phellinus weirii*). Many of the largest-diameter Douglas-fir trees have bore holes and pitch tubes from bark beetle (*Dendroctonus pseudotsugae*) attack.

The upper-elevation block on Opax Mountain (127 ha) has a similar open and patchy spatial structure, but a different species balance. The stocking is approximately two-thirds Douglas-fir and one-quarter to one-third lodgepole pine by basal area, with minor components of spruce and aspen. Some harvesting took place in 1957 close to the roads, but large areas remained uncut. Growth is more vigorous and regeneration healthier and more abundant than in the Mud Lake block, although many shaded understorey trees have weak crowns and poor form. Less spruce budworm damage and top dieback is apparent; however, the lodgepole pine suffers from *Lophodermella* needle disease and the large Douglas-fir have beetle attack symptoms.

Stand Structure before and after Treatment

Before harvesting, a consultant installed a systematic grid of 159 temporary 0.04-ha fixed-area strip plots (each 8 × 50 m) across the entire 266 ha experimental area for a pre-harvest silviculture prescription (PHSP) cruise. The plots were located 130–140 m apart along east-west transects which were spaced about 150 m apart and crossed freely over the treatment unit boundaries. Seventy-seven plots were established in the Mud Lake block and 82 in the Opax Mountain block, with an average of 13 plots falling within each of the 12 treatment units. This represents a sampling intensity of about 2.2% by area over the two blocks and was designed to estimate the total volume on each block to within 10% of the true volume. Data recorded for all trees 12.5 cm diameter or larger on each plot included:

- tree species,
- diameter at breast height,
- standard damage and defect codes,
- tree height, and
- samples of tree age.

Advanced regeneration was tallied on one 0.005-ha circular plot nested within each larger plot.

To implement the target volume removals, the consultant compiled stand and stock tables for each treatment unit from data collected on plots that fell within the unit's boundaries. These tables were then used to estimate upper diameter limits for the uniform single-tree selection and the size and number of openings for the patch-cut areas, based on the volume of all conifer stems larger than 12.5 cm diameter at breast height (Table 1). The diameter limits were not the same for the two blocks because the pre-treatment stocking was different and because the treatments were defined by the percentage of timber removed from the original overstorey.

To study the stand structure more closely and to obtain data for growth modelling, I visited each of the plots before the treatments and recorded plot aspect, slope, mesoslope position, elevation, crown length, additional tree ages, and samples of past 10-year diameter growth and past 5-year height growth. I then compiled my own plot summaries and tables, using methods similar to those used by the consultant (without deductions for defect and decay, however). After the harvesting was completed in the winter of 1993/1994, I relocated and remeasured each plot to estimate the residual stand structure. All previously measured stems that had been cut or damaged were noted. To compare with pre-treatment data, post-harvest stand and stock tables were constructed for each block as a whole and for each treatment unit within each block. To estimate the actual percent volume removed and confidence limits for this estimate, a ratio R_{iv} of the difference between the initial and after-treatment volumes over the initial volume was calculated for each plot:

$$Riv = \frac{V_{i2} - V_{i1}}{V_{i2}} \qquad (1)$$

where: V_{i1} = initial volume of plot i

V_{i2} = post-treatment volume of plot i

TABLE 1 *Treatment prescriptions for the Opax Mountain silviculture systems trial*

Treatment	Block	Treatment unit area (ha)	Prescription
20% removal of standing volume via uniform single-tree selection.	Mud Lake	F (23.7)	Diameter limits: 40 cm for Fdi[a] and Py, 32.5 cm for Pl and Sx; snags felled.
	Opax Mountain	L (28.3)	Diameter limits: 57.5 cm for Fdi and Py, 37.5 cm for Pl and Sx; snags felled.
50% removal of standing volume via uniform single-tree selection	Mud Lake	B (21.3)	Diameter limits: 44.0 cm for Fdi; snags felled.
	Opax Mountain	G (22.2)	Diameter limits: 42.5 cm for Fdi and Py, 22.5 cm for Pl; snags felled.
20% removal of standing volume via patch cuts	Mud Lake	C (24.5)	16 square patch cuts of sizes ranging from 0.1 to 1.7 ha.
	Opax Mountain	K (25.8)	As for Unit C, except only 14 patch cuts.
50% removal of standing volume via patch cuts	Mud Lake	E (21.1)	21 square patch cuts of sizes ranging from 0.1 to 1.7 ha.
	Opax Mountain	J (22.0)	As for Unit E, except only 18 patch cuts.
50% removal of the standing volume by single-tree harvest from above, but with reserve patches left untouched within the unit around snags, large veterans, and broadleaf groups for a total removal of about 35% of the original standing volume.	Mud Lake	A (20.9)	6–7 reserve patches of sizes 0.25–1.5 ha to total 6.5 ha; diameter limits: 37.5 cm for Fdi, Py, Pl, and Sx.
	Opax Mountain	H (15.2)	As for Unit A
Control	Mud Lake	D (27.5)	No harvest
	Opax Mountain	I (21.4)	No harvest

a Species codes: Fdi = Douglas-fir; Py = ponderosa pine; Pl = lodgepole pine; Sx = hybrid spruce.

From these plot ratios, a mean ratio, $\bar{R}_v$, and 95% confidence interval was calculated for each treatment unit.

Tree Growth and Stand Development Permanent Sample Plots

In 1995 (1 year after treatment), the temporary strip plots within the single-tree selection treatment units and the controls (Units B, F, and D in the Mud Lake block; Units G, L, and I in the Opax Mountain block) were upgraded to create a system of 90 small permanent sample plots: 49 in the Mud Lake block and 40 at Opax Mountain. Permanent plot stakes and tree tags were installed and the trees remeasured to the more rigorous standards recommended by the Forest Productivity Councils of B.C. (1995) for permanent sample plots. Twelve additional strip plots were added to improve the volume estimates within the most variable treatment units, A and D. These plots will be protected within the Opax Mountain demonstration forest and remeasured every 5 years.

RESULTS

Stand Structure Before and After Treatment

Before harvest, the average total volume across the two blocks was approximately 175 m^3/ha.[1] Pre-treatment volume and stem distributions by diameter class for the two blocks are shown in Figures 1 and 2. Figure 1b clearly shows the significant lodgepole pine population in the Opax Mountain block. This was dominated by a 100- to 120-year-old cohort. The Douglas-fir trees on both blocks were roughly even-aged within single clumps and patches, but these cohorts overlapped extensively to produce a multi-cohort stand.

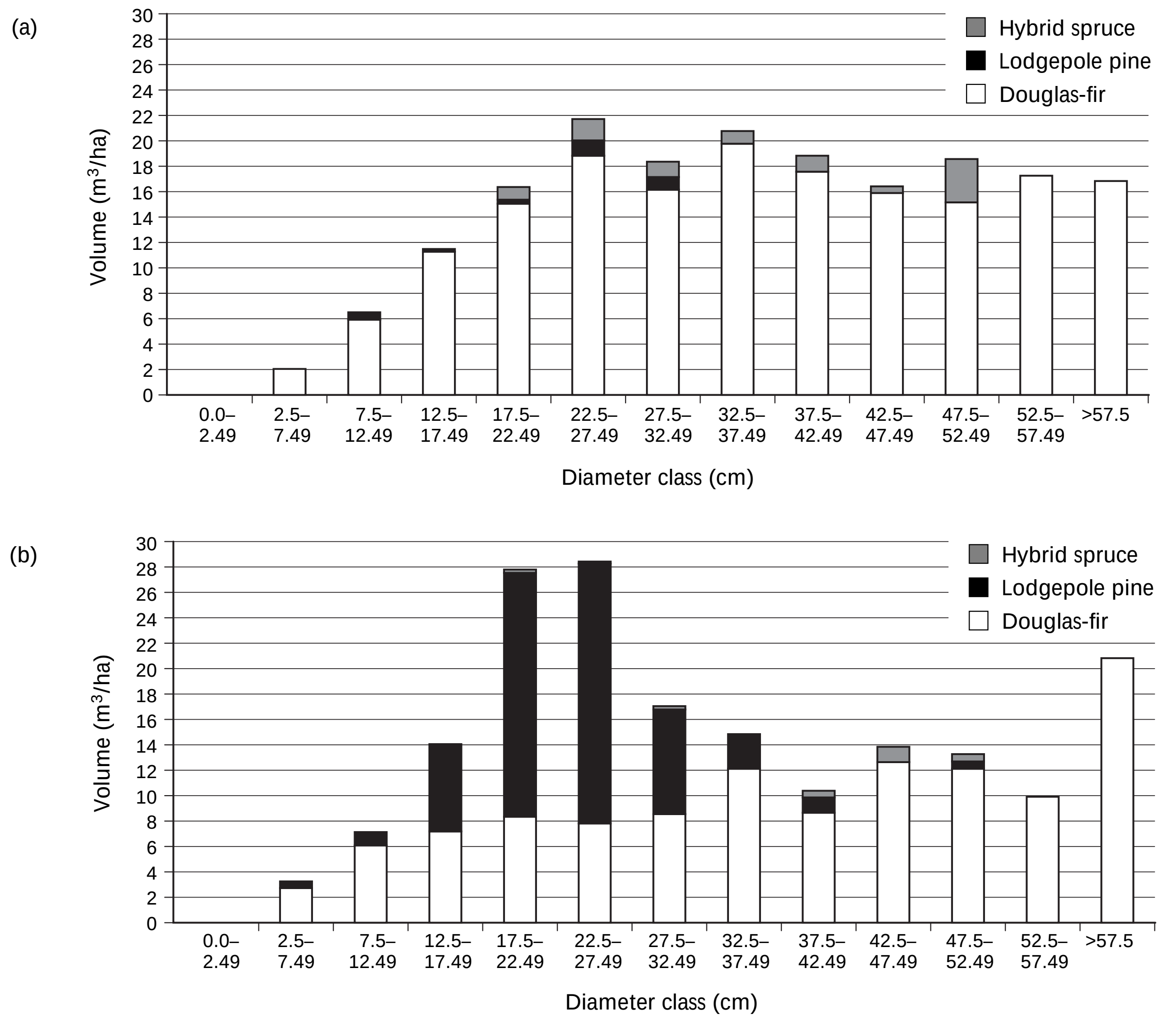

FIGURE 1 *Pre-treatment total volume distributions by 5-cm diameter classes: (a) Mud Lake block (b) Opax Mountain block.*

1 All volumes reported in this paper were calculated by numerical integration of Kozak's taper equations (Kozak 1988) and summation by Newton's formula (Husch et al. 1972) for individual tree volume. Volume does not include reductions for defect, decay, waste, or breakage.

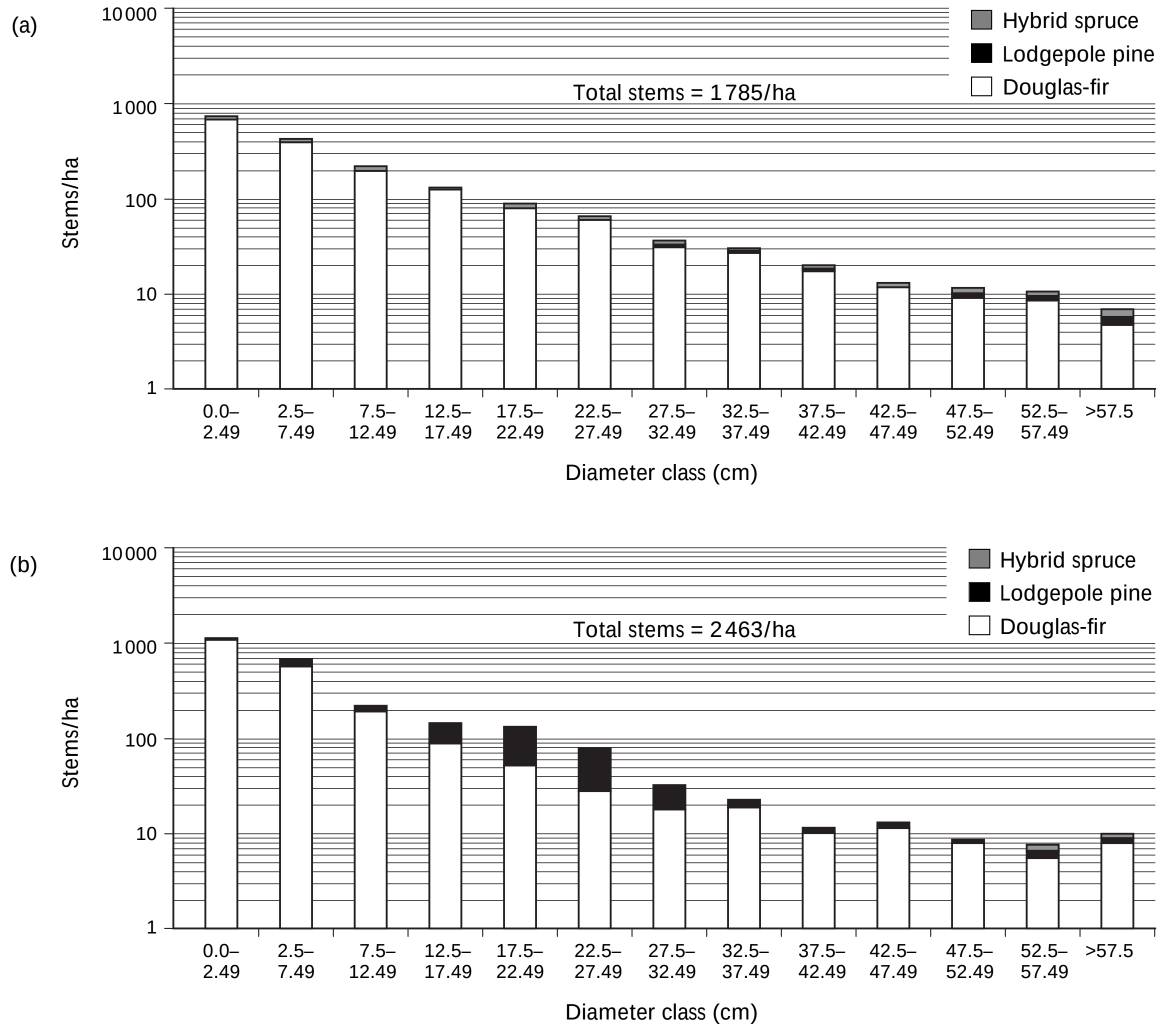

FIGURE 2 *Pre-treatment diameter distributions by 5-cm classes: (a) Mud Lake block; (b) Opax Mountain block.*

Overstorey trees ranged from 70 to 220 years old, with a few 250- to 400-year-old veterans. Subdominants and intermediates were 40–70 years and regeneration 8–40 years old. Except within these broad groups, tree age was not well correlated with tree size. Increment core samples indicated that many trees were suppressed for a period of time. A logarithmic scale has been used for the diameter distributions (Figures 2a, 2b) to more clearly show the detail in the larger, but less frequent, size classes. Initial stem densities were about one-third higher in the Opax Mountain block than at Mud Lake. Most of the difference was attributable to the smaller size classes of the Mud Lake block, which had about 600 fewer trees per hectare under 12.5 cm diameter. At Opax Mountain, 459 stems per hectare were larger than 12.5 cm, as were 406 stems per hectare at Mud Lake.

Within the treatment units, coniferous volumes (stems at least 12.5 cm diameter) ranged from 133 to 207 m^3/ha (Table 2). In some cases, the two treatment replications had very similar initial standing volume (e.g., the 50% patch-cut areas E and J, and the control units D and I); in others, the initial volumes differed by 20–25 m^3/ha. Initial basal area within the units ranged from 21.2 to 27.8 m^2/ha. In the Mud Lake block, most of this was Douglas-fir,

TABLE 2 *Stocking of Opax Mountain treatment units before and after harvesting, showing prescribed and actual residual volume*

	Pre-treatment stand					Post-treatment stand				
Unit[a]	Volume[b] m^3/ha	Basal area m^2/ha	% Fdi[c]	% Pl	No./ha[d]	Volume m^3/ha	Basal area m^2/ha	% Fdi	% Pl	No. /ha
F (20%)	157.8	21.2	91	4	1827	108.2	15.4	97	3	1602
L (20%)	132.6	21.1	65	35	3683	93.4	15.4	58	40	2992
B (50%)	207.1	27.8	100	0	1938	67.5	11.0	100	0	959
G (50%)	190.5	24.2	68	31	2290	75.4	10.1	52	46	2044
C (20%)	170.5	22.9	88	0	1369	145.5	19.5	86	0	1205
K (20%)	167.2	23.4	72	27	2455	126.1	18.0	67	28	2521
E (50%)	184.1	24.7	97	3	1769	73.4	10.1	99	1	700
J (50%)	180.0	24.2	57	43	2106	92.9	12.9	58	40	1977
A (35%)	164.6	21.7	91	0	2026	97.6	13.6	98	0	1462
H (35%)	185.8	26.0	67	29	1914	85.8	12.9	56	44	1932
D	174.0	22.4	88	0	1646	174.0	22.4	88	0	1646
I	171.9	25.8	83	16	2302	171.9	25.8	83	16	2302

a Prescribed removal in brackets.
b All volumes are total gross, including coniferous stems at least 12.5 cm diameter.
c Species codes: Fdi = Douglas-fir; Pp = ponderosa pine; Pl = lodgepole pine; Sx = hybrid spruce.
d Stem counts include all coniferous trees at least 0.30 cm tall.

with small components of lodgepole pine and spruce. The Opax Mountain block had a greater proportion of lodgepole pine (one-quarter to one-half) and less spruce. Only treatment units F, G, and L had a measurable broadleaf component (mainly trembling aspen). The harvesting reduced the basal area from 10.1 to 18.0 m^2/ha, but maintained roughly the same species composition, with only small decreases in Douglas-fir in some of the treatments.

Table 3 shows the actual percent volume removed in each treatment unit based on the mean plot ratio calculations (Equation 1). These data highlight the difficulty in measuring compliance with specified partial-cutting targets. With the exception of Unit G, the prescribed percent volume removal is within the 95% confidence interval for each treatment unit. However, these confidence intervals are very wide and indicate that the sampling precision is capable of detecting only very large deviations from the prescriptions. The 50% uniform cut of Unit G has definitely been over cut, but not enough sample plots exist to determine whether the deviations in the other units are significant. For the volume estimates themselves, the 95% confidence intervals for the overall block volumes are within 10% of the estimates as prescribed for the PHSP, but those for the individual treatment units are much wider. In some cases, the confidence limits for the pre- and post-treatment volume estimates overlap, meaning that the amount of volume removed was too small to detect with the measurement procedures.

Figures 3–5 display pre- and post-treatment diameter distributions for a sample of the units to compare and contrast the initial stand conditions and the effects of the different treatments. The dramatic overstorey reduction in Units B and G can be contrasted with Units A and H, in which some large trees and surrounding stems were retained, and the very light reductions in Units F and L. The number of larger trees in Unit H was more severely reduced than in the replicated treatment, Unit A. This is because the retained

TABLE 3 *Sampling statistics for pre- and post-treatment surveys*

Unit[a]	Survey	No. plots	Volume estimate (m^3/ha)	95% confidence limits (m^3/ha)	Actual % removal ($\bar{R}_v$)[b]
Mud Lake					
	Pre-treatment	77	176.6	164.6–194.3	
	Post-treatment	77	110.6	94.2–124.6	
A (35%)	Pre-treatment	15	164.6	116.3–212.8	30
	Post-treatment	15	97.6	58.3–137.1	(10–40)
B (50%)	Pre-treatment	14	207.1	180.2–233.9	63
	Post-treatment	14	67.5	44.8–90.1	(47–79)
C (20%)	Pre-treatment	11	170.5	131.0–210.0	21
	Post-treatment	11	145.5	92.3–198.9	(0–43)
D (0%)	Pre-treatment	11	174.0	123.5–224.0	0
E (50%)	Pre-treatment.	13	184.1	143.7–224.5	61
	Post-treatment	13	73.4	33.1–113.8	(37–85)
F (20%)	Pre-treatment	13	157.8	148.0–201.3	24
	Post-treatment	13	126.2	92.2–157.3	(5–43)
Opax Mountain					
	Pre-treatment	82	169.8	157.7–184.9	
	Post-treatment	82	106.0	94.5–119.1	
G (50%)	Pre-treatment	14	190.5	161.5–233.0	60
	Post-treatment	14	75.4	57.7–97.4	(51–69)
H (35%)	Pre-treatment	12	185.8	135.5–236.0	43
	Post-treatment	12	85.8	71.4–100.3	(24–62)
I (0%)	Pre-treatment	11	171.9	134.9–208.9	0
J (50%)	Pre-treatment	13	180.2	146.3–214.0	41
	Post-treatment	13	92.9	56.4–129.7	(13–69)
K (20%)	Pre-treatment	16	167.2	134.5–199.8	22
	Post-treatment	16	126.1	88.0–164.4	(3–41)
L (20%)	Pre-treatment	16	132.6	109.6–158.9	24
	Post-treatment	16	93.4	75.7–114.5	(9–3)

a Prescribed volume removal in brackets.
b The figure in parentheses represents the 95% confidence interval.

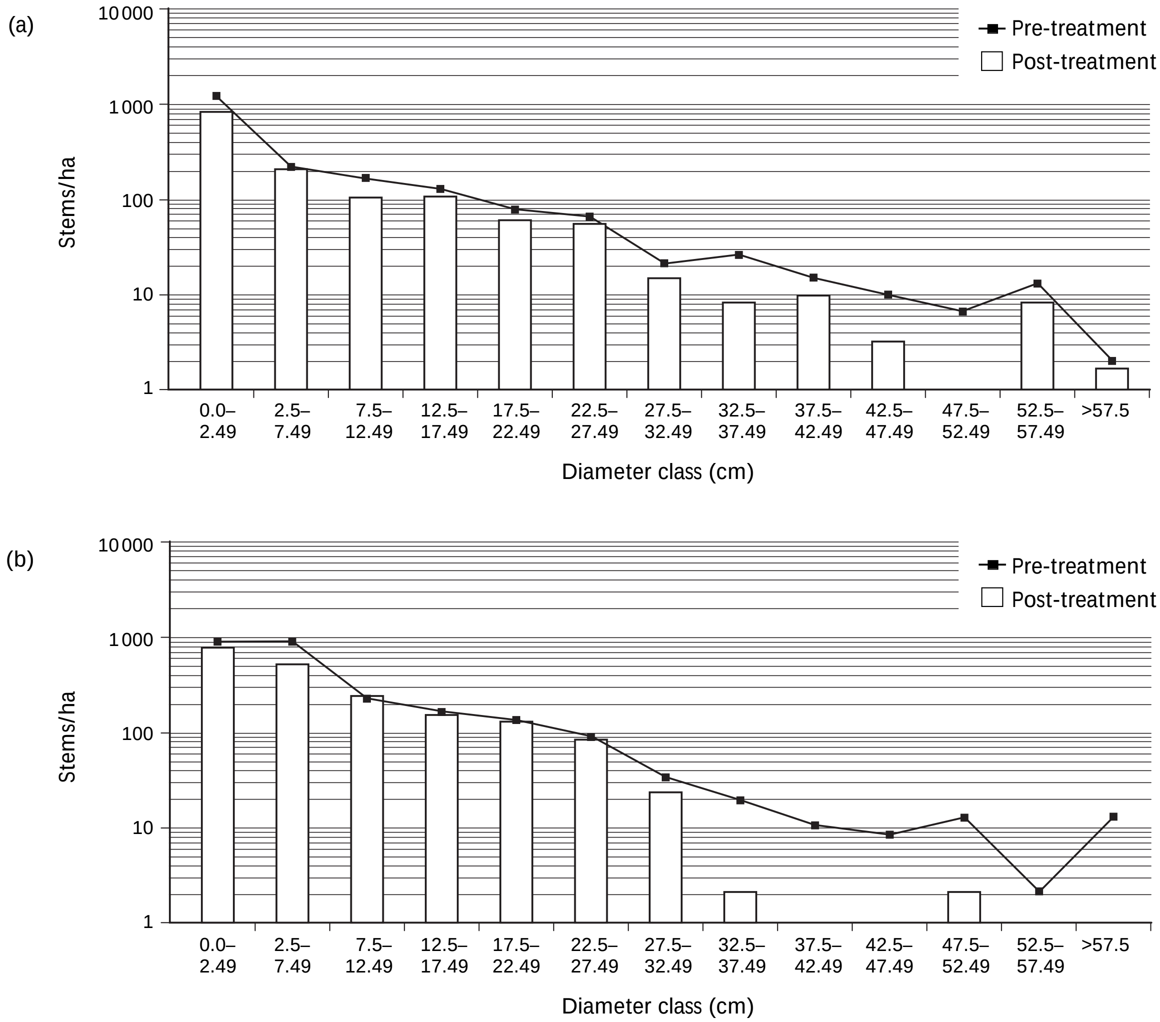

FIGURE 3 *Pre- and post-treatment diameter distributions for the 50% volume-removal treatments with patch retention: (a) Mud Lake block, Unit A; (b) Opax Mountain block, Unit H.*

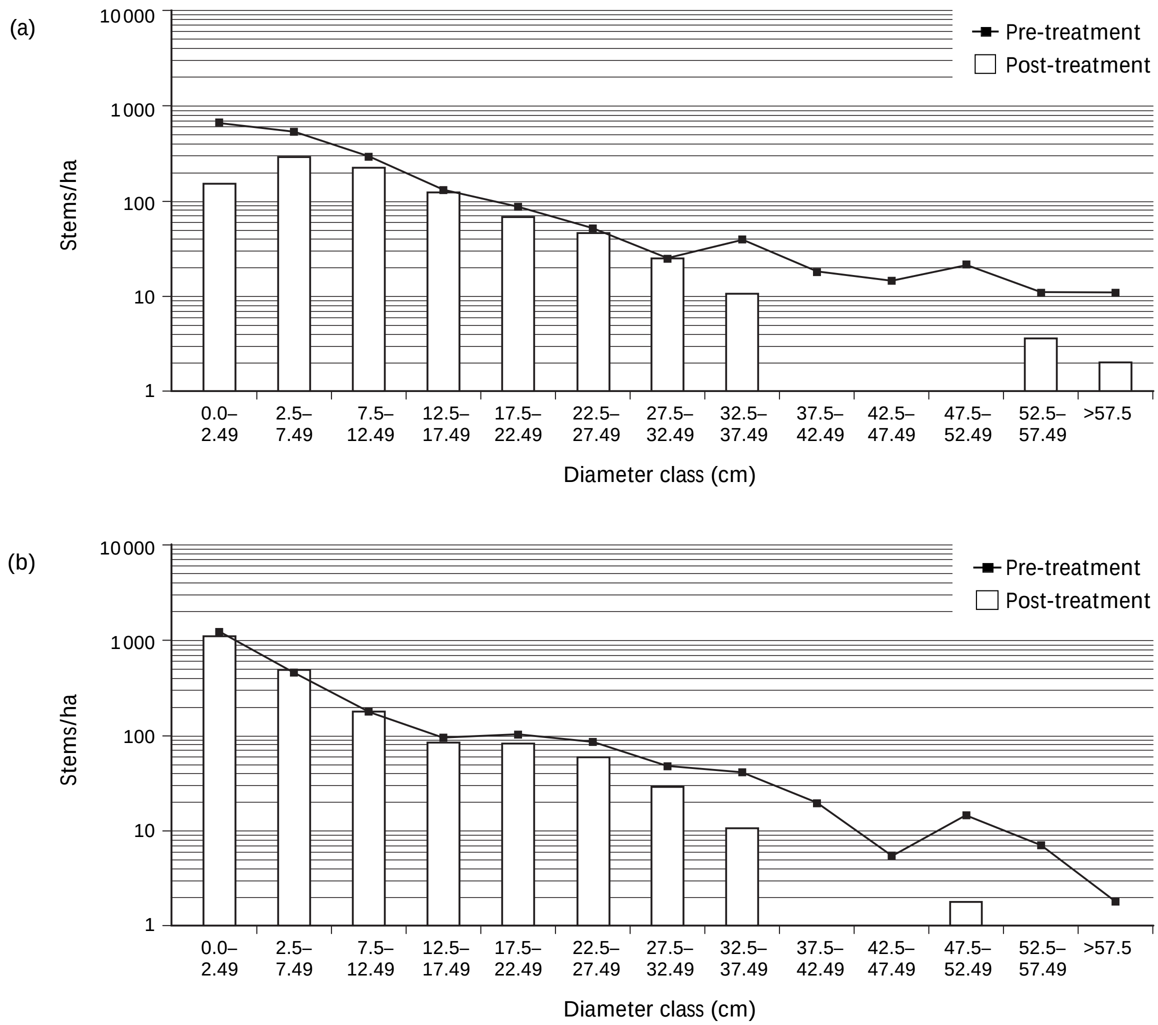

FIGURE 4 *Pre- and post-treatment diameter distributions for the 50% uniform volume-removal treatments: (a) Mud Lake block, Unit B; (b) Opax Mountain block, Unit G.*

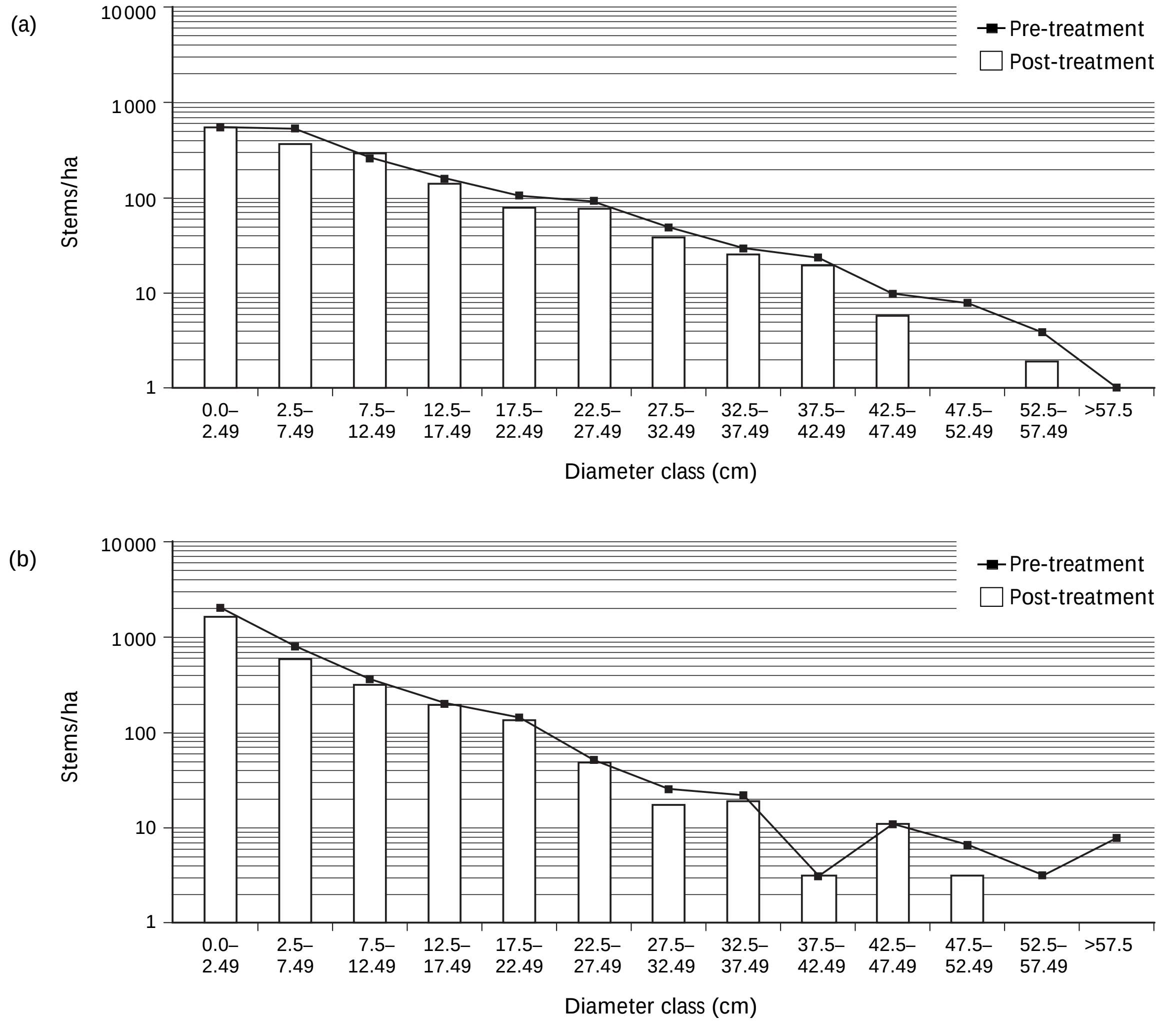

FIGURE 5 *Pre- and post-treatment diameter distributions for the 20% uniform volume-removal treatments: (a) Mud Lake block, Unit F; (b) Opax Mountain block, Unit L.*

green-tree patches in Unit H included more smaller-diameter lodgepole pine instead of larger Douglas-fir. In all three units, harvesting removed some of the trees under the diameter limits, but left a few that were over the limits, which suggests that at least some of the differences between the prescribed and actual volume removal (Table 3) are attributable to errors in marking and/or harvesting. Distributions for the patch-cut treatments (Units C and E) and control (Unit D) are not shown. These treatments did not alter the size-class distributions in the units, but simply changed the horizontal structure to a mosaic of forested and nonforested areas. The diameter distributions of the control Units D and I are similar to those of Figures 2a and 2b.

DISCUSSION

This examination of a silviculture systems trial highlights the difficulties in setting prescribed partial-cutting targets and measuring the effects of such treatments in patchy, multi-cohort stands. Several main points can be made:

- The initial cruise, although adequate for the PHSP requirements and for logging contract administration for the two blocks, was insufficient to reliably estimate the diameter limits and patch-cut dimensions required to achieve the target removals within each treatment unit. The consequences of this were not serious for the objectives of the trial because the approximate range of effect on the animal habitat was achieved. If this had been a growth-and-yield trial, however, each treatment unit would have required more intensive measurement. Increasing the within-treatment sample size to about 50 plots (7–13% sampling intensity) would have improved the initial stand volume estimates to within 10–15% of the true volumes, and allowed the more precise setting of marking targets. Using variable-radius plots could minimize the cost of this undertaking.
- Defining partial-cutting treatments as "percent volume removed" is problematic, not only because the same percent removal does not necessarily result in the same level of "treatment" if the growth of the residual stand is of interest, but also because "volume" is a somewhat variable stand-level attribute. Volume depends on merchantability limits, defect and decay reductions, and tree heights, all of which are difficult to measure accurately.
- Based on this experience, the problems associated with the use of "operational" partial-cutting trials to assess growth-and-yield responses to treatment are plain. Although considerable care was taken at Opax to choose similar sites and prescribe and control the harvesting in this research trial, measuring the growth of the treatment units will still be complicated by "noise" in the volume estimates and comparing the treatments will be hampered by site and structure differences. Time and cost constraints are usually more severe in operational settings and will limit the effort that can be applied to harvest supervision and pre- and post-treatment stand assessments. This study determined that a much larger sample size, at least 50 plots per 20-ha stand, would be needed at Opax simply to measure the standing volumes to within 10–15%. Differences in growth rates among the treatments that cannot be attributed to site differences would have to be greater than this amount to be detectable under these circumstances.

ACKNOWLEDGEMENTS

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REFERENCES

Forest Productivity Councils of British Columbia. 1995. Minimum standards for the establishment and remeasurement of permanent sample plots in British Columbia. B.C. Ministry of Forests, Inventory Branch, Victoria, B.C.

Husch, B., C.I. Miller, and T.W. Beers. 1972. Forest mensuration. 2nd ed. The Ronald Press Co., New York, N.Y.

Kozak, A. 1988. A variable-exponent taper equation. Canadian Journal of Forestry Research 18:1363–8.

Advanced Regeneration of Interior Douglas-fir Forests following Uniform Partial cut and Small Patch-cut Logging at Opax Mountain

JAANA KAIPAINEN, PASI PUTTONEN, AND ALAN VYSE

INTRODUCTION

Advanced regeneration is present in many of British Columbia's older undisturbed forest types (Weetman and Vyse 1990). When foresters decide to use advanced regeneration as part of their harvesting prescriptions, careful planning is required to meet legislated standards of quality and quantity. The crop potential of the advanced regeneration must be determined. Is the species suitable for the site under study? Are there sufficient stems, and are they in good condition, free of defects, and well shaped, and are they likely to respond vigorously to the flood of light, moisture, and nutrients when the overstorey is removed? If the answers to these questions are encouraging, then the logging conducted must attempt to protect the advanced regeneration.

Advanced regeneration has been used as the primary source of regeneration in dry Douglas-fir forests for 50 years or more (Glew and Cinar 1966; Vyse et al. 1991), but few if any studies of crop potential have been undertaken and none in relation to the different levels of overstorey removal. We have installed such a study at the Opax Mountain Silvicultural Systems Project site in the Kamloops Forest Region (Klenner and Vyse, these proceedings, p. 128), where our aim is to gain an understanding of the acclimation processes that lead to successful post-release growth of advanced regeneration in interior Douglas-fir forests. Our hypothesis is that pre-release height growth (e.g., 5-year growth), live crown ratio, tree size (small trees < 1.5 m respond quickest), and site series (site moisture) will explain most of the variation in post-release growth in interior Douglas-fir forests.

The broader objectives of our research are:

- to determine density, spacing, and spatial distribution of advanced regeneration before and after harvest;
- to follow the recruitment of germinants at the site;
- to identify pre-release variables that account for most of the variability in post-release tree growth;
- to gain an understanding of the processes leading to a successful post-release growth of trees; and
- to help in developing guidelines to predict the post-release growth of advanced regeneration.

In the remainder of this paper, we describe the status of advanced regeneration by treatments at the Opax research site, 3 years after harvesting. We report information on the number of germinants and the density, height and height distributions, height growth rate, base diameter distributions, live crown, stem form and tree vigour, and stocking status of advanced regeneration.

MATERIAL AND METHODS

Study Site

The research site is located southwest of McQueen Lake and Pass Lake about 15–20 km northwest of North Kamloops, B.C. It consists of two areas: one, on the southerly slopes of Opax Mountain, is at 1200–1370 m and includes an almost equal basal area of Douglas-fir and lodgepole pine; and the other is at Mud Lake (950–1100 m elevation) with almost 100% Douglas-fir. In both areas, the topography varies from steep slopes to depressions, and soil moisture ranges from submesic to hygric/subhydric.

Sampling Design and Plot Layout

The overall design at Opax Mountain provides a sampling population for advanced regeneration that consists of different site treatments and opening sizes. Six different treatment units were created at both Mud Lake and Opax Mountain in the winter of 1991/1992: two patch-cut areas, three uniformly logged areas, and one untreated control area. The patch-cut areas include both small group-selection cuts (openings of 0.1 and 0.4 ha) and small patch clearcuts (1.7 ha) with either low (20%) or high (50%) volume removal. In uniformly cut areas, the volume of removals was low (20%), high (50%), or moderate (50% removal with 25% of the area left as no-harvesting reserves).

The treatment units at the Mud Lake site belong to the Interior Douglas-fir (IDF) xh2 biogeoclimatic subzone in site classes 01 (Douglas-fir–pinegrass –feathermoss) and 05 (Douglas-fir/ponderosa pine–pinegrass). The treatment units at Opax Mountain are mostly in the IDFdk1 biogeoclimatic subzone, including site classes 01 (Douglas-fir/lodgepole pine–pinegrass–feathermoss) and 04 (Douglas-fir–pinegrass–yarrow). The topmost part of the northern Opax Mountain site is in the Montane Spruce (MS) biogeoclimatic zone.

Systematic sampling was used in the controls and uniformly cut areas. The aim was to collect data from 14–15 plots per area based on an initial, overall target of sampling 1% of the area. Plots were located along east–west lines that were either 100 or 120 m apart depending on the size of the treatment unit. The distance from plot centre to plot centre was also 100 or 120 m.

In the patch-cut areas sampled, openings were randomly selected from the population of openings in the treatment unit. One circular 10-m^2 plot was established in the estimated centre of the 0.1-ha opening. In the 0.4-ha openings, three plots separated by a distance of 32 m were established. Additionally, two plots were established outside the opening. Nine plots, 45 m apart, were systematically placed in 1.7-ha openings with four plots outside the opening at both north and south ends of the outermost plot line.

Systematic sampling was used because locating the sample units using this method is easier and cheaper than with random sampling. Systematic sampling is frequently used in surveys and inventories in British Columbia (B.C.

Ministry of Forests 1993). The sampling scheme was designed before the site series mapping was completed and thus site series (soil moisture) could not be used as a criterion for sample plot selection.

Measurements

Measurements were made in the summer of 1995. The circular plot size of 10 m^2 (radius of 1.78 m) has been shown as accurate for regeneration surveys. This is important because counting errors increase with an increase in plot size (Pohtila 1977). Each plot was marked with a steel stake and every tree taller than 10 cm was labelled with a metal number tag. For every plot, slope (%), aspect (degrees), and basal area (m^2) of the stand were measured and the amount of slash estimated. The altitude of individual plots was determined from the topographic map. A small subplot of 1 m^2 at the centre of the plot was established to count the number of germinants (i.e., seedlings < 10 cm in height). All were counted, but were not identified by species.

For every tagged tree (> 10 cm in height) the following measurements were taken: height (cm), leader growth for 1995 and 5 previous years' growth (accuracy of 0.5 cm), length and width of the live crown (cm), and diameter at the base and at breast height (mm). The distance to the plot centre and to the edge of the nearest tall tree from the tagged tree were measured along with the diameter of the tall tree. Stem form was classified according to the severity of deformity, and the crown shape was categorized according to the portion of the living crown. Information about microsite, damages, the colour of the needles, and the overall vigour of the tree was also collected.

In the openings, site preparation and planting was completed before the data collection in the summer of 1995, but did not take place in the uniform treatments until the spring of 1996. In the summer of 1996, all plots were surveyed for possible damage. No treatment damage was identified, but snow damage from the previous winter was evident.

Analyses

The sampling scheme provided data to formulate a model of post-release growth that is precise as well as site specific. Growth characteristics were analyzed using analysis of variance (ANOVA) for a completely randomized, split-plot design where the factors were: treatment (fixed, three levels [patch, uniform, control]), treatment unit (random, six levels), site series (fixed, two levels), and their interactions.

RESULTS AND DISCUSSION

General

The regeneration survey results from Mud Lake and Opax Mountain were analyzed separately because these sites were located in different subzones. For statistical reasons, the patch cuts (i.e., areas with different size openings) were treated as one area. It was not possible to compare the different opening sizes with uniformly cut areas and controls.

This study included 232 plots and 475 trees. Eighty percent of the measured advanced regeneration seedlings were at the Opax Mountain site and 20% at the Mud Lake site. The majority of measured trees (> 10 cm) at both sites were Douglas-fir: 93.7% at Opax Mountain and 92.5% at Mud Lake. At Opax Mountain, 3.4% of measured trees were lodgepole pine, while 2.9% were subalpine fir and interior spruce. At Mud Lake, 6.5% of seedlings were lodgepole pine and 1.1% interior spruce (Table 1). In the following statistical analyses, all tree species were combined.

Density of Advanced Regeneration

Generally, the Opax Mountain site was better stocked than Mud Lake. At Mud Lake, the density varied between 433 (±213; standard error of the mean) stems per hectare in treatment unit E (50% patch) to 1533 (±729) stems per hectare in unit A (35% uniform) (Figure 1a). In all treatment units, the number of advanced regeneration stems was over the minimum stocking standard, but only treatment units A (35% uniform) and F (20% uniform) were well stocked. The differences between the treatment units are not statistically significant ($P > F = 0.5271$).

The density of advanced regeneration at Opax Mountain was highest in the control unit (I) with 7688 (±2536) stems per hectare and lowest in the patch cut with 20% removal (K) with 640 (±244) stems per hectare (Figure 1b). Only treatment unit K had an advanced regeneration density that was below the target stocking standard, but the density was over the recommended minimum stocking standard. Although the density of the advanced regeneration varied considerably between treatment units, the differences were not statistically significant ($P > F = 0.0622$).

Most (94%) of the advanced regeneration was established in humus on relatively flat surfaces. Only a small proportion of seedlings was growing on mounds (5%) or fallen logs (1%).

Number of Germinants

Few seedlings less than 10 cm in height occurred in the study area, whether in the treatment units or the controls. Only two treatment units were observed to have some germinants: treatment unit F (20% uniform) had 143 (±143) germinants per hectare and unit K (20% patch) had 480 (±405) germinants per hectare. In both units, germinants were growing on mounds.

Height and Height Distribution

The average seedling height at Mud Lake was 0.99 (±0.085) m and across units varied between 0.48 (±0.05) m (35% uniform) and 1.87 (±0.27) m (20% uniform). At Opax Mountain, the average height was 0.53 (±0.03) m and varied from 0.41 (±0.05) m (20% uniform) to 1.00 (±0.13) m (35% uniform) (Figures 2a and 2b).

TABLE 1 *Number of plots and advanced regeneration seedlings measured in this study. See Klenner and Vyse (these proceedings) for a map and study area description.*

		Mud Lake						Opax Mountain					
Treatment		**Unit**	**No. plots**	**Fd[a]**	**Pl[b]**	**Other**	**Total**	**Unit**	**No. plots**	**Fd**	**Pl**	**Other**	**Total**
Control		D	14	12	1	0	13	I	16	118	0	5	123
Openings	20%	C	25	17	3	0	20	K	25	15	1	0	16
	50%	E	30	11	1	1	13	J	30	32	1	4	37
Uniform	35%	A	18	23	0	0	23	H	14	45	2	1	48
	50%	B	15	10	0	0	10	G	15	45	3	1	49
	20%	F	14	13	1	0	14	L	16	103	6	0	109
Total			116	86	6	1	93		116	358	13	11	382
Total for both sites			232	444	19	12	475						

a Fd = Douglas-fir
b Pl = lodgepole pine

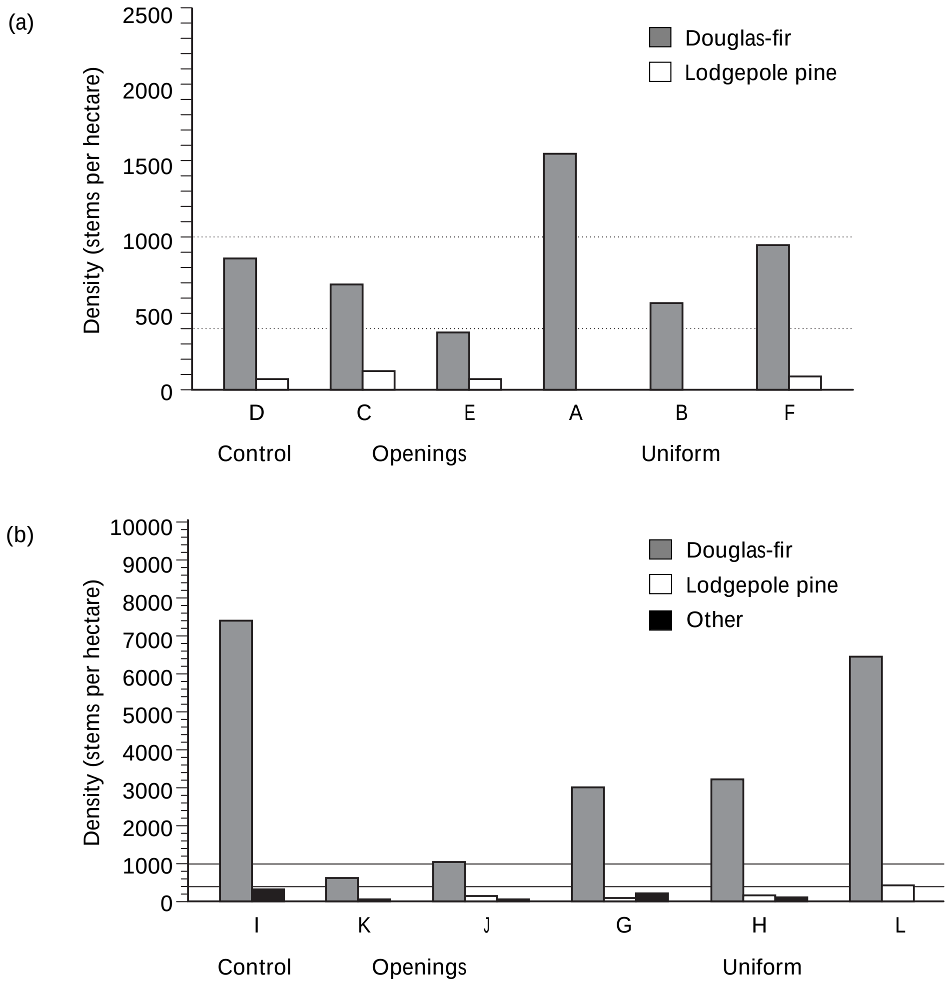

FIGURE 1 *The average density of advanced regeneration in different treatment units at (a) Mud Lake, and (b) Opax Mountain. The lines indicate the minimum (400 stems per hectare) and target (1000 stems per hectare) stocking standard. See Table 1 for descriptions of the treatments (A–L).*

The differences in average height between treatment units were not statistically significant ($P > F = 0.352$) at the Opax Mountain site. At Mud Lake, the seedlings in the uniformly logged unit with 20% removal (F) were significantly ($P > F = 0.005$) taller than seedlings in treatment units C (20% patch), E (50% patch), A (35% uniform), and B (50% uniform). The difference in the average height between the 35% uniform removal (A) and control (D) treatment units was also statistically significant ($P > F = 0.047$).

One-half of the seedlings at the Opax Mountain site were under 30 cm in height and most (75%) of them were under 60 cm (Figure 3a). At the Mud Lake site, seedlings were generally taller: only 10% of the seedlings were under 30 cm and 50% of the seedlings were under 50 cm in height (Figure 3b).

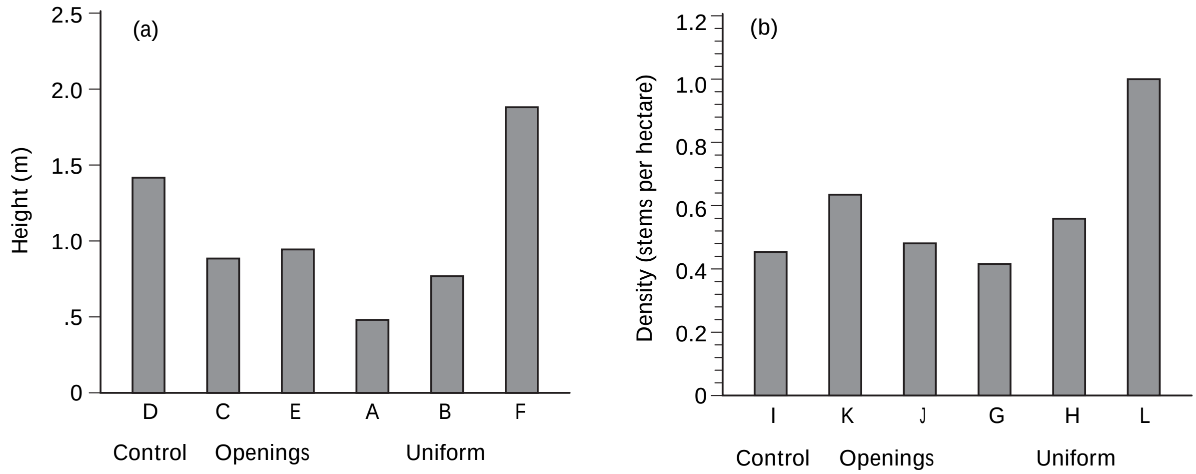

FIGURE 2 *The average height (m) in the treatment units at (a) Mud Lake, and (b) Opax Mountain. See Table 1 for descriptions of the treatments (A–L).*

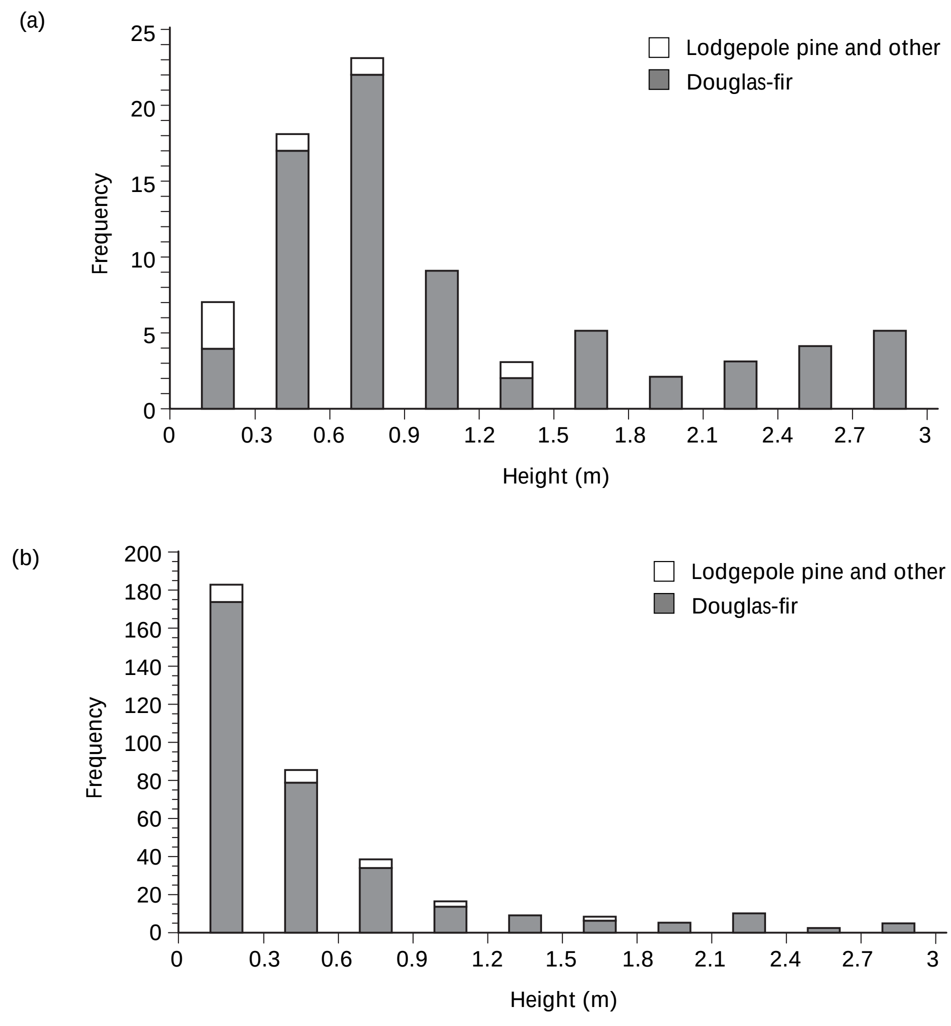

FIGURE 3 *The height distribution of advanced regeneration at (a) Mud Lake, and (b) Opax Mountain.*

Height Growth Rate

The height growth rate was measured for 1991–1995. The annual height growth varied between 2.15 (±0.10) cm and 3.27 (±0.18) cm at the Opax Mountain site, and from 3.80 (±0.41) to 7.29 (±0.64) cm at the Mud Lake site (Figures 4a–4l). The differences between treatment units were not statistically significant either at the Opax Mountain or Mud Lake sites. In both areas, the height growth was lowest in 1992 and highest in 1995. The differences between these years were statistically significant (Opax Mountain: $p = 0.0001$; Mud Lake: $p = 0.0003$). The increased growth rate is probably not entirely due to release because the growth rate has also increased in control units.

Diameter at Base and Diameter Distribution

The average diameter of the stem base was 9.0 (±0.7) mm at Opax Mountain and 21.4 (±2.5) mm at Mud Lake. At the Mud Lake site, the average diameter ranged from 11.4 (±1.3) mm (35% uniform) to 37.5 (±9.3) mm (20% uniform); at the Opax Mountain site, the diameter ranged from 6.4 (±0.7) mm (control) to 15.1 (±1.91) mm (35% uniform) (Figures 5a and 5b). The differences were not statistically significant ($P > F = 0.1261$ for Mud Lake; $P > F = 0.9996$ for Opax Mountain).

At Opax Mountain, the diameter at the base was under 5 mm for 40% of the seedlings and under 10 mm for 85% of the seedlings (Figure 6a). At Mud Lake, seedlings were generally bigger. Forty percent of the seedlings were thinner than 10 mm and 80% of the seedlings had diameters under 30 mm (Figure 6b).

Percentage Live Crown

Percentage of the living crown is a measure of the vertical proportion of a stem with live foliage. At Mud Lake, the percentage of living crown ranged from 67 (±7)% to 81 (±4)%, with an average of 74 (±2)%; at Opax Mountain this percentage ranged from 53 (±2)% to 67 (±3)%, with an average of 61 (±1)%. The differences between treatment units were not statistically significant ($P > F = 0.6704$ for Mud Lake; $F = 0.2813$ for Opax Mountain). According to a study on advanced regeneration of red fir (*Abies magnifica*) in California, trees with live crowns of 40% or more tend to grow more rapidly (Oliver 1986). In this study 90% of the seedlings at the Opax Mountain site and 95% of the trees at Mud Lake had more than 40% of living crown.

The size of opening in treatment units C, E, J, and K did not affect variability of the live crown ratio. Only in unit C was the difference between 0.4-ha and 1.7-ha openings relatively large. However, this variation in the live crown ratio is mostly due to small sample sizes.

Stem Form and Tree Vigour

Most of the trees at the Mud Lake site were assessed as having relatively good stem form: 42% had straight stems and 37% of stems had only minor sweeps (Table 2). In treatment unit F (20% uniform), the stem form distribution was somewhat worse: one-half of the stems had minor sweeps and 25% of stems had extreme sweeps. The stem form was generally poorer at the Opax Mountain site compared to Mud Lake: only 14% of stems were straight and almost 50% had minor sweeps. Twenty-six percent of the stems were assessed to have moderate sweeps and 14% extreme sweeps.

Vigour was also judged to be poorer at the Opax Mountain site, with only 15% of the seedlings assessed as having good vigour compared to 45% at Mud Lake (Table 3). At both sites, about one-third of the seedlings was poor in vigour.

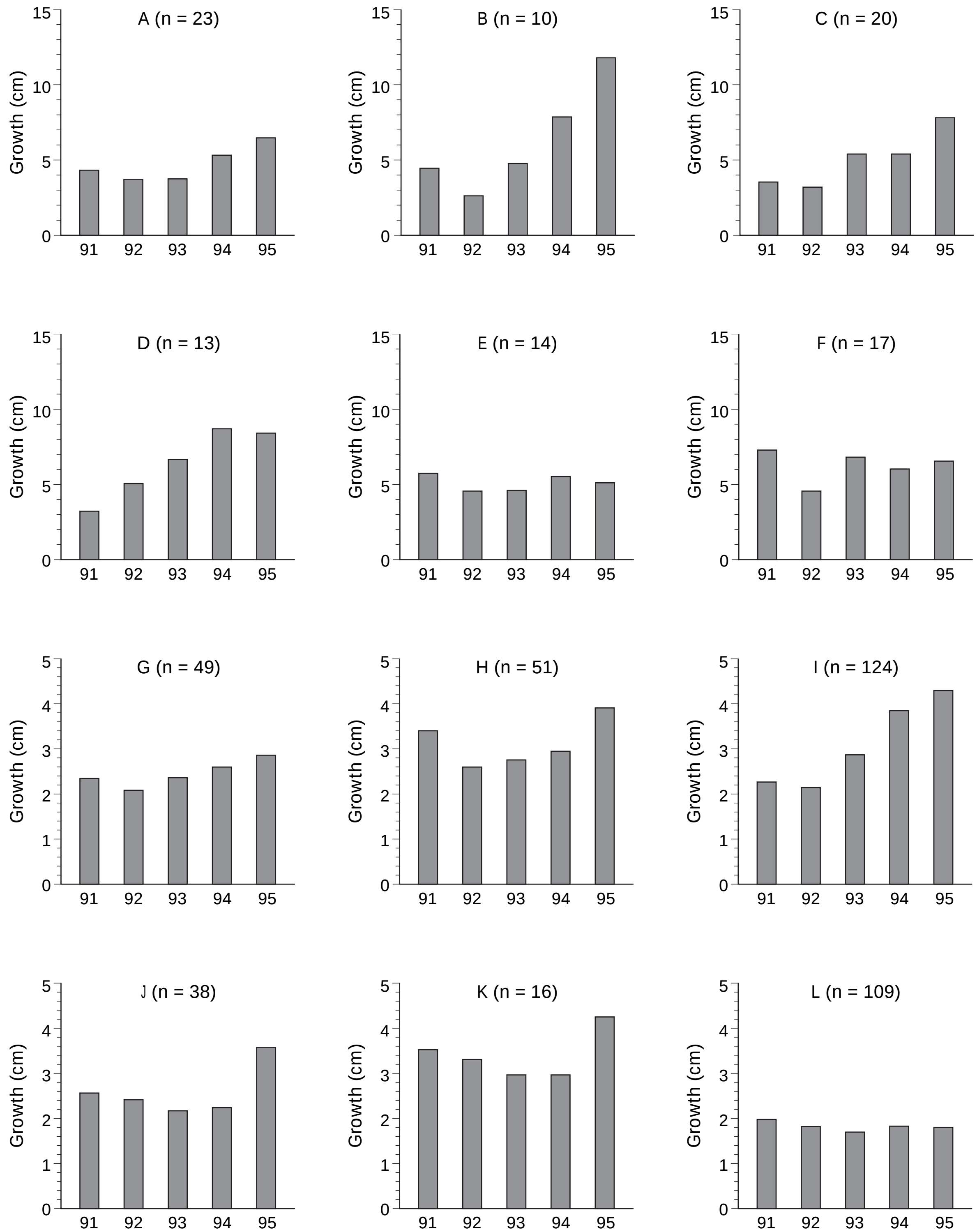

FIGURE 4 *The height growth rates in treatment units A–L during 1991–1995. See Table 1 for descriptions of the treatments (A–L).*

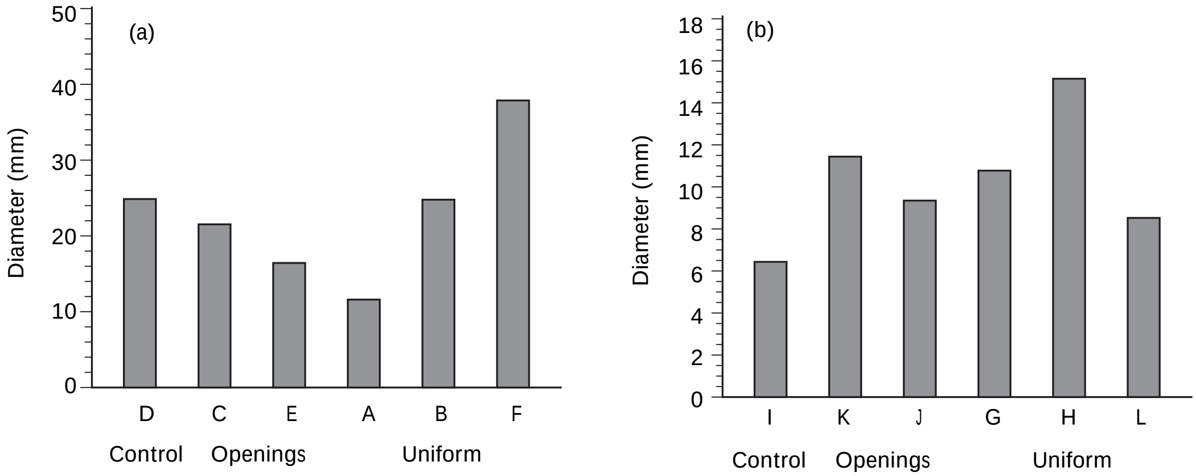

FIGURE 5 *The average diameter at base in different treatment units at (a) Mud Lake, and (b) Opax Mountain. See Table 1 for descriptions of the treatments (A–L).*

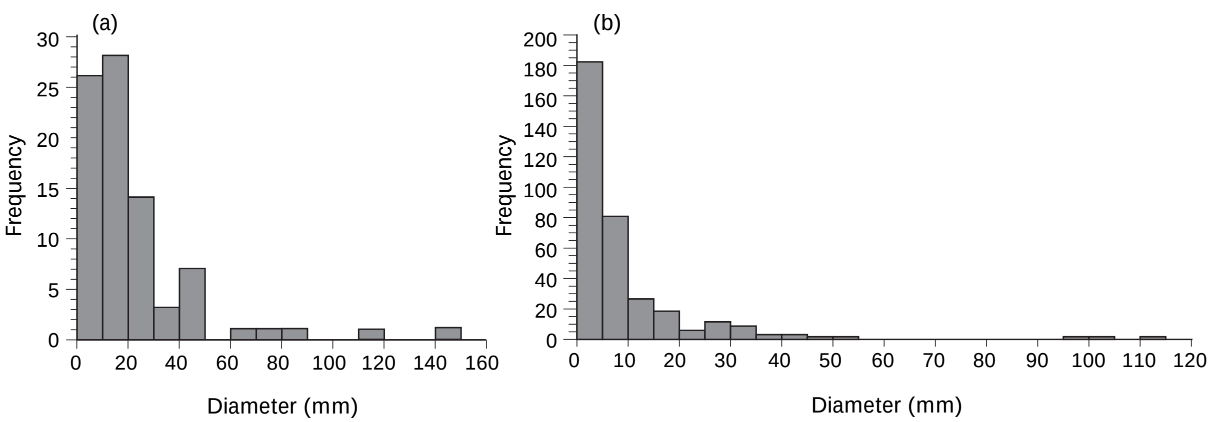

FIGURE 6 *The base diameter distribution of advanced regeneration at (a) Mud Lake, and (b) Opax Mountain.*

TABLE 2 *The number and percentage of seedlings in different stem form categories. See Table 1 for descriptions of the treatments (A–L).*

		Mud Lake					Opax Mountain				
Treatment		**Unit**	**Straight**	**Minor sweeps**	**Moderate sweeps**	**Extreme sweeps**	**Unit**	**Straight**	**Minor sweeps**	**Moderate sweeps**	**Extreme sweeps**
Control	No.	D	7	4	1	1	I	21	60	27	16
	%		53.8	30.8	7.7	7.7		16.9	48.4	21.8	12.9
Openings	No.	C	9	6	3	3	K	4	5	3	4
	%		42.9	28.6	14.3	14.3		25	31.3	18.8	25
	No.	E	6	4	4	0	J	5	12	13	8
	%		42.9	28.6	28.6	0		13.2	31.6	34.2	21.1
Uniform	No.	A	10	10	2	1	H	6	20	13	12
	%		43.5	43.5	8.7	4.3		11.8	39.2	25.5	23.5
	No.	B	5	4	1	0	G	5	24	13	7
	%		50	40	10	0		10.2	49	26.5	14.3
	No.	F	4	8	1	4	L	12	60	31	6
	%		23.5	47.1	5.9	23.5		11	55	28.4	5.5
Total	No.		41	36	12	9		53	181	100	53
	% of total		41.8	36.7	12.2	9.2		13.7	46.8	25.8	13.7

TABLE 3 *The number and percentage of seedlings in different tree vigour categories. See Table 1 for descriptions of the treatments (A–L).*

Treatment		Mud Lake					Opax Mountain				
		Unit	Good	Minor damage	Weakened	Poor	Unit	Good	Minor damage	Weakened	Poor
Control	No.	D	4	4	1	4	I	16	31	43	34
	%		30.8	30.8	7.7	30.8		12.9	25	34.7	27.4
Openings	No.	C	8	3	3	7	K	4	3	5	4
	%		38.1	14.3	14.3	33.3		25	18.8	31.3	25
	No.	E	7	1	2	4	J	11	7	13	7
	%		50	7.1	14.3	28.6		28.9	18.4	34.2	18.4
Uniform	No.	A	18	2	1	2	H	4	10	14	23
	%		78.3	8.7	4.3	8.7		7.8	19.6	27.5	45.1
	No.	B	6	2	2	0	G	1	10	11	27
	%		60	20	20	0		2	20.4	22.4	55.1
	No.	F	1	2	3	11	L	20	32	31	26
	%		5.9	11.8	17.6	64.7		18.3	29.4	28.4	23.9
	No.	All	44	14	12	28	All	56	93	117	121
	% to total		44.9	14.3	12.2	28.6		14.5	24	30.2	31.3

Stocking Status

When estimating stocking status, trees with extreme and moderate stem deformities, as well as trees with poor vigour, were excluded. About one-half of the advanced regeneration in the control units was judged as unacceptable, and no discernible pattern of reduced acceptability with intensity of harvesting was detected. (Figure 7).

The minimum stocking standard at both sites was 400 stems per hectare, with a target stocking standard of 1000 stems per hectare. The stocking status, based solely on the advanced regeneration, was best at Mud Lake in the 35% uniformly logged area (treatment unit A) with 1333 (±637) stems per hectare and worst in treatment unit E (50% patch), where only 300 (±153) stems per hectare were acceptable. At Opax Mountain, the maximum density

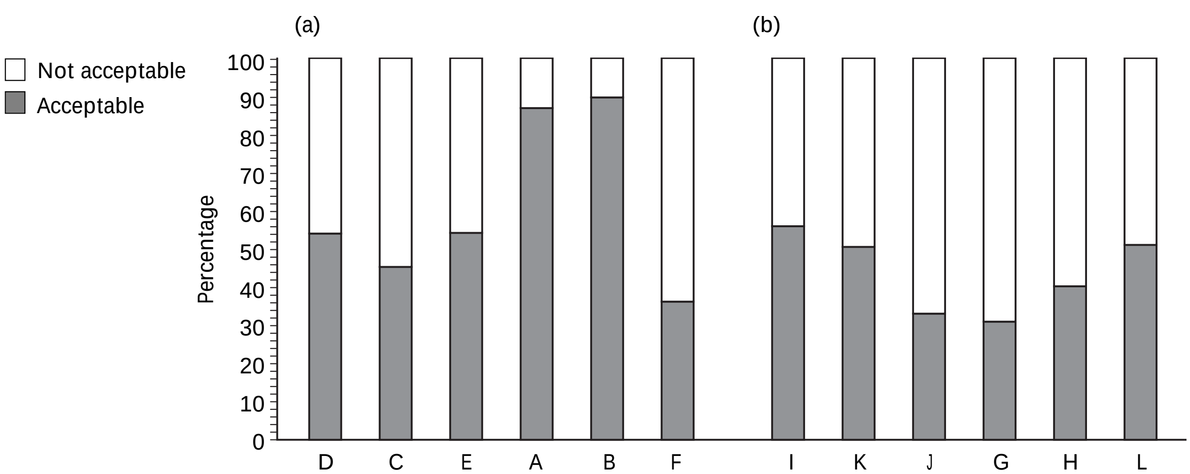

FIGURE 7 *The percentage of acceptable and not acceptable seedlings in different treatment units at (a) Mud Lake, and (b) Opax Mountain. See Table 1 for descriptions of the treatments (A–L).*

of acceptable seedlings was 4375 (±1546) stems per hectare in the control area (treatment unit I) and the minimum density 320 (±170) stems per hectare in treatment unit K (20% patches). Although the stocking status at Mud Lake was relatively poor in all the treatment units, only unit E (50% patch) could be judged as understocked. At the Opax Mountain site, only treatment unit K (20% patch) was understocked. In both of these understocked treatments, planting has taken place to ensure that stocking is sufficient to meet provincial standards.

Treatment units I (control) and L (20% uniform removal) at Opax Mountain had significantly ($P > F = 0.0223$) more acceptable seedlings than the rest of the treatment units (J, K, G, and H). At Mud Lake, the differences between treatment unit A (35% removal), C (20% patch), E (50% patch), and F (20% uniform) were statistically significant ($P > F = 0.0436$).

At Mud Lake, the effect of logging on advanced regeneration was not discernible, probably because the sample sizes were too small. At the Opax Mountain site, however, a relationship existed between the intensity of logging and the number of advanced regeneration.

CONCLUSIONS

Following logging, the advanced regeneration on the Opax Mountain Silvicultural Systems Project site is highly variable in amount and quality. About one-half of the regeneration is acceptable, and because release is evident, this regeneration is expected to form part of the future forest. The quantity of advanced regeneration is sufficient to meet minimum stocking standards in all but two of the patch-cut treatments; however, the amount is less than expected and less than ideal. The lower-elevation units at Mud Lake in the IDFxh2 subzone contain the least amount of regeneration. Seedlings less than 10 cm in height that could compensate for stocking deficiencies are currently scarce in the project area.

Logging damage is one possible explanation for the relatively low numbers of seedlings at Opax Mountain. There were considerably fewer stems in the patch cuts at the upper-elevation site where 100% of the merchantable trees had been removed, but this pattern was not evident at the lower elevation. Another possible explanation is that repeated defoliation by insects has decreased the quantity and quality of advanced regeneration. A severe outbreak of western spruce budworm (*Christoneura occidentalis*), which mines buds and feeds on newly emerging foliage of Douglas-fir, persisted in the Opax Mountain area for a decade or more until 1994. Dead advanced regeneration stems resulting from budworm defoliation are common in the project area, and especially at the Mud Lake site. The budworm outbreak would also have reduced conifer seedfall to zero in many years, thus contributing to the low numbers of seedlings less than 10 cm. Now that the outbreak of budworm is over, further regeneration can be expected.

British Columbia Ministry of Forests. 1993. Minimum standards for the establishment and re-measurement of permanent sample plots in British Columbia. Secretariat, Forest Productivity Councils of British Columbia, Victoria, B.C.

Glew, D.R. and S. Cinar. 1966. The results of stand treatment following seed tree marking in the dry belt fir types. B.C. Forest Service, Forest Management Division, Victoria, B.C. Note 5.

Oliver, W.W. 1986. Growth of California red fir advance regeneration after overstory removal and thinning. U.S. Department of Agriculture Forest Service, Berkeley, Calif. Research Paper PSW-180.

Pohtila, E. 1977. Taimiston inventoinnin tarkkuus. Summary: Accuracy of regeneration surveys. Communicationes Instituti Forestalis Fenniae 98.1.

Vyse, A., R.A. Smith, and B.G. Bondar. 1991. Management of Interior Douglas-fir stands in British Columbia: past, present and future. *In* Interior Douglas-fir: the species and its management. D.M. Baumgartner (editor). Washington State University, Department of Natural Resource Sciences, Pullman, Wash.

Weetman, G and A. Vyse. 1990. Natural Regeneration. *In* Regenerating British Columbia's Forests. D.P. Lavender et al. (editors). University of British Columbia Press, Vancouver, B.C. p. 118–30.

Assessment of Forest Insect Conditions at Opax Mountain Silviculture Trial

DAN MILLER AND LORRAINE MACLAUCHLAN

SITUATION OVERVIEW

Forest management in British Columbia requires that all resource values are considered along with a variety of appropriate management practices. For the past 100 years, partial-cutting practices were the method of choice when harvesting in Interior Douglas-fir (IDF) zone ecosystems. Along with a highly effective fire suppression program and minimal stand tending, these practices have created new and distinct stand structures. These range from low-density stands of uniform height to variable-density, multi-layered stands with patchy distributions of tree clumps and canopy gaps.

However, some management practices in IDF ecosystems have created ideal conditions for epidemics of insects and diseases, which are detrimental to both stand and landscape values. The Douglas-fir beetle (*Dendroctonus pseudotsugae*) is the principal bark beetle attacking mature Douglas-firs (Furniss and Carolin 1980). Timber losses attributed to the Douglas-fir mortality caused by this beetle were estimated at 2.4 million m^3 from 1956 to 1994. These losses occurred primarily in the province's Southern Interior (Humphreys 1995). Visual quality values associated with stands and landscapes can be strongly affected by the removal of the principal cover species, whether by clearcut activities or widespread tree mortality. By eliminating the mature component of Douglas-fir trees within a stand, bark beetles can ultimately affect mule deer by removing their winter cover and browse.

The risk of attack by the Douglas-fir beetle is determined by such stand attributes as age, species composition, size, and growth rate (B.C. Ministry of Forests and B.C. Environment 1995a). Epidemics of Douglas-fir beetles are generally associated with factors such as root disease, tree defoliation, and excessive amounts of logging slash and debris (Furniss and Carolin 1980; Humphreys 1995). The prevalence of all three factors is strongly influenced by forestry practices, which significantly influence all forest resources. Past practices of selective logging have increased the incidence and prevalence of root diseases such as *Armillaria ostoyae*, *Phellinus weirii*, and *Phaeolus schweinitzii* (B.C. Ministry of Forests and B.C. Environment 1995b).

Epidemics of Douglas-fir beetles have historically followed epidemics of the western spruce budworm (*Choristoneura occidentalis*), known widely as the most destructive forest defoliator in western North America (Furniss and Carolin 1980). Seven epidemics of western spruce budworm have been recorded in British Columbia since 1909 (Harris et al. 1985; Koot and Hodge 1992b). The last epidemic involved 800 000 ha at its peak in 1987, mostly in the Southern Interior of British Columbia (Koot and Hodge 1996).

Initiation and decline of budworm outbreaks in susceptible forests are influenced primarily by weather, and therefore fluctuate in an irregular and unpredictable fashion (Thomson et al. 1984). However, the duration, intensity, and frequency of outbreaks are also influenced by host quality and availability and natural enemy complexes (Schmidt 1985). Fire suppression and minimal stand-tending practices have created an abundance of high-quality host material in the form of multilayered canopies, and dense understorey and intermediate canopies (B.C. Ministry of Forests and B.C. Environment 1995c). Selective harvesting of ponderosa pine and exclusion of pine regeneration has created a predominance of Douglas-fir forests, with little species diversity

Repeated years of Douglas-fir defoliation by the western spruce budworm resulted in scattered tree mortality over large areas. Mortality of Douglas-fir during the 1970–1974 epidemic near Pemberton, B.C. was estimated at 39% in heavily defoliated stands (Alfaro et al. 1982). As well as timber losses attributed to tree mortality, stands suffered additional volume losses because of reduced radial and height growth, and stem deformities that arose from topkill (Van Sickle 1985; VanderSar 1987). After an infestation has collapsed, defoliated trees may require several years to regain a full complement of foliage and thereby re-acquire their growth potential. For example, up to 5 years after the collapse of the infestation, reduced growth was noted in the stand near Pemberton (Alfaro et al. 1982). The incidence of topkill can also be quite high. Topkill was found in 85% of 65 stands surveyed in the Vancouver and Kamloops forest regions, and 25% of all trees had crown topkill (Alfaro 1986). Losses over a 20-year period were estimated at 6–33% of total stand volume (Alfaro and Maclauchlan 1992).

Forest health has traditionally focused on situations involving excessive numbers of a few species, generally termed "pests." However, we should also consider the absence or reduction of other species, or guilds of insects. Ecological processes are highly dependent on the activities of insects. The biodiversity guidelines of the Forest Practices Code attempt to protect such groups through the "the development of an ecosystem management approach that provides suitable habitat conditions for all native species" (B.C. Ministry of Forests and B.C. Environment, Lands and Parks 1995d). At the landscape level, biodiversity management should consider:

- seral stage distribution;
- temporal and spatial patch size distribution;
- landscape connectivity;
- stand structure; and
- species composition.

The relative benefits of various options should be supported with an understanding of the insect fauna.

A silvicultural trial was established at Opax Mountain near Kamloops, B.C. in the IDFxh2 and IDFdk1 subzones to evaluate the effects of different partial-cutting regimes on such factors as growth and yield, biodiversity, forest health, and visual quality. Six treatments were allocated to each of two areas:

1. no-treatment unit;
2. low-density patch cuts;
3. high-density patch cuts;
4. 80% uniform retention;
5. 50% uniform retention; and
6. 35% island retention.

The overall goal for this and subsequent studies is to develop integrated forest management tactics and strategies for IDF forests that use silviculture as an integral component for creating stable, healthy forested ecosystems. Four research programs were conducted in 1996 at Opax Mountain to address the following objectives:

- Assess the risk from forest insects on the viability of the demonstration trial;
- Evaluate the general diversity of forest beetles associated with the trial; and
- Determine obvious correlations between insects and silvicultural treatments.

DIVERSITY OF ARBOREAL BEETLES

Forest health depends on a diversity of ecological processes, and also on the diversity of species that participate in such processes. Site productivity is closely linked with nutrient recycling, and therefore closely linked with a diverse assemblage of insect species. The most prominent processes in forested stands involve arboreal beetles such as bark and wood boring beetles. Arboreal species of beetles greatly influence stand structure and composition, and serve as the initiators of decomposition and nutrient recycling.

Therefore, methods are required to monitor patterns of insect diversity across landscapes, preferably ones that provide linkages with other resource sectors. A major focus of the Opax Mountain silviculture trial was to promote collaboration and interaction between various research sectors.
Our objectives were:

- to estimate baseline levels of beetle diversity in stands of Douglas-fir; and
- to develop a sound and comprehensive monitoring tactic for arboreal beetles.

Methods

Arboreal beetles can be monitored with baited multiple-funnel traps. On May 22–23, 1997, three replicates of three multiple-funnel traps (8-unit) per replicate were set within each of the six treatment blocks at both sites of the Opax Mountain silviculture trial, for a total of 108 traps. Each trap was suspended by rope between trees such that the trap bottle was 1.0–1.5 m above ground level. No trap was within 2 m of any tree. Each trap bottle contained a Vapona square (2 × 2 cm) to kill captured insects and prevent damage by insect predators.

Host compounds, such as ethanol and monoterpenes, attract a wide range of beetle species. Release devices containing monoterpenes and ethanol were used as attractants in the funnel traps. The following three bait treatments (based on published data and personal experience) were assigned randomly among traps within each replicate:

- ethanol + a-pinene;
- myrcene + terpinolene; and
- b-pinene + 3-carene.

Ethanol was released from a 25-cm black polyethylene pouch. Each monoterpene was released separately from closed 15-mL polyethylene screw-cap bottles.

Catches were collected every 7–14 days, starting on June 7, 1997 and ending on October 10, 1997. All catches were stored in a freezer until analyses during the winter months. Baits were replaced in August. All beetles were identified to family, genus, and species when possible. A reference collection was assembled and insect identifications were verified periodically at the Pacific Forestry Centre in Victoria.

Results and Discussion

During 1996, beetles in at least 38 families and 150 species were captured in the funnel-trap program at Opax Mountain (Table 1). Additional specimens have yet to be identified. Most species were in the Cerambycidae, Elateridae, Scarabeidae, Scolytidae, and Staphylinidae families.

Relative abundance of arboreal beetles varied by species, treatment block, and variant. Some species, such as *Spondylis upiformis* (Cerambycidae), *Thanasimus undatulus* (Cleridae), *Ampedus brevis* (Elateridae), *Serropalpus substriatus* (Melandryidae), and *Dendroctonus pseudotsugae* (Scolytidae), were very abundant. Other species such as *Byrrhus kirbyi* (Byrrhidae), *Podabrus piniphilus* (Cantharidae), *Anthonomus robustulus* (Curculionidae), and *Thymalus marginicollis* (Trogossitidae) were much less abundant with only a single specimen each. Although not exhaustive, these data do provide an initial baseline estimate of the diversity of arboreal beetles in dry stands of Douglas-fir.

A funnel-trap program with ethanol and monoterpenes seemed an effective tool for comparing stands. Further work is needed to improve the trapping program and elucidate correlations between trap catches and ecological processes. The utility of this method lies in its comparative approach to the study of stand and landscape features. It is not possible or feasible to assess all species and families of insects.

BLOWDOWN AND ASSOCIATED BARK AND WOOD BORING BEETLES

Blowdown events in 1995 and 1996 led to concern about and interest in bark and wood boring beetles at Opax Mountain. Infestations of the Douglas-fir beetle, *Dendroctonus pseudotsugae*, typically start with attacks on slash and windfall material, and can result in widespread patch mortality of large-diameter Douglas-fir. Our objectives were:

- to assess the distribution and categories of blowdown in all 12 blocks of the Opax Mountain silviculture trial; and
- to determine the use of windfall material by bark and wood boring beetles.

TABLE 1 *Relative abundance of arboreal beetles captured in baited multiple-funnel traps at Opax Mountain silviculture trial from May 23 to October 10, 1997 (n = 18)*

	Total number of beetles					
FAMILY (Species[a])	**Control**	**Patch (low)**	**Patch (high)**	**Retain 80%**	**Retain 50%**	**Retain islands**
ANOBIIDAE						
Microbregma emarginatum	5	5	7	5	4	3
BUPRESTIDAE						
Anthaxia aenogaster	2		1	3	1	
Buprestis langi (xh)			1			
Buprestis lyrata (xh)			1	1	7	1
Melanophila drummondi (xh)						
BYRRHIDAE						
Byrrhus sp.			1	1		
Byrrhus kirbyi (dk)				1		
CANTHARIDAE						
Podabrus piniphilus (dk)				1		
CARABIDAE						
Anisodactylus sp. (dk)			1			
Bembidion mutatum	3	5	8	1	2	
Cincindela longilabris (xh)						1
CEPHALOIDAE						
Cephaloon sp.	4	1	5	6	5	10
Cephaloon tenuicorne (dk)				1		
CERAMBYCIDAE						
Acmaeops proteus	4	3	5	3		1
Anastranglia sanguinea (xh)			1			
Anoplodera sexmaculata (xh)				1		
Cosmosalia chrysocoma (dk)		1	2	2		
Dicentrus bluthneri (xh)				1		
Leptura plagifera (dk)		1		1		
Megasemum asperum	22	28	7	2	32	5
Monochamus scutellatus				1	1	1
Neanthophylax mirificus	2	1	2	2	5	2
Neoclytus muricatulus (xh)						1
Phymatodes dimidiatus (xh)						1
Pygoleptura nigrella (dk)			1		2	
Rhagium inquisitor		2	1			
Spondylis upiformis	15	43	33	71	65	65
Strictoleptura canadensis	1	5	3	3		1
Tetropium velutinum	1			4	2	2
Trachysida aspera	1	1	7		1	2
Xestoleptura crassipes	1	1	3	1		
Xylotrechus longitarsis	3	18	9	9	3	8
CHRYSOMELIDAE						
Plateumaris pusilla (dk)						1
Syneta sp. (dk)	1	1	1			2
CLERIDAE						
Enoclerus lecontei				2		
Enoclerus sphegeus	22	7	18	29	33	10
Thanasimus undatulus	63	75	57	118	122	78
COCCINELLIDAE						
Coccinella septempunctata (xh)	1					

TABLE 1 *Continued*

FAMILY (Species[a])	Total number of beetles					
	Control	Patch (low)	Patch (high)	Retain 80%	Retain 50%	Retain islands
COLYDIIDAE						
Lasconotus complex (dk)	2		1			
Lasconotus vegrandis	1		1	3	1	
Oxylaemus californicus (xh)					3	
CRYPTOPHAGIDAE						
Antherophagus pallidivestis			1	1	2	
Atomaria quadricollis	1	2	1	2		
CUCUJIDAE						
Cucujus clavipes				7	6	5
Dendrophagus cygnaei			3	2	4	1
Leptophloeus alterans (dk)	3		1			1
Silvanus bidentatus (xh)	2					
CURCULIONIDAE						
Anthonomus robustulus (dk)				1		
Cossonus pacificus (dk)					1	
Magdalis gentilis (xh)	1					1
Pissodes sp.	13	13	14	22	6	6
Rhyncollis brunneus				2	2	
DERMESTIDAE						
Attagenus unicolor (xh)					1	
Megatoma sp.	25	6	5	11	13	11
Megatoma cylindrica	13	7	10	19	18	23
DYTISCIDAE						
Agabus sp. (xh)				1		
ELATERIDAE						
Acteniceromorphus umbricola	1		2	1	1	2
Agriotella occidentalis		1	1			2
Agriotes sparsus	2	1		3	1	
Agriotes tardus (dk)			2			1
Ampedus brevis	18	29	26	22	30	28
Ampedus glauca (xh)		1				
Ampedus moerens	2	9	3	9	16	3
Ampedus nigricollis	4	7	7	2	10	8
Ampedus nigrinus	7	5	7	4	6	3
Ampedus occidentalis		4		2	1	
Ampedus phoenicopterus		1	4	2	2	4
Ampedus pullus	3	1	10	3	6	3
Ampedus varipillis (dk)		1				
Ctenicera bombycina (xh)						1
Ctenicera propola	6	14	19	12	18	17
Ctenicera pudica	6	4	15	4	8	16
Ctenicera resplendens	1	1	7	3	3	1
Ctenicera rupestris	1	1	3		3	
Ctenicera silvatica (dk)						1
Dalopius sp.	1		2	1	4	
Danosoma brevicorne	1	2	6	4	4	1
Drasterius debilis	4	2	3	3	5	4
Eanus estriatus (dk)	1	1		7		
Limonius aeger	12	5	15	14	33	34

TABLE 1 *Continued*

FAMILY (Species[a])	Total number of beetles					
	Control	Patch (low)	Patch (high)	Retain 80%	Retain 50%	Retain islands
ELATERIDAE (continued)						
Pseudanostirus nebraskensis	20	21	28	17	11	15
Selatosomus aeripennis (xh)		2				
Selatosomus cruciatus			1		1	2
EROTYLIDAE						
Triplax dissimulator	1			1	1	1
HISTERIDAE						
Margarinotus umbrosus (xh)			1			
Paromalus mancus	1	2	1	6	1	3
Psiloscelis subopaca (xh)						
HYDROPHILIDAE						
Sphaeridium scarabaeiodes (xh)			2			
Tropisternus sp.				1	1	
LANTHRIDIIDAE						
Aridius nodifer	10	3	7	3	15	9
LEIODIDAE						
Catops egenus (dk)					1	
Catoptrichus frankenhauseri (xh)		1				
LYCIDAE						
Dictyopterus sp.	1	2	2	4	4	1
MELANDRYIDAE						
Melandryia striata (xh)		1		1		1
Phryganophilus collaris	1	1				
Serropalpus substriatus	284	159	69	50	348	128
Xylita laevigata	6	5	19	23	14	12
NITIDULIDAE						
Glischrochilus vittatus (dk)					1	
OEDEMERIDAE						
Calopus angustus (dk)	2				1	2
PYTHIDAE						
Pytho americanus (dk)			1	2		
RHIZOPHAGIDAE						
Rhizophagus sp.	4	11	2	8	6	6
SALPINGIDAE						
Rhinosimus viridiaeneus	1	1	2	4	5	6
SCAPHIDIIDAE						
Scaphisoma castaneum	4	6	5	3	4	7
SCARABEIDAE						
Aphodius sp.		2	4		1	1
Aphodius congregatus	5	6	12	6	10	8
Aphodius fimetarius	1	8	35	3	6	7
Aphodius fossor (dk)						2
Aphodius guttatus	3		1	1		
Aphodius leopardus		6	3	7	3	
Aphodius opacus	7	7	9	3	5	12
Aphodius vittatus	1	3	2	1		2
Dialytes ulkei (xh)			1			
Dichelonyx fulgida	11	17	11	4	3	6
Diplotaxis brevicollis	1		3	1	1	
Onthophagus nuchicornis		7	3	2	1	

TABLE 1 *Concluded*

FAMILY (Species[a])	Total number of beetles					
	Control	**Patch (low)**	**Patch (high)**	**Retain 80%**	**Retain 50%**	**Retain islands**
SCIRTIDAE						
Cyphon variabilis (dk)						3
SCOLYTIDAE						
Dendroctonus ponderosae			2	5	2	1
Dendroctonus pseudotsugae	172	131	311	234	185	234
Dendroctonus valens		5	4	13	3	1
Dryocoetes affaber	4		3	6	9	9
Dryocoetes autographus		1	3	2	4	4
Gnathotrichus retusus			5	9	3	3
Hylastes longicollis	81	281	111	211	39	104
Hylastes nigrinus	5	13	49	22	16	16
Hylastes ruber	2	9	9	8	19	11
Hylurgops porosus	64	129	195	253	197	101
Hylurgops rugipennis	11	13	12	16	22	10
Ips emarginatus (xh)						1
Ips latidens		1	4	2	5	1
Ips mexicanus (dk)			2	1		
Ips pini (dk)	1	1	6	3	9	
Pityogenes knechteli	1	1	12	7	6	1
Pityokteines minutus	2					
Pityophthorus sp.					3	5
Polygraphus rufipennis	1				1	3
Pseudohylesinus nebulosus				4	1	1
Scierus annectans	12	24	14	33	19	25
Scolytus monticolae	1	2	1	3		5
Trypodendron lineatum	249	9	322	25	32	23
SPHINDIDAE						
Odontosphindus denticollis (xh)				1		
STAPHYLINIDAE						
Lordithion sp.	6	5	3		6	3
Mycetoporus sp.	15	21	25	21	22	19
Oxyporus occipitalis (xh)		1	1			
Quedius sp.	17	21	40	40	35	17
Quedius giffinae (xh)			1			
Staphylinus pleuralis (dk)		1				
Stenus sp. (xh)			1	1		
Xenodusa reflexa (dk)		1				
TENEBRIONIDAE						
Corticeus sp. (dk)				1		
Corticeus strublei	9	9	15	14	12	7
TETRATOMIDAE						
Tetratoma concolor (dk)	1		1			
TROGOSITIDAE						
Calitys scabra		2	5	3	1	
Ostoma ferruginea			2	1		4
Temnochila chlorodia	1				1	
Thymalus marginicollis (xh)				1		

a Species found at both sites unless otherwise noted.

Methods

In 1996, a two-person crew conducted surveys of windfall material from August 12 to 15 and 19 to 21. All forested areas within patch-cut treatment blocks (C, E, J, and K) were assessed. The remaining blocks were assessed by a 10% (by area) sampling scheme. Parallel transect lines were spaced 100–150 m apart. Each transect was 10 m wide and 150–250 m long. We measured the length and diameter of all windfall material in blocks C, E, J, and K, and of all material encountered within 10-m transects in the remaining blocks. Only trees with stumps lying within the 10-m transect were included in density calculations for blocks A, B, D, F, G, H, I, and L. Leaning trees (> 30° from vertical) were distinguished from broken tops, high stumps, and uprooted trees.

The occurrence and prevalence of bark and wood boring beetles attacks were assessed for the stem or bole portion of each piece of windfall. Prevalence was estimated as a proportion of bark surface with frass piles and feeding galleries. Galleries were periodically examined for species identification. Round- and flatheaded wood borers (Buprestidae and Cerambycidae, respectively) were distinguished by coarse frass (fecal and cutting material), which consisted of white and reddish brown fragments. Ambrosia beetles (Scolytidae) were distinguished by fine white frass, while other scolytids produced fine to coarse reddish-brown frass. Species of *Dendroctonus* produced coarser frass than smaller beetles. Voucher specimens of bark beetles were collected for species verification.

Results and Discussion

The density of downed trees was low at the IDFxh2 site (Figure 1), with most of the area averaging 0–10 trees per hectare. The highest density of windfall at this site was found in the northern section of block F. Douglas-fir was the most abundant tree species in blowdown, accounting for 87% of 89 downed trees that were examined. Trembling aspen, paper birch, spruce, and lodgepole pine accounted for the remainder.

FIGURE 1 *Distribution of downed trees at (a) IDFxh2 (A–F) and (b) IDFdk1 (G–L) at the Opax Mountain silviculture trial: (A, H) 35% island retention; (B, G) 50% uniform retention; (C, K) low-density patch cuts; (D, I) no-treatment unit; (E, J) high-density patch cuts; and (F, L) 80% uniform retention.*

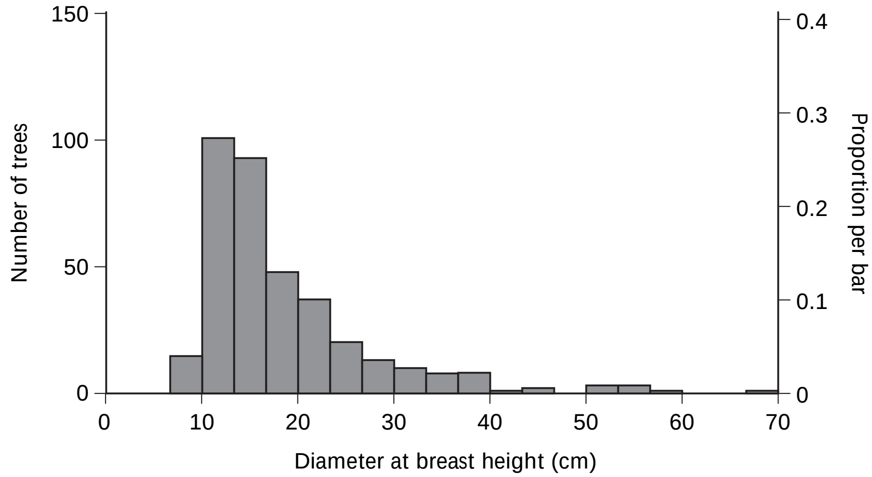

FIGURE 2 *Distribution of diameters of downed Douglas-fir at the Opax Mountain silviculture trial (n = 337).*

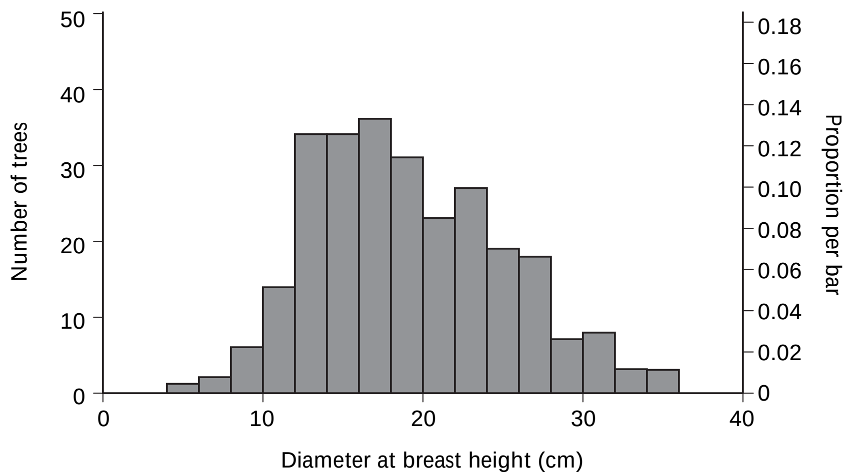

FIGURE 3 *Distribution of diameters of downed lodgepole pine at the Opax Mountain silviculture trial (n = 196).*

In contrast, blowdown densities were quite high at the IDFdk1 site, with most of the area averaging 10–40 trees per hectare. The highest density of windfall was in block I, followed by block J and block G. Both Douglas-fir and lodgepole pine trees were abundant in windfall material, accounting for 57% and 42% of 481 downed trees, respectively. The remaining downed material was spruce.

The diameter of downed Douglas-fir ranged from 7 to 68 cm at breast height, with a large component of small-diameter trees and few older veterans (Figure 2). In contrast, the diameters of downed lodgepole pine trees ranged from 6 to 36 cm at breast height, with considerably less variation (Figure 3). Some variation in mean diameter and height of downed Douglas-fir occurred among treatment blocks (Table 2). The largest-diameter downed trees were found in blocks C and F. Variation in tree height was less pronounced. Only trees with intact boles were used for height determination. Height and diameter of downed lodgepole pine were less affected by treatment blocks at the IDFdk1 site (Table 3).

TABLE 2 *Heights and diameters of downed Douglas-fir in each block of the Opax Mountain silviculture trial*

Block	Diameter (dbh) *n*	Diameter (dbh) Mean (± SE) (cm)	Height *n*	Height Mean (± SE) (m)
A	3	15 ± 0.9	2	11 ± 3.0
B	13	19 ± 1.9	7	16 ± 2.2
C	25	23 ± 2.8	12	13 ± 1.2
D	2	15 ± 4.5	1	6 ± 0
E	17	21 ± 3.4	13	15 ± 1.8
F	15	24 ± 2.1	15	16 ± 1.4
G	33	20 ± 1.4	26	15 ± 1.1
H	12	17 ± 1.6	9	11 ± 1.4
I	33	19 ± 1.6	23	12 ± 1.1
J	107	18 ± 0.8	80	13 ± 0.6
K	62	18 ± 1.2	44	12 ± 0.5
L	15	14 ± 0.7	12	9 ± 0.6

TABLE 3 *Heights and diameters of downed lodgepole pine in each block of the Opax Mountain silviculture trial*

Block	Diameter (dbh) *n*	Diameter (dbh) Mean (± SE) (cm)	Height *n*	Height Mean (± SE) (m)
G	28	21 ± 1.0	25	16 ± 0.8
H	4	21 ± 4.7	2	14 ± 5.5
I	19	21 ± 1.3	6	15 ± 2.3
J	104	20 ± 0.6	59	16 ± 0.7
K	28	20 ± 1.0	17	16 ± 1.0
L	10	22 ± 2.5	6	14 ± 1.6

Downed trees were classified into one of four categories: leaning, broken, uprooted, and felled. Uprooted and broken trees accounted for most of the blowdown (Figure 4). Leaners were defined as trees with root disturbance caused by wind or snowpress, with stems or boles deviating from vertical by more than 30°. Broken trees consisted of secure stumps with boles broken at heights greater than 1 m above ground level. Uprooted trees had most, if not all, of their root mass exposed, with the bole lying close to horizontal. Ten felled trees were encountered near patches 7–9 in block C. These trees had been felled by chainsaw, delimbed, and bucked into lengths of 2–3 m. Significant variation was evident in the relative proportion of the four types of blowdown across treatment blocks at both the IDFxh2 site (χ^2, df = 15, $p < 0.001$) and the IDFdk1 site (χ^2, $df = 10$, $p = 0.02$). The proportion of uprooted trees ranged from 0 to 94% (Table 4).

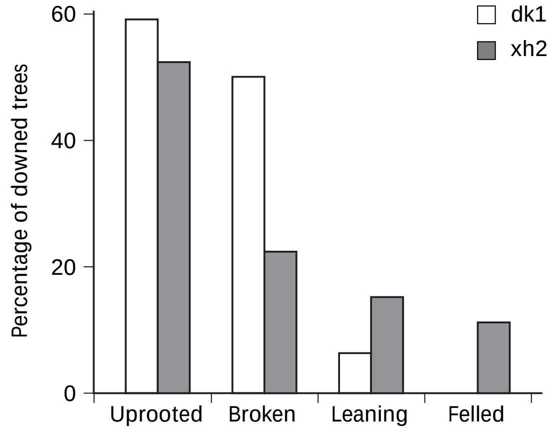

FIGURE 4 *Percentage of downed trees by type for IDFdk1 (n = 481) and IDFxh2 (n = 99) sites.*

The various categories of blowdown resulted in four types of large woody debris:

- leaning trees;
- uprooted trees;
- high stumps; and
- broken tree tops.

The latter two were derived from broken trees. Felling debris was not included because this was isolated to one area within block C. Uprooted trees, high stumps, and broken tops were the prevalent types of large woody debris for both Douglas-fir and lodgepole pine (Figure 5). Leaning trees made up only a small component of blowdown. The dimensions of each type are given

TABLE 4 *Percentage of the four categories of downed trees across all treatment blocks*

		Percentage of downed trees			
Block	***n***	**Uprooted**	**Broken**	**Leaning**	**Felled**
A	5	40	40	20	0
B	14	50	43	7	0
C	29	31	14	21	34
D	4	0	50	50	0
E	20	60	30	10	0
F	17	94	0	6	0
G	68	71	22	7	0
H	17	65	35	0	0
I	63	46	54	0	0
J	213	58	33	9	0
K	92	61	34	1	0
L	28	64	32	4	0
All	570	58	33	7	2

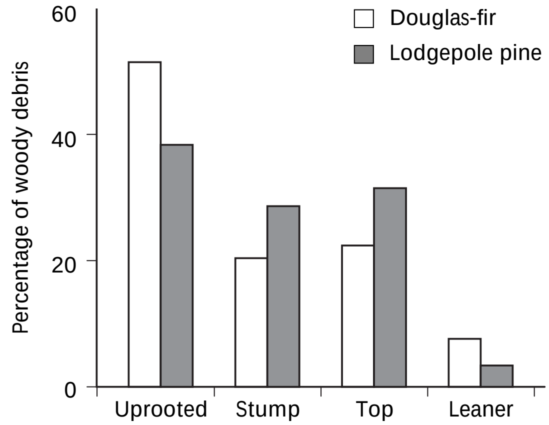

FIGURE 5 *Percentage of large woody debris by type for Douglas-fir (n = 423) and lodgepole pine (PL) (n = 286).*

in Table 5. We expect that bark and wood boring beetles to vary in their use of these types of woody debris.

Bark and wood boring beetles were abundant in large woody debris, using 40% of recent windfall (n = 734). Fourteen species of bark and ambrosia beetles were found in windfall. *Dendroctonus murrayanae, D. ponderosae, D. valens, Hylastes macer, Ips mexicanus, I. pini*, and *Pityogenes knechteli* were common in lodgepole pine material. Douglas-fir windfall was used by *D. pseudotsugae, Pseudohylesinus nebulosus*, and *Scolytus monticolae. Dendroctonus rufipennis* was found in one downed spruce. *Polygraphus rufipennis* was found in both spruce and Douglas-fir material. The ambrosia beetle, *Trypodendron lineatum*, was found in lodgepole pine and Douglas-fir material.

Round- and flatheaded wood borers (Cerambycidae and Buprestidae, respectively) were not identified by species because there were few adults in galleries. Most of the observed larvae in phloem tissue were flatheaded wood borers. Roundheaded wood borers such as Sawyer beetles were likely present in windfall, but not easily discernible because they would have invaded the sapwood at the time when surveys were conducted.

Forty percent of windfall items at the IDFdk1 site were used by beetles (n = 627), while 41% of material was used at the IDFxh2 site (n = 107). At both sites, broken tree tops had the highest incidence of attack, ranging from 59 to 60% of available tops (Figure 6). The lowest attack incidences were found in leaning trees. Attack incidences was also quite low in high stumps, ranging from 11 to 23%.

TABLE 5 *Dimensions of large woody debris at the Opax Mountain silviculture trial*

	Douglas-fir			Lodgepole pine		
	n	**Mean (± SE) diameter (cm)**	**Mean (± SE) height/length (m)**	***n***	**Mean (± SE) diameter (cm)**	**Mean (± SE) height/length (m)**
Uprooted	213	19.4 ± 0.64	13.4 ± 0.35	108	20.7 ± 0.62	16.0 ± 0.47
High stump	83	18.6 ± 1.04	4.5 ± 0.27	79	20.6 ± 0.59	5.5 ± 0.34
Broken top	68	15.2 ± 0.86	8.7 ± 0.35	79	16.2 ± 0.51	10.4 ± 0.39
Leaner	31	14.2 ± 0.91	10.2 ± 0.51	9	18.5 ± 1.74	4.85 ± 1.63

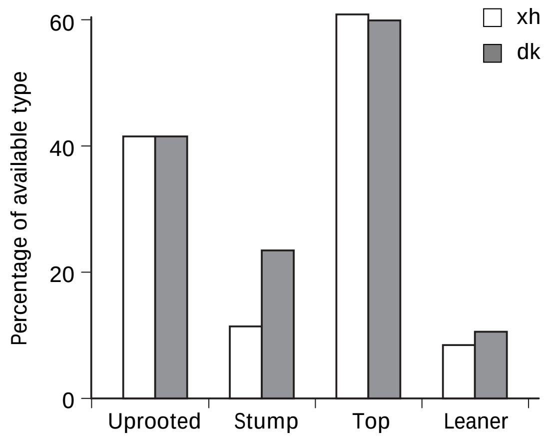

FIGURE 6 *Percentage of all available woody debris attacked by bark and wood boring beetles at two sites at the Opax Mountain silviculture trial.*

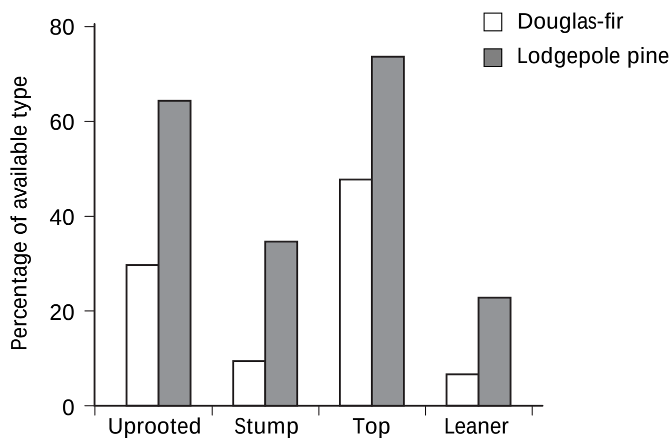

FIGURE 7 *Percentage of all available Douglas-fir and lodgepole pine woody debris attacked by bark and wood boring beetles.*

Beetles attacked much more of the available lodgepole pine windfall debris than the available Douglas-fir debris for all types of debris (Figure 7). Very low proportions of uprooted trees and stumps were used by species such as Douglas-fir beetles. The lowest use of Douglas-fir was found in stumps and leaning trees. Overall the incidence of attack was highest in lodgepole pine with 57% of woody debris infested by beetles ($n = 286$). By contrast, only 30% of Douglas-fir windfall was attacked by beetles ($n = 433$). Only 3 of 13 pieces of spruce windfall were used by beetles.

For all types of woody Douglas-fir debris, larger-diameter material was preferred by bark and wood boring beetles (Table 6). However, beetles showed little, if any, preference for larger-diameter lodgepole pine windfall, although large-diameter uprooted lodgepole pine trees showed a slight increase in use.

The most common species of bark beetle in infested downed Douglas-fir material was *Scolytus monticolae*, which favoured uprooted trees and broken tops (Figure 8). The Douglas-fir beetle, *Dendroctonus pseudotsugae*, was present in approximately 40% of windfall items infested with beetles,

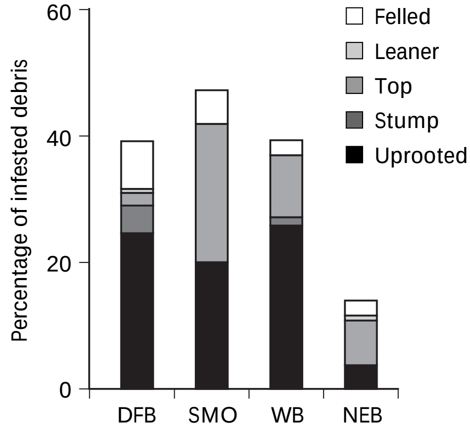

FIGURE 8 *Percentage of infested Douglas-fir blowdown by beetle species:* Dendroctonus pseudotsugae *(DFB),* Scolytus monticolae *(SMO), flatheaded wood borers (WB), and* Pseudohylesinus nebulosus *(NEB).*

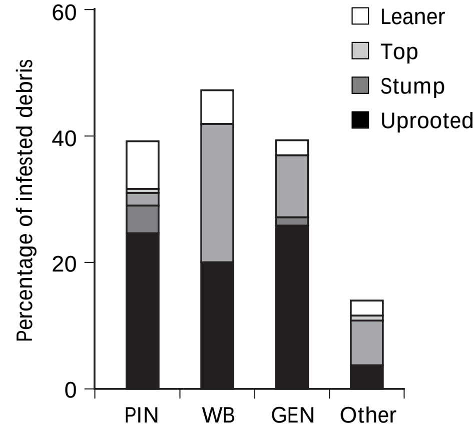

FIGURE 9 *Percentage of infested lodgepole pine blowdown by beetle species:* Ips pini *(PIN), flatheaded wood borers (WB),* Pityogenes knechteli *(GEN), and the combination of* Dendroctonus valens*,* D. murrayanae*,* Ips mexicanus*, and* Trypodendron lineatum *(Other).*

particularly in uprooted trees. Flatheaded wood borers were common in infested uprooted trees and tree tops.

Well over 50% of infested lodgepole pine debris contained the pine engraver, *Ips pini* (Figure 9), which favoured uprooted trees and broken tops. These same types of woody debris were favoured by flatheaded wood borers and the bark beetle, *Pityogenes knechteli*. Stumps were also used by the pine engraver, as well as other bark beetle species, such as *Dendroctonus valens, D. murrayanae*, and *I. mexicanus*.

Infested pieces of spruce windfall were used by four species of bark beetles: *Dendroctonus rufipennis*, *Ips tridens*, *Dryocoetes affaber*, and *Polygraphus rufipennis*. However, only three pieces were attacked.

The exploitation of available bark surface on infested blowdown varied considerably between types of debris (Figure 10). The phloem tissue in most

TABLE 6 *Diameters of used windfall pieces*

		Mean (± SE) diameter at base (cm)[a]			
Host	**Type**	***n***	**Used**	***n***	**Not used**
Douglas-fir	Uprooted trees	63	27 ± 1.4 a	150	16 ± 0.5 b
	High stumps	8	29 ± 7.5 a	75	18 ± 0.8 b
	Broken tops	39	17 ± 1.3 a	29	13 ± 0.9 b
	Leaners	2	24 ± 13.9 a	29	14 ± 0.4 b
Lodgepole pine	Uprooted trees	70	22 ± 0.7 a	38	17 ± 1.1 b
	High stumps	27	22 ± 1.1 a	52	20 ± 0.7 a
	Broken tops	59	16 ± 0.5 a	20	17 ± 1.3 a
	Leaners	2	17 ± 3.3 a	7	19 ± 2.1 a

a Means followed by the same letter within the same row are not significantly different at $p = 0.05$ (two-sided t-test, pooled variances).

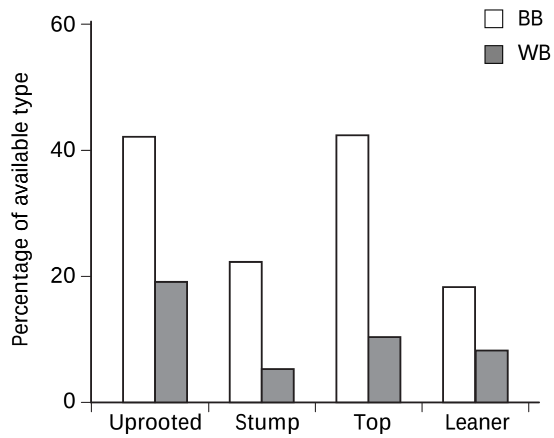

FIGURE 10 *Percentage of bark surface area of infested blowdown exploited by bark (BB) and wood boring beetles (WB) at the Opax Mountain silviculture trial.*

pieces was rarely exploited fully by beetles. Less than half of the bark surface in infested uprooted trees and broken tops was used by bark beetles; wood borers used 10–20% of the available area. Stumps were only lightly used by wood borers. Generally, beetles preferred to concentrate attacks at the butt end of trees and stumps rather than being loosely dispersed over the entire item. One significant difference existed between sites: mean (± SE) exploitation of bark surface of infested uprooted trees was 43.8 ± 3.1% at the IDFdk1 site, but only 29.7 ± 6.9% at the IDFxh2 site. Values for other types of downed material were similar.

The exploitation of bark area in infested Douglas-fir and lodgepole pine windfall material was relatively the same (Table 7). Bark beetles generally used less than 50% of the available phloem; wood borers generally used less than 20%. The only difference was in the use of lodgepole pine stumps where use by both bark beetles and wood borers was lower than in Douglas-fir stumps.

The Douglas-fir beetle, *Dendroctonus pseudotsugae*, used 50% of the available bark surface area of infested Douglas-fir windfall (Table 8). Pieces attacked by *D. pseudotsugae* were usually of large diameter and of long length compared to pieces attacked by other species. The pine engraver, *Ips pini*, used 55% of available area of infested lodgepole pine windfall. All species of beetles in pine seemed to use pieces of windfall with the same characteristics.

TABLE 7 *Exploitation of bark surface area by beetles in infested Douglas-fir and lodgepole pine*

		Mean (± SE) percentage of available bark exploited		
Host	**Type**	***n***	**Bark beetles**	**Wood borers**
Douglas-fir	Uprooted trees	63	40 ± 4.1	18 ± 3.1
	High stumps	8	41 ± 14.7	16 ± 12.8
	Broken tops	45	39 ± 4.1	10 ± 2.5
Lodgepole pine	Uprooted trees	70	43 ± 4.3	18 ± 2.8
	High stumps	27	16 ± 4.2	8 ± 5.1
	Broken tops	65	44 ± 4.6	10 ± 2.4

TABLE 8 *Exploitation of available bark surface by various species of beetles for Douglas-fir and lodgepole pine windfall*

Host	Beetle	Mean (± SE) % of surface area used by beetles	Mean (± SE) diameter of infested material (cm)	Mean (± SE) height/length of infested material (m)
Douglas-fir	*D. pseudotsugae*	49 ± 3.9	33 ± 1.9	18 ± 1.0
	S. monticolae	37 ± 2.9	19 ± 1.0	13 ± 0.8
	P. nebulosus	40 ± 5.6	24 ± 2.6	14 ± 1.1
	Flatheaded borers	33 ± 3.0	25 ± 1.3	15 ± 0.8
Lodgepole pine	*I. pini*	55 ± 3.3	20 ± 0.5	13 ± 0.6
	P. knechteli	37 ± 4.4	18 ± 1.2	12 ± 1.0
	Flatheaded borers	32 ± 2.9	19 ± 0.7	14 ± 0.6

Fresh woody material varied considerably at the Opax Mountain trial sites. Some sites had very little windfall, while others, such as blocks I and J, had large numbers of downed trees. Some trees were totally uprooted, while others had snapped at mid-bole, which resulted in an abundance of high stumps still firmly planted in the ground. Less than one-half of the pieces of woody debris were used by beetles. Of those, less than one-half of the available bark surface area was attacked. Phloem and sapwood resources were still available for beetles in 1997. In particular, high stumps and leaning trees may not be infested for several years.

STANDING DOUGLAS-FIR TREES INFESTED WITH DOUGLAS-FIR BEETLE

The Douglas-fir beetle, *Dendroctonus pseudotsugae*, is a significant factor in stands of Douglas-fir, affecting stand structure, nutrient cycling, and biodiversity. Spot infestations scattered over landscapes are common at endemic population levels (Furniss and Carolin 1980). Typically, groups of trees are attacked by beetles, generally in association with slash or windfall, root diseased trees, or trees defoliated by either the western spruce budworm or the Douglas-fir tussock moth, *Orgyia pseudotsugae*. The objective of the assessment was to determine the distribution of Douglas-fir recently attacked by the Douglas-fir beetle.

Methods

Assessments were conducted from August 12 to November 11, when sites were surveyed for blowdown and topkill. At that time, transects were run throughout all blocks. Patch-cut blocks were again surveyed on November 11–12, specifically to find infested trees. Each attacked tree was examined for brood production.

Results and Discussion

A total of 16 mature Douglas-fir were attacked by the Douglas-fir beetle in 1996 at the Opax Mountain trial site, mostly in blocks A and C. One spot infestation was found within island 6 of block A and consisted of two dead trees (dbh = 43 and 52 cm, respectively) as well as six trees with current brood (mean dbh ± SE = 45 ± 3.8 cm). Douglas-fir beetles had emerged from the dead trees, although associated beetles such as flat- and roundheaded wood borers were still present. Attacks on three of the attacked trees seemed to be partial. These trees will likely survive to 1997, but will probably be re-attacked by emerging Douglas-fir beetles in the spring.

A second spot infestation was found along the eastern edge of patch 15 in block C. The spot consisted of five trees with current brood (mean dbh ± SE = 51 ± 6.2 cm), two trees previously infested with beetles (dbh 31 and 80 cm, respectively), and two trees unsuccessfully attacked by Dougflas-fir beetles (dbh 27 and 32 cm, respectively). Several trees appeared to have been used in successive years for brood production. Single infested trees were found at the following locations:

- between patches 10 and 11 in block C (dbh 41 cm);
- along the southern margin of patch 14 in block C (dbh 43 cm);
- at the southwest corner of patch 6 in block E;
- in the northwest sector of block F; and
- along the northern margin of patch 12 in block J.

One additional Douglas-fir was killed in 1996 along the southern margin of patch 12 in block J (dbh 15 cm). The tree had been infested by two secondary types of bark beetles, *Scolytus monticolae* (= *S. tsugae*) and *Pseudohylesinus nebulosus*, presumably after the tree was weakened or killed by other factors. These species do not attack and kill vigorous, healthy trees.

It is very likely that additional tree mortality will occur in 1997. The combination of weakened trees, beetle populations arising from infested trees, and beetle populations arising from infested blowdown will result in risk of additional attacks on healthy trees in some areas, such as along the eastern margin of patch 15 in block C, within retention island 6 in block A, and in blocks J and K. Forty windthrown Douglas-fir were attacked by Douglas-fir beetles in 1996, mostly in blocks C, G, J, and K. The diameter of 16 trees was greater than 35 cm; eight were greater than 45 cm. This material will contribute a large number of beetles to the area. Large amounts of blowdown were not used by beetles. Standing stumps (4–6 m in height) and wind-weakened trees will be favourable for attacks by Douglas-fir beetles. With sufficient beetle numbers, attacks on adjacent trees will also occur, resulting in spot infestations.

However, endemic levels of Douglas-fir beetles are generally quite high, since they breed in small amounts of blowdown and in trees attacked previously. Widely scattered spot infestations are a normal landscape feature for endemic populations of Douglas-fir beetles (Furniss and Carolin 1980). Several infested trees appear to have been attacked in successive years, which has resulted in lower populations than if the trees were used within only one year. Moreover, the beetles are spread over a large area (> 120 ha at each site). Tree mortality should be assessed again in 1997.

DEFOLIATORS AND TOPKILL OF DOUGLAS-FIRS

Opax Mountain is located in a region with a high risk of severe Douglas-fir defoliation by western spruce budworm and Douglas-fir tussock moth (B.C. Ministry of Forests and B.C. Environment 1995c). However, little significant defoliation of Douglas-fir by either species occurred at Opax Mountain in 1996. Evidence of past epidemics was clearly discernible by the numerous trees with topkill spires.

The last infestation of western spruce budworm at Opax Mountain began in 1984. This outbreak resulted from widespread infestations that had originated near Lillooet and Cache Creek and had spread eastward throughout the Kamloops Forest Region (Erickson and Loranger 1987, 1988; Koot and Loranger 1989, 1990; Koot and Hodge 1991, 1992a, 1992b, 1993, 1994). Defoliation occurred over a 9-year period until the infestation collapsed in 1993. The intensity of defoliation was heavy in some years, although it was minimal in 1989 and 1990. An infestation of Douglas-fir tussock moth occurred during 1990–1992 (Koot and Hodge 1993, 1994).

The objective was to assess the prevalence and characteristics of Douglas-fir with topkill at the IDFxh2 site. Little, if any, topkill was evident at the IDFdk1 site. Infestations of the Douglas-fir tussock are more typical in IDFxh sites than IDFdk ones (Erickson 1995).

Methods

Trees in blocks A–F were assessed for topkill October 25–27 and November 10–11, 1996. In each block, 50–75 trees were randomly selected at preset intervals ranging from 25 to 35 m, depending on predicted lengths of transect lines. Transect lines in blocks C and E were placed throughout the residual forested corridors. In the remaining blocks, parallel transect lines were spaced 100–150 m apart with orientations similar to those used in the slash survey.

The height of live and dead stem was determined for each tree, as well as the percentage of live stem occupied by live crown. Stem biomass (SBM) volumes were calculated by the following equation:

$$\text{SBM} = \frac{\pi\, h_t d_t^2}{12},$$

where: ht = height of tree, and
dt = diameter of tree at breast height.

Volumes of spires were subtracted from stem biomass estimates for top-killed trees using the following estimate for basal diameter:

$$d_s = d_t \left(\frac{h_s}{h_t}\right),$$

where: ds = basal diameter of spire, and
hs = height of spire.

Results and Discussion

Douglas-fir was the predominant tree species in blocks A–F, accounting for 98% of all trees sampled ($n = 325$). The percentage (±SE) of live stem occupied by live crown (crown/stem) averaged 59 ± 0.9% over all blocks (Table 9). Crown/stem percentages were highest in blocks D–F and lowest in blocks A and B. Spruce, lodgepole pine, trembling aspen, and paper birch accounted for the remainder and were not considered in subsequent analyses.

The mean height of Douglas-fir throughout blocks A–F was 16.5 ± 0.38 m. However, the distribution of heights was quite broad, reflecting a multi-layered stand (Figure 11). One obvious explanation for such a distribution was the prevalence of topkill in the stand. Almost 50% of the Douglas-fir had topkill spires, with the lowest incidence in blocks E and F (Table 9).

The height of trees without topkill spires averaged 20.2 ± 0.46 m across all blocks, with the tallest trees in blocks C–E (Table 10). In contrast, the height of trees with topkill spires averaged 12.5 ± 0.43 m across treatment blocks,

TABLE 9 *Crown/stem percentages and topkill incidences for Douglas-fir at the IDFxh2 site of the Opax Mountain silviculture trial*

Block	*n*	Mean percent live crown[a]	Topkilled trees (%)
A	50	53 ± 2.7 ab	52
B	50	52 ± 2.2 a	50
C	74	57 ± 1.9 abc	50
D	48	64 ± 2.0 c	52
E	59	62 ± 2.0 bc	44
F	44	65 ± 2.9 c	39
All	325	59 ± 0.9	48

a Means followed by the same letter within a column are not significantly different at $p = 0.05$ (Tukey's Multiple Comparison Test).

significantly less than that for trees without spires (*t*-test, pooled $df = 323$, $p < 0.001$). Differences in tree heights are only partially explained by heights of spires, which averaged 2.9 ± 0.14 m. Fifteen percent of all spires had broken off and fallen to the ground.

The mean diameter of topkilled trees was significantly less than the mean diameter of trees without topkill (*t*-test, pooled $df = 323$, $p < 0.001$) (Table 11). As with tree heights, the diameters of trees without topkill spires were largest for trees in block D and smallest for trees in block B. No significant difference existed in diameters of trees with topkill spires.

The mean stem biomass of topkilled trees was 54% less than the mean biomass of trees with no topkill (t-test, pooled df = 323, p < 0.001) (Table 12). The lowest volumes of stem biomass with no topkill were found in trees in blocks A, B, and F, while the highest were in trees in blocks C and D. No significant difference existed in stem biomass of trees with topkill among the treatment blocks. Volumes within spires accounted for 1% of the difference in stem biomass between trees with topkill compared to those without topkill.

Risk of defoliation by either western spruce budworm or Douglas-fir tussock moth is currently low because little, or no, defoliation occurred in 1996. However, the silvicultural system site is in a high-risk region and will likely

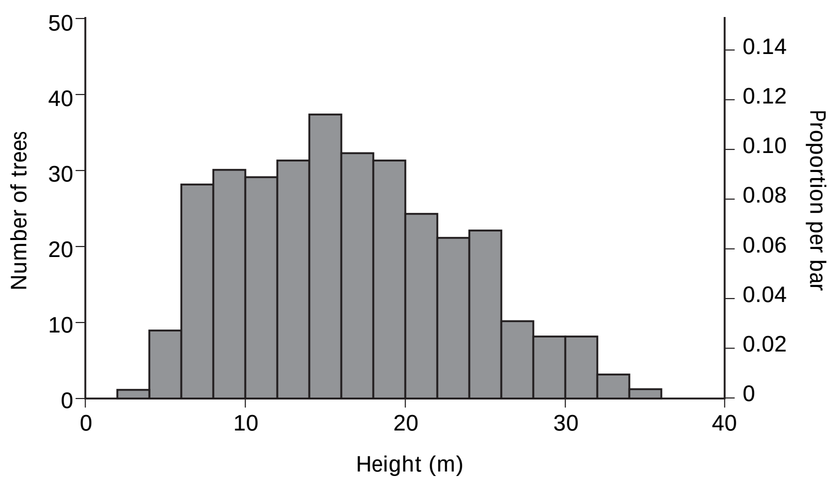

FIGURE 11 *Frequency distribution of heights of live stems of Douglas-fir at the Opax Mountain silviculture trial (n = 325).*

TABLE 10 *Height of trees with and without topkill spires at the IDFxh2 site of the Opax Mountain silviculture trial*

	Without topkill spires		With topkill spires		
Block	***n***	**Mean (± SE) height of live stem (m)a**	***n***	**Mean (± SE) height of live stem (m)[a]**	**Mean (± SE) height of topkill spire (m)[a]**
A	24	18.0 ± 0.99 a	26	10.2 ± 0.70 a	3.2 ± 0.49 a
B	25	18.5 ± 0.74 a	25	12.5 ± 0.91 a	2.4 ± 0.29 a
C	37	21.7 ± 1.06 ab	37	13.6 ± 1.07 a	2.7 ± 0.25 a
D	23	23.6 ± 1.35 b	25	12.1 ± 1.13 a	3.5 ± 0.35 a
E	33	20.8 ± 0.93 ab	26	13.1 ± 0.96 a	2.7 ± 0.30 a
F	27	17.8 ± 1.21 a	17	13.1 ± 1.36 a	3.2 ± 0.46 a
All	169	20.2 ± 0.46	156	12.5 ± 0.43	2.9 ± 0.14

a Means followed by the same letter within a column are not significantly different at $p = 0.05$ (Tukey's Multiple Comparison Test).

TABLE 11 *Diameter at breast height (dbh) of trees with and without topkill spires at the IDFxh2 site of the Opax Mountain silviculture trial*

	Without topkill spires		With topkill spires	
Block	***n***	**Mean (± SE) dbh (cm)[a]**	***n***	**Mean (± SE) dbh (cm)[a]**
A	24	27.2 ± 2.28 ab	26	19.5 ± 1.12 a
B	25	25.4 ± 1.27 a	25	21.3 ± 1.33 a
C	37	34.1 ± 2.43 ab	37	23.9 ± 1.82 a
D	23	37.2 ± 2.69 b	25	23.5 ± 2.30 a
E	33	31.8 ± 2.54 ab	26	23.3 ± 1.84 a
F	27	25.9 ± 2.02 a	17	25.5 ± 3.18 a
All	169	30.5 ± 1.00	156	22.8 ± 0.79

a Means followed by the same letter within a column are not significantly different at $p = 0.05$ (Tukey's Multiple Comparison Test).

TABLE 12 *Stem biomass (sbm) of trees with and without topkill spires at the IDFxh2 site of the Opax Mountain silviculture trial*

	Without topkill spires		With topkill spires	
Block	***n***	**Mean (± SE) sbm (m3)[a]**	***n***	**Mean (± SE) sbm (m3)[a]**
A	24	0.46 ± 0.085 ab	26	0.15 ± 0.025 a
B	25	0.36 ± 0.043 a	25	0.22 ± 0.039 a
C	37	0.94 ± 0.155 bc	37	0.39 ± 0.104 a
D	23	1.10 ± 0.171 c	25	0.37 ± 0.115 a
E	33	0.78 ± 0.159 abc	26	0.33 ± 0.098 a
F	27	0.44 ± 0.086 ab	17	0.46 ± 0.201 a
All	169	0.70 ± 0.058	156	0.32 ± 0.042

a Means followed by the same letter within a column are not significantly different at $p = 0.05$ (Tukey's Multiple Comparison Test).

experience an intense epidemic in the near future. The multi-layered conditions of the stands at the Opax Mountain trial site, combined with patches of high-density, closed-canopy stands will provide an ideal environment for a protracted infestation of western spruce budworm, likely resulting in severe levels of defoliation.

Past infestations have created heterogeneity in stand structure, with almost 50% of trees with topkill. These stagheads provide perching and nesting opportunities for birds, as well as sheltered areas for canopy insects. Over time, the spires fall to the ground, providing additional small, woody debris to the forest floor.

Topkilled trees were dramatically different in height, diameter, and stem biomass volume compared to trees with no topkill. The spires account for only a small portion of these differences. The rest can be ascribed to growth losses attributed to defoliation by the western spruce budworm and the Douglas-fir tussock moth and/or feeding preferences of defoliators on smaller, sub-dominant trees.

SUMMARY AND RECOMMENDATIONS FOR 1997

The lack of buffer zones and replication at the Opax Mountain trial site severely restricts conclusive interpretation of the silvicultural treatments on beetle fauna. Arboreal beetles are highly vagile, dispersing over large areas, and often flying distances greater than 0.5–1.0 km. The distribution of beetles within a stand is often patchy such as spot infestations of Douglas-fir beetle. Replication over a broad range of site characteristics is required for objective evaluations of stand treatments on forest health.

However, the Opax Mountain Silvicultural Systems Project does provide an invaluable opportunity to gain baseline information on the diversity of arboreal beetles, as well as to develop sampling methods geared to a broad range of resource objectives. The combined research endeavours at Opax Mountain have provided considerable insight into measures of interest to other researchers, such as the distribution of woody debris and the associated beetles that initiate decomposition.

Work conducted in 1996 has resulted in the compilation of a large reference collection, which consists of over 40 families of Coleoptera. This collection will be invaluable in the training of technicians for further replicated studies, as well as a reference source for the University College of the Cariboo and other public facilities and organizations.

The funnel-trap method is well suited to studies of forest health and biodiversity. The site offers an excellent opportunity to assess the relative benefits of various combinations of trap position, lures, and preservatives. Evaluation of the following could significantly improve this method:

1. trap height (crown, mid/upper bole, and breast height);
2. preservative (Vapona, ethylene glycol, and propylene glycol);
3. lures (monoterpene, ethanol, and pheromone blends);
4. inter-trap distance (10, 25, 50, and 100 m);
5. visual stimulus (baited and nonbaited); and
6. sample size (one, three, and five traps per site).

In conclusion, the survival of the trial is not currently at risk from forest pests. However, the presence of Douglas-fir beetles in slash and standing trees (albeit at low levels) combined with an abundance of tall, standing stumps may result in a significant increase in the abundance of Douglas-fir beetle and subsequent mortality of standing Douglas-fir. In contrast to up-rooted trees and broken tops, standing stumps with intact root systems will be favourable to attack by beetles because of the minimal degradation of the phloem resource. The level of attacks to stumps, new slash material, and standing trees should be monitored.

REFERENCES

Alfaro, R.I. 1986. Mortality and top-kill in Douglas-fire following defoliation by the western spruce budworm in British Columbia. Journal of the Entomological Society of British Columbia 83:19–26

Alfaro, R.I. and L.E. Maclauchlan. 1992. A method to calculate the losses caused by western spruce budworm in uneven-aged Douglas-fir forests of British Columbia. Forest Ecology and Management 55: 295–313.

Alfaro, R.I., G.A. Van Sickle, A.J. Thomson, and E. Wegwitz. 1982. Tree mortality and radial growth losses caused by the western spruce budworm in a Douglas-fir stand in British Columbia. Canadian Journal of Forest Research 12:780–7.

British Columbia Ministry of Forests and B.C. Environment. 1995a. Bark Beetle Management Guidebook. Forest Practices Code of British Columbia. B.C. Ministry of Forests, Victoria, B.C.

______. 1995b. Root Disease Management Guidebook. Forest Practices Code of British Columbia. B.C. Ministry of Forests, Victoria, B.C.

______. 1995c. Defoliator Management Guidebook. Forest Practices Code of British Columbia. B.C. Ministry of Forests and B.C. Environment, Victoria, B.C.

______. 1995d. Biodiversity Guidebook. Forest Practices Code of British Columbia. B.C. Ministry of Forests and B.C. Environment, Victoria, B.C.

Erickson, R.D. 1995. Douglas-fir tussock moth. Natural Resources Canada, Canadian Forest Service. Forest Pest Leaflet No. 9.

Erickson, R.D. and J. Loranger. 1987. Forest Insect and Disease Conditions. Kamloops Forest Region 1986. Canadian Forestry Service FIDS File Report 87-2.

______. 1988. Forest Insect and Disease Conditions. Kamloops Forest Region 1987. Canadian Forestry Service FIDS File Report 88-2.

Furniss, R.L. and V.M. Carolin. 1980. Western Forest Insects. United States Department of Agriculture Forest Service Miscellaneous Publication No. 1339.

Harris, J.W.E., R.I. Alfaro, A.F. Dawson, and R.G. Brown. 1985. The western spruce budworm in British Columbia 1909-1983. Canadian Forestry Service Information Report BC-X–257.

Humphreys, N. 1995. Douglas-fir beetle in British Columbia. Natural Resources Canada, Canadian Forest Service. Forest Pest Leaflet No. 14.

Koot, H.P. and J. Hodge. 1991. Forest insect and disease conditions Kamloops Forest Region 1990. Forestry Canada FIDS Report 91-2.

______. 1992a. Forest insect and disease conditions Kamloops Forest Region 1991. Forestry Canada FIDS Report 92-2.

______. 1992b. History of population fluctuations and infestations of important forest insects in the Kamloops Forest Region 1912-1991. Forestry Canada FIDS Report 92-11.

______. 1993. Forest insect and disease conditions Kamloops Forest Region 1992. Forestry Canada FIDS Report 93-2.

______. 1994. Forest insect and disease conditions Kamloops Forest Region 1993. Natural Resources Canada, Canadian Forest Service FIDS Report 94-2.

______. 1996. Forest insect and disease conditions Kamloops Forest Region 1995. Natural Resources Canada, Canadian Forest Service FIDS Report 96-2.

Koot, H.P. and J. Loranger. 1989. Forest insect and disease conditions Kamloops Forest Region 1988. Forestry Canada FIDS Report 89-2.

______. 1990. Forest insect and disease conditions Kamloops Forest Region 1989. Forestry Canada FIDS Report 90-2.

Schmidt, W.C. 1985. Historical considerations. *In* Managing Trees and Stands Susceptible to Western Spruce Budworm. M.H. Brooks, J.J. Colbert, R.G. Mitchell, and R.W. Stark (Technical Coordinators). United States Department of Agriculture Forest Service Technical Bulletin No. 1695 p. 1–5

Thomson, A.J., R.F. Shepherd, J.W.E. Harris, and R.H. Silversides. 1984. Relating weather to outbreaks of western spruce budworm, *Choristoneura occidentalis* (Lepidoptera: Tortricidae), in British Columbia. Canadian Entomologist 116: 375–81.

VanderSar, T. 1987. The western spruce budworm feeding on Douglas-fir. Information Forestry 14: 1–10.

Van Sickle, G.A. 1985. Effects of infestations on trees and stands. *In* Managing Trees and Stands Susceptible to Western Spruce Budworm. M.H. Brookes, J.J. Colbert, R.G. Mitchell, and R.W. Stark (Technical Coordinators). United States Department of Agriculture Forest Service Technical Bulletin No. 1695 p. 15–21

Depth and Duration of Snow at the Opax Mountain Silvicultural Systems Site

DAVID HUGGARD, RUSS WALTON, AND WALT KLENNER

INTRODUCTION

Snow is a dominant factor in the ecology of temperate forests. Even in the relatively warm, dry forests of the Interior Douglas-fir (IDF) zone, snow covers the ground for one-half of the year, and influences many aspects of floral and faunal ecology. For large mammals, snow increases the energetic costs of travel and can impede travel completely. Mule deer, for example, are severely impeded by snow greater than 40 cm deep (Trottier et al. 1983) and rarely use sites with deeper snow (see Huggard and Klenner, these proceedings, p. 292). Snow reduces forage access for ungulates and can eliminate many foraging substrates for resident bird species. Pileated woodpeckers, for example, forage primarily on well-decayed logs and stumps in summer, but are restricted to less-preferred sites off the ground in periods of deep snow (see Klenner and Huggard, these proceedings, p. 277). At the same time, deeper snowpacks enable small mammals to safely reach feeding sites, such as the tops and bark of regenerating conifers, that would not otherwise be affected by feeding damage (Harestad et al. 1986). Snow cover also acts as insulation—depths greater than 30 cm can maintain ground surface temperatures near 0° (Perla and Martinella 1976). This insulation protects an active and diverse arthropod community in winter, with many species uniquely adapted to subnivean conditions (Aitchison 1979). By preventing extensive freezing of the soil, snow cover allows the survival of many species of plants that cannot tolerate severely frozen roots. However, snow cover can physically damage many species, eliminating them from sites with deep snowpacks (Kimmins 1987). While a persistent snowpack in spring can retard early plant growth, in dry ecosystems it can also increase the availability of groundwater during the middle of the growing season (Kimmins 1987).

Because of the many ecological ramifications of snow, knowledge of the effects of forest harvesting on snow depth and duration is essential to understand how ecosystems will respond to forest management. Numerous hydrological studies have shown that the amount of snow that accumulates is affected by forest canopy cover, the size of openings in the forest, and the distance from the edge of openings. However, these studies have also shown that snow responds to these aspects of forest management in a site-specific manner (Golding 1982). The study summarized here was designed to measure the effects of different degrees of canopy removal, opening sizes, and distances from cutblock edges on snowpack depth and duration at the Opax Mountain Silvicultural Systems site. These measures will provide basic information for numerous integrated studies of the ecological effects of various harvesting systems in the IDF forest.

Study Variables

The Opax Mountain Silvicultural Systems site (for a site description, see Klenner and Vyse, these proceedings, p. 128) provided an experimental setting for evaluating the effects of three forest management variables on snowpack depth and duration.

1. Canopy cover: a range of canopy cover was provided by replicated blocks of uncut forest, 20 and 50% volume removal in individual-tree selection cuts, and small clearcut patches.
2. Opening size: several square patch cuts of 0.1 ha (1.1 tree heights across), 0.4 ha (2.2 tree heights across), and 1.6 ha (4.4 tree heights across) were available in four replicate blocks.
3. Edge distance: the patch cuts allowed sampling of distances up to 72 m into openings and 48 m into surrounding leave strips of uncut forest.

We measured snow depth only (not density or water equivalency), using a graduated pole inserted through the snowpack to the ground. Sampling stations were established before snowfall at locations where large logs or rocks would not interfere with depth measurements. Observers avoided walking within 1 m of sampling stations.

Sampling Uncut, Partial-Cut, and Clearcut Treatments

In 1995/1996, we sampled 30 stations 30 m apart on a 3 × 10 grid in each uniform treatment unit (uncut control, 20 and 50% partial cuts) at the Opax Mountain and Mud Lake sites. At each station, we established three substations spaced 2 m apart, and within 10 cm of each substation we measured three snow depths ("subsamples") for each sampling period. Depths in clearcuts were indicated by stations at 48 m and 65 m into the 1.6-ha openings (see "Sampling across Edges," below).

In 1996/1997, we reduced our sampling effort to 20 stations spaced 60 m apart on a 4 × 5 grid to cover a wider area. At each station, we placed two substations 4 m apart, and at each substation we measured one subsample (see "Sampling Optimization," Appendix 1).

Sampling across Edges

We ran transects into both the forest and the clearings on the north and south sides of six 0.1-ha, six 0.4-ha, and seven 1.6-ha openings. We established sampling stations at 0, 6, and 12 m from the edge in small patch cuts; at 0, 6, 12, and 24 m from the edge in medium patch cuts; and at 0, 6, 12, 24, and 48 m from the edge in large patch cuts. In 1996/1997, we added a sampling station to the centre of the large patch cuts, located 65 m from each edge. The number of substations at each station on the edge transects, and the number of subsamples was the same in 1995/1996 and 1996/1997 as in the uncut and partial-cut grids, above.

Sampling Schedule

In the winter of 1995/1996, we sampled the Opax Mountain and Mud Lake sites three times (Feb. 19–29, Apr. 1–4, and Apr. 15–17) and the Opax Mountain site by itself on April 24. In the winter of 1996/1997, we sampled both sites six times (Jan. 6–7, Feb. 24–26, Mar. 27–Apr. 1, Apr. 15–16, 24–26, and 29) and sampled the Opax Mountain site by itself on May 15. In both years and at both sites, virtually no snow was present 1 week after the final sampling period.

Data Analysis

Because the variance in snow depths between the Opax Mountain and Mud Lake sites could mask the forest management effects we were trying to measure, we standardized the data at each sampling period by subtracting the mean snow depths for each area. The resulting residual values show how the depth in a particular treatment unit or at a particular distance from an edge differed from the mean values for that area at that period.

RESULTS

Differences between Years and Sites

Average peak snow depths in 1996/1997 were 14 cm greater at Mud Lake and 21 cm greater at Opax Mountain than in the previous winter. Despite this yearly difference in peak snow depth, the effects of the treatments and the edge relationships were similar for both sites and both years. For brevity, we present results from 1996/1997 only.

Effects of Harvest Treatments on Snow Depths

Average snow depths were consistently 15–20 cm lower in the control and the partial-cut treatment units than in the clearcut units, and this trend continued until rapid snowmelt in April (Figure 1). We observed no obvious differences in snow depths among the control and the partial-cut treatment units (Figure 1), despite the range in cutting intensity from 0 to 50% volume removal.

Edge Effects

Clearcuts had deeper snowpacks than adjacent uncut forests (Figure 2). In early January, snow depths across the clearcuts were consistently 10 cm above average for the study area, regardless of the distance from the edge (Figure 2a). As snow accumulations peaked in late February, snow depths decreased from the north- to the south-facing edges of clearcuts (Figure 2b).

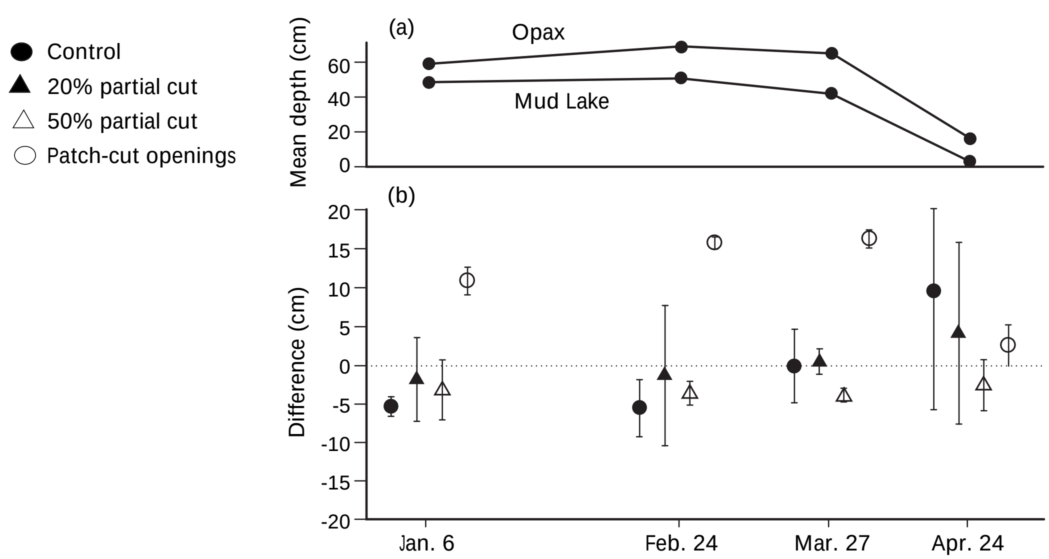

FIGURE 1 *(a) Average snow depth for four representative sampling sessions in 1996/1997, and (b) difference in snow depth from the area mean for control, 20% partial cut, 50% partial cut, and patch-cut openings. Error bars are 2 SE.*

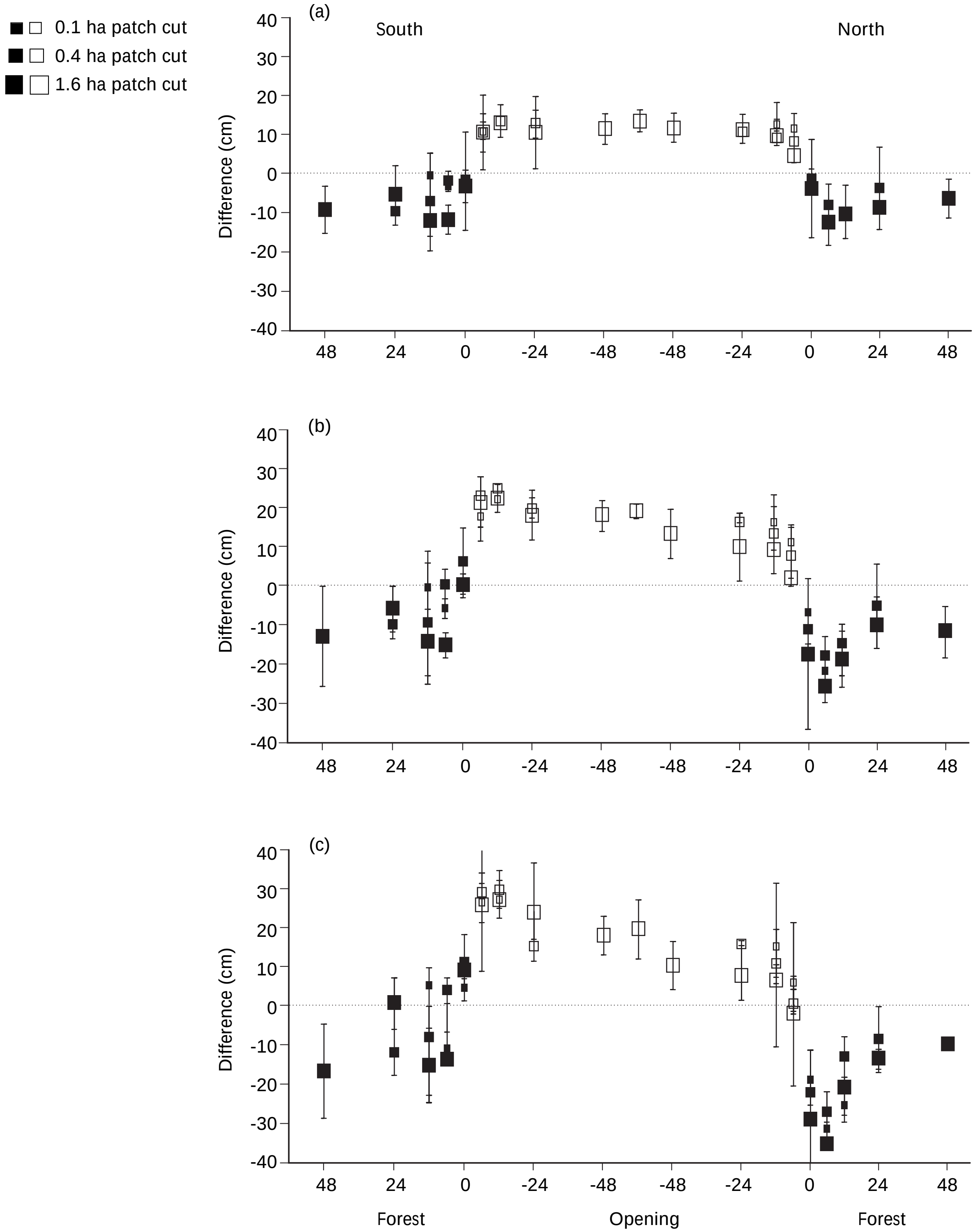

FIGURE 2 *Difference (± 2 SE) in snow depths from the area mean for transects extending across edges of patch cuts. Solid squares represent stations within the forest. Sampling session shown are: (a) January 6–7, 1997; (b) February 24–26, 1997; and (c) March 27–April 1, 1997.*

The importance of edge aspect increased as the season progressed, increasing the difference between snow depths on the northern and the southern edges of openings in April (Figure 2c). By late April, most of the snow remaining in clearcuts was concentrated within 12 m of the southern edges.

The effect of edge aspect also extended into the uncut forest adjacent to openings. Forest within 12 m of the south-facing edge of clearcuts recorded the lowest snow depths in the study area, especially as winter progressed (Figures 2b and 2c). This effect disappeared beyond 12 m into the forest.

Effects of Patch Cut Size on Snow Depths

No obvious differences were evident in snow depths or edge patterns between the 0.1-, 0.4-, and 1.6-ha openings (Figure 2). Snow depths tended to be slightly lower in the forest near the edges of the largest openings compared to the smaller openings, but this difference was minor compared to the main effect of distance from edge.

Timing of Complete Melt

Exposure of bare ground at the sampling stations began in late March at both sites, but proceeded more rapidly at the lower-elevation Mud Lake site (Figure 3). At both sites, the south-facing edges of openings, especially 0–12 m into the forest, were the first to melt completely, while the north-facing edges, in both the forest and the opening, melted the latest. All sites on south-facing edges had melted completely by mid-April at Mud Lake and later April at Opax, while almost all the sites on north-facing edges still had snow cover. The last patches of snow disappeared from the north-facing edges at Mud Lake in the beginning of May and in the middle of May at Opax. We observed the same temporal pattern in 1995/1996, except that complete melt occurred about 10 days earlier.

The uncut patches and the 20 and 50% partial cuts had a similar timing for complete melt at Mud Lake (Figure 3a), whereas the snowpack in the 50% partial cut at Opax disappeared faster than in the uncut or light partial cut (Figure 3b). However, the 50% partial cut replicate at the Opax area is located on rolling terrain with steep southern exposures. Aspect differences, rather than the harvesting itself, may explain the faster melt in this treatment unit.

DISCUSSION

From a hydrological perspective, it is important to understand the causes of observed effects of harvesting on snow depths. Lower snow depths in forests, for example, might be caused by increased interception and sublimation of snow in the canopy, or by redeposition of snow from the canopy to openings. The former cause would reduce the amount of water in the watershed, while the latter would have no overall hydrological effect (Golding and Swanson 1986). Similarily, a decrease in snow depth in the spring could be caused by either melting or the compaction of the snowpack, with different implications for the timing and magnitude of peak runoff. However, this study was not designed to explore the causes of differences in snow depth and duration, but simply to measure them because snow affects so many ecological patterns.

The most obvious effect of harvesting was the large increase in snow depth in openings, both during peak accumulation and during spring melt or com-

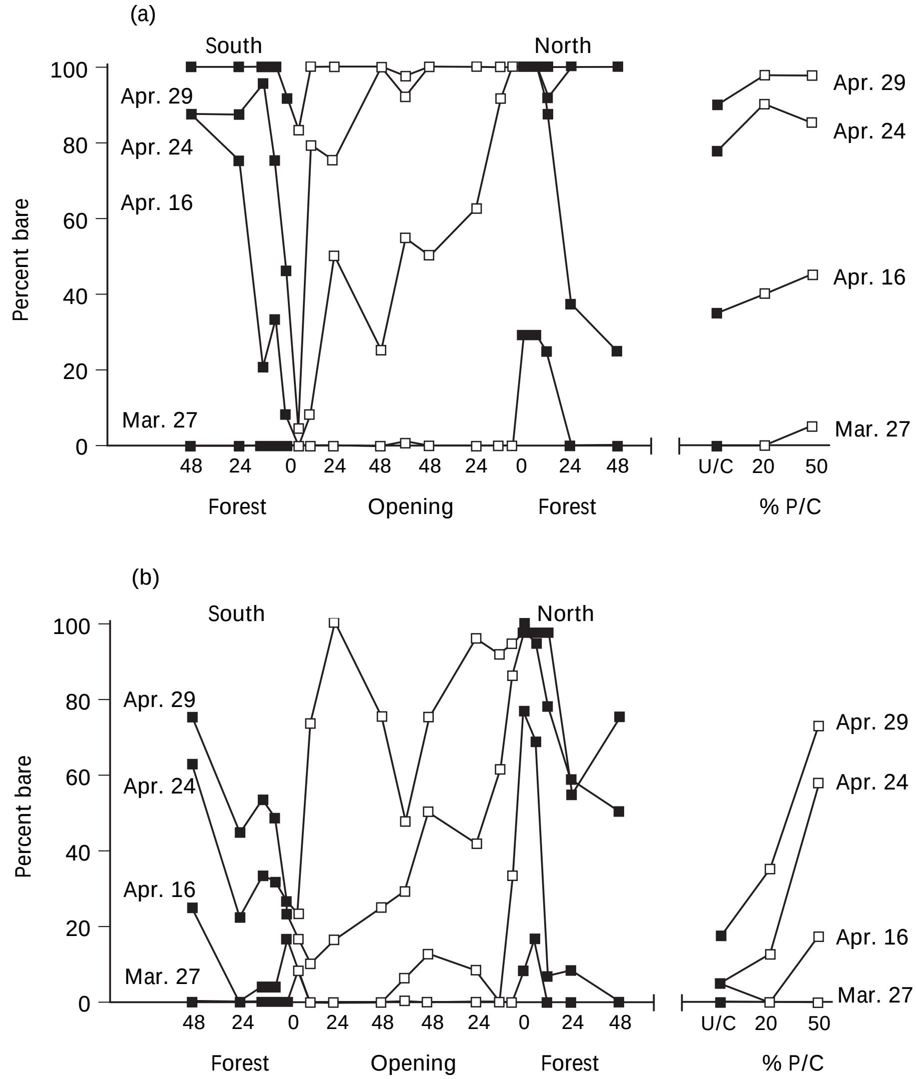

FIGURE 3 *Percentage of sampling stations with bare ground in four sampling sessions in 1997 for (a) Mud Lake and (b) Opax Mountain sites. Results shown for edge transects (left side of each figure) and uniform treatments (right side).*

paction. Our observations suggest that deeper snow in the openings prevents the ground from freezing, while much of the ground in the uncut forest is frozen throughout the winter. This difference may have implications for plant and winter arthropod communities. Snow accumulations in openings far exceeded depths that impede deer movement, and a tracking study at the site showed that deer did not cross the openings in winter (see Huggard and Klenner, these proceedings, p. 292). Opening size is shown to affect snow depths in many studies (Troendle and Leaf 1981; Golding 1982), but the range of opening sizes at the Opax Mountain site produced no pronounced differences. However, the partial cuts, which effectively create numerous small openings (< 0.5 tree heights across), did not have any noticeable effects on snow depths when compared to the depths in uncut forest. This observation suggests that very small patch cuts may be an alternative harvesting system in areas where managers want to avoid increasing the depth of snowpacks.

Edge effects in openings and the adjacent forest were negligible during peak accumulation times, but became pronounced during spring compaction and melting. The pattern of decreased snow on the north side of an opening, with very low snow accumulations in the immediately adjacent south-facing forest, has been reported in several studies (Troendle and Leaf 1981; Golding and Swanson 1986). This pattern is attributed to unshaded direct solar radiation and subsequent re-radiation from fully exposed trees. The deeper snow in the southern part of the opening and in the adjacent north-facing forest must indicate drifting of snow as well as shading, because the snowpack along this edge was deeper and more persistent than similarly shaded uncut forest away from edges.

Ecologically, the most important implication of our observed edge effects is probably that the bare ground along the south-facing edges is exposed much earlier, and that the complete melting of snow along the north-facing edge is delayed longer, resulting in a snow-free period that extends up to 1 month longer on the south-facing edge. Plant composition, and particularly phenology, should respond to the earlier start of the growing season on south-facing edges (see Miege et al., these proceedings, p. 211). Shrews apparently responded in spring with increased abundance near the south-facing edge of openings (Huggard and Klenner, these proceedings, p. 235). However, exposure of bare ground on the south-facing edges as early as March also increases the risk of ground freezing during cold periods in spring, and probably decreases the availability of groundwater later in the summer, both of which could reduce plant productivity.

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REFERENCES

Aitchison, C.W. 1979. Winter-active subnivean invertebrates in southern Canada. II. Coleoptera. Pedobiologia 19:121–8.

Golding, D.L. 1982. Snow accumulation patterns in openings and adjacent forest. Canadian Hydrology Symposium 82: 91–112.

Golding, D.L. and R.H. Swanson. 1986. Snow distribution patterns in clearings and adjacent forest. Water Resources Research 22:1931–40.

Harestad, A.S., F.L. Bunnell, T.P. Sullivan, and L. Andrusiak. 1986. Key to injury of conifer trees by wildlife in British Columbia. B.C. Ministry of Forests, Victoria, B.C. WHR-23.

Kimmins, J.P. 1987. Forest ecology. Macmillan Publishing Company, New York, N.Y.

Perla, R.I. and M. Martinelli, Jr. 1976. Avalanche handbook. U.S. Department of Agriculture Forest Service, Washington, D.C. Handbook No. 489.

Sit, V. 1995. Analyzing ANOVA designs. B.C. Ministry of Forests, Victoria, B.C. Biometrics Information Handbook No. 5.

Troendle, C.A. and C.F. Leaf. 1981. Effects of timber harvest in the snow zone on volume and timing of water yield. *In* Interior West watershed management. D.M. Baumgartner (editor). Washington State University, Pullman, Wash.

Trottier, G.C., S.R. Rollans, and R.C. Hutchison. 1983. Range, habitat and foraging relationships of ungulates in Riding Mountain National Park. Canadian Wildlife Service, Ottawa, Ont. Large Mammal Systems Studies Report No. 14.

We assessed the efficiency of our first season's sampling design using Monte Carlo simulations based on the observed sources of sampling variation in the first year's data. Using data for the control and partial-cut grids from the late February and early April 1996 sampling periods, we used analysis of variance and determined the percent of variance (following Sit 1995) contributed by each source of variation in the doubly nested, unreplicated randomized block design. Table A1.1 shows the percentage of observed variance attributable to each of the sources. Differences between the Opax Mountain and Mud Lake sites alone accounted for between 9 and 22% of the total variation, which suggests that analyses should be standardized by site. Over 40% of the overall variance was attributed to differences among stations within a replicate block, while differences between substations, or between subsamples at a substation were of less importance.

Based on these variance components, we conducted a Monte Carlo simulation to examine the expected error in mean snow depth for a treatment unit under various sampling designs with differing numbers of stations per block, substations per station, and subsamples per substation. One hundred simulation runs were conducted for each sampling design, and a mean expected error calculated. We calculated the costs of each sampling design based on the estimated time to move between stations, to sample additional substations at each station, and to record additional subsamples at each substation.

Simulation results showed that reducing the number of substations from three to two per station, and reducing the subsamples from three to two per substation would have virtually no effect on expected error, but would reduce the time per station by over 50%. Similarly, reducing the number of stations per block from 30 to 20 would increase the expected error only slightly, but would allow us to spread out the stations without increasing the total time spent moving between stations. The reduced sampling design was still intensive enough to "average out" almost all of the sampling error effects, which left only the variance between the two sites as the main source of sampling error. Because the study design is constrained to only two sites, more intensive sampling at each block would only slightly reduce the sampling error. However, by reducing the sampling time of each block by 60%, the optimized sampling design allowed us to sample more often throughout the winter without increasing field costs.

TABLE A1.1 *Relative importance of sources in explaining variation in snow depths among control and partial-cut treatments in 1995/1996*

	% variation explained	
Source	**Feb. 19–29**	**Apr. 1–4**
Mud Lake vs Opax (sites)	9.3	21.6
Treatments	4.4	3.7
Area × treatment (= replicate blocks)	3.9	3.2
Stations	50.3	42.4
Substations	17.0	17.5
Subsamples	15.1	11.6

Net Nitrogen Mineralization and Nitrification in the Forest Floor and Mineral Soil at the Opax Mountain Silvicultural Systems Trial

CINDY PRESCOTT AND GRAEME HOPE

INTRODUCTION

Through the processes of decomposition and mineralization, the nitrogen bound in organic matter is released as inorganic nitrogen in forms (e.g., ammonium [NH_4-N] and nitrate [NO_3-N]) that can be taken up by plants. Some portion of the mineralized nitrogen is also assimilated and temporarily immobilized by the microbial biomass and then remineralized upon death. The rate of net nitrogen mineralization (i.e., the difference between rates of mineralization and immobilization) is an index of the nitrogen available to plants. Nitrification is the process in which NH_4-N is converted to NO_3-N by micro-organisms. Nitrate is also rapidly reassimilated by micro-organisms, so the net rate of nitrification is measured. Rates of nitrogen mineralization are affected by factors that influence microbial activity, particularly moisture, temperature, and the nature of the organic material. Because clearcutting can affect each of these factors, it has the potential to substantially alter rates of nitrogen mineralization and nitrification in forest floors. This will result in higher nitrogen availability to plants, but also greater losses of nitrogen from the site through nitrate leaching.

The availability of inorganic nitrogen in the forest floor and soil usually increases after clearcutting forests (Vitousek 1981). However, little is known about the effect of opening size on nitrogen mineralization (i.e., whether to expect a gradient or a threshold effect). In lodgepole pine forests, higher rates of net nitrogen mineralization have been reported in a 0.25 ha patch cut (Prescott et al. 1992), and after removal of at least 15 trees (Parsons et al. 1994).

The influence of opening size on rates of net nitrogen mineralization and nitrification are being assessed in the Opax Silvicultural Systems Trial. In the first phase of the study, which is reported here, rates were measured at different distances from the edge within a single large opening (1.7 ha) to establish the relationship with distance from edge. Rates were also measured in the forest on the north and south side of the opening for comparison with rates in the opening. Simultaneous incubations of soil in the laboratory under controlled temperature conditions indicated the extent to which differences along the transect resulted from differences in temperature at each location. The degree of correlation between the nitrogen mineralization rate and moisture content was measured to assess the influence of differences in moisture conditions at different locations across the transect. In the second phase, which began in 1997, rates of these processes were measured in the centre of different-sized openings. Comparison with results of the transect will tell us if the effects of different opening sizes can be predicted from relationship to edges in large openings. The questions addressed here are:

1. Is there a difference between rates of nitrogen mineralization in the opening compared to the forest?
2. Does the effect of opening increase with distance from the forest edge?
3. Is the effect different on north and south sides of the opening?
4. Are rates of net nitrification greater in the opening?
5. Are differences related to differences in the temperature or the moisture content of the forest floor and soil at each location?

METHODS

All sampling was in one 1.7 ha opening in the Mud Lake block. A transect with nine stations was established from the south side of the opening to the north side. These stations were designated by uppercase letters from A to I and were located in the following positions:

- A: in the forest at south edge of the opening;
- B: 2 m from south edge into opening;
- C: 10 m from south edge into opening;
- D: 25 m from south edge into opening;
- E: 65 m from south edge at centre of opening;
- F: 25 m from north edge of opening;
- G: 10 m from north edge of opening;
- H: 2 m from north edge of opening; and
- I: in the forest at north edge of opening.

Separate samples of the forest floor and the upper 10 cm of mineral soil were collected at eight random locations near each station. Each sample was divided into four subsamples. One subsample was placed in a thin plastic bag and replaced in the forest floor or in the mineral soil near its place of origin, and incubated in the field. The second subsample was incubated in the laboratory at about 20°C for about 6 weeks. The third subsample was extracted immediately in potassium chloride and concentrations of NH_4-N and NO_3-N were measured with a Alpkem RFA autoanalyzer. The fourth subsample was weighed and then oven-dried to determine its moisture content. Following the field and laboratory incubations, samples were similarly extracted and concentrations of inorganic nitrogen were measured. In total, five sets of samples were incubated; dates for field (and lab, if different) incubations were:

- November 1, 1995 to May 1, 1996 (January 3, 1996)
- May 16 to June 19, 1996
- June 19 to August 7, 1996
- August 7 to October 1, 1996
- October 1 to October 31, 1996

Following incubation, the concentrations of NH_4-N and NO_3-N in each sample was subtracted from the initial (pre-incubation) concentrations in the sample. The difference is an estimate of the amount of NH_4-N and NO_3-N mineralized during the incubation. Amounts of the two nitrogen ions were summed to estimate the amount of inorganic nitrogen mineralized or nitrified during the incubation.

For each incubation, differences between stations in the net amounts of N, NH_4-N, and NO_3-N produced during the incubation were compared using one-way analysis of variance followed by Bonferroni's multiple range test. The Windows software program, SPSS, was used for all analyses. Because this phase of the study was primarily to identify trends related to distance from edge, results of the statistical analyses are not presented in the figures. Correlations between moisture content and mineralized nitrogen were assessed with Pearson correlation coefficients. These analyses were performed on data from the June–August and August–October incubations, during which the effects of moisture should have been most pronounced.

RESULTS

Results of the field incubations are shown in Figure 1. Results of the laboratory incubation are not shown, but are described below. Most of the inorganic nitrogen extracted was in the form of nitrate, particularly in the opening. The proportion of mineralized nitrogen that was nitrified was greater in the centre of the opening (stations D–F) than in the forest or at the edge. Rates of net nitrogen mineralization and nitrification during the five field incubations follow.

Incubation 1: November–May (Figure 1a)

All samples from stations A and B were missing at the end of the field incubation. In the forest floor, nitrogen mineralization was highest in the opening at D, nitrification peaked in the centre at E. The response in the lab was roughly similar, but more pronounced peaks occurred in the material from stations A and H than in the field. In the mineral soil, rates were low in the field, except for a very pronounced peak at station F, attributed to very high nitrate production. This peak was also apparent in the lab incubation, but rates were higher in all samples from the opening compared to the forest.

Incubation 2: May–June

Only three stations (forest, edge, and centre) were sampled because of problems with the availability of proper bags for incubation. In the forest floor, nitrate was higher in the centre of the cut than in the forest, but ammonium was lower in the centre. In the mineral soil, more nitrogen and nitrate occurred in the centre of the opening. Patterns in the lab incubations were similar to those in the field.

Incubation 3: June–August (Figure 1b)

In the forest floor, the highest rates were in the centre of the opening (D–F) because of higher nitrate concentrations. The pattern in the lab was different because of a peak that occurred at the edge (H). In the mineral soil, a peak occurred at the edge (B) in the field and at Station C in the lab. The overall trends were similar in the lab and the field.

Incubation 4: August–October (Figure 1c)

No lab incubations were included with this set. In contrast to other incubation periods, the amounts of ammonium were similar or greater than nitrate during this incubation. In the forest floor, more nitrate occurred in the

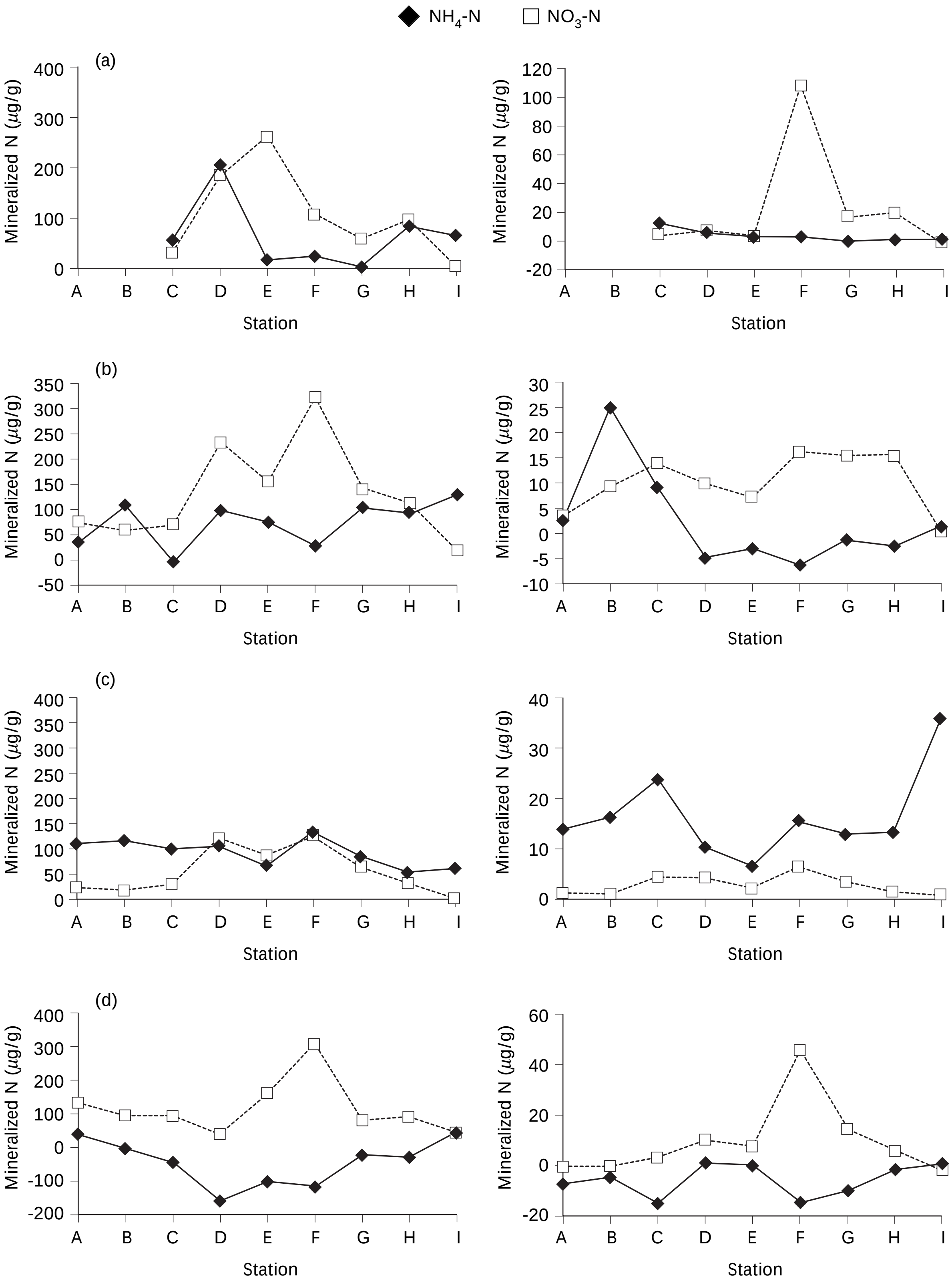

FIGURE 1 *Net nitrogen mineralized and nitrified in forest floor and mineral soil samples during incubations from November 1995 to November 1996 along a north (A) – south (I) transect in a 1.7 ha clearcut. Incubation periods are: (a) November 1995 to May 1996; (b) June 1996 to August 1996; (c) August 1996 to October 1996; (d) October 1996. Each value is the mean of eight samples.*

opening (D–F), which resulted in greater nitrogen in the opening. Ammonium concentrations were similar at all stations. In the mineral soil, nitrate concentrations were low, and ammonium and nitrogen concentrations peaked at stations C and I.

Incubation 5: October 1–31 (Figure 1d)

During this incubation, nitrate production was high, but ammonium mineralization was often negative, indicating that ammonium was being converted to nitrate faster than it was being replenished by mineralization. In all incubations, a pronounced peak in nitrate concentration occurred at station F. Trends in the lab incubations closely resembled those in the field.

DISCUSSION

Rates of nitrogen mineralization in the forest floor tended to be higher in the centre of the opening than at the edges or in the forest. This is consistent with findings from the Sicamous Creek trial (Prescott and Hope 1997), in which elevated rates of nitrogen mineralization occurred in openings of 1.0 ha or larger. In the mineral soil, nitrogen mineralization did not tend to occur more quickly in the opening, apart from very large peaks on two occasions. Concentrations of nitrogen in the mineral soil were generally ten times lower than those in forest floor. However, given the much greater mass of mineral soil than forest floor, changes in nitrogen concentrations in soil are very important. Results of the overwinter (November–May) incubation demonstrate that significant mineralization of nitrogen occurs during the winter at this site.

The higher nitrogen availability after clearcutting forests is usually attributed to greater microbial activity, which results from the warmer, damper conditions in clearcuts (Edmonds and McColl 1989; Frazer et al. 1990; Smethurst and Nambiar 1990). In this study, trends similar to those observed in field incubation were often observed in the lab incubation, when all samples were at the same temperature. Therefore, the differences in mineralization rates among stations do not appear to be the result of temperature differences. Moisture contents of the samples fluctuated over the year, but did not show consistent differences between sampling locations. The correlations were weak between moisture content and nitrogen mineralization in forest floor and mineral soil samples during both field and lab incubations. The strongest correlations were in mineral soil during the June–August lab incubation ($r^2 = 0.25$). We therefore cannot conclude that differences in nitrogen mineralization between the sampling stations were the result of differences in either temperature or moisture conditions. Higher nitrogen availability in openings might be attributed to reduced litter input, reduced plant uptake of nitrogen, and decay of dead roots and slash in clearcuts (Vitousek 1981). Recently, Stark and Hart (1997) suggested that increases in nitrate availability in clearcuts might also be attributed to reduced microbial re-assimilation of nitrogen as the supply of available carbon declines after clearcutting. Further experimentation is needed to establish the mechanisms behind the increased availability of ammonium and nitrate in the opening at Opax Mountain.

While the predominance of nitrate in the opening is consistent with reports from other clearcuts, it contrasts with the low concentrations of nitrate reported in other clearcuts in British Columbia, including other silvicultural systems trials (Prescott et. al. 1993; Prescott 1997; Prescott and Hope 1997). Nitrate was more common in the uncut forest at Opax Mountain than at other sites. These findings are consistent with other recent reports of high rates of nitrogen mineralization in forest floors of Douglas-fir stands. Prescott and Preston (1995) reported higher rates of nitrogen mineralization in forest floors of coastal Douglas-fir stands than in western hemlock or western redcedar stands. Prescott (1996) found that rates of nitrogen mineralization and nitrification were higher in forest floors of coastal Douglas-fir stands than in red alder stands. Thomas (1997) measured greater nitrogen mineralization in forest floors under interior Douglas-fir than under paper birch or lodgepole pine. Together, these findings suggest that nitrogen in Douglas-fir forest floors is rapidly mineralized and nitrified. It might also indicate a lower potential for chemical immobilization or "tanning" of nitrogen in Douglas-fir forest floors, such that nitrogen is mineralized rather than bound in humus. Whatever the mechanism, this suggests that the potential for nitrogen losses through nitrate leaching after clearcutting may be greater in Douglas-fir forests than in forests of other species. Small openings may have smaller nitrogen losses; this will be tested by measuring rates of mineralization and nitrification in the different-sized openings at Opax Mountain. However, the potential for subsurface water flow is low in dry IDF forests, so it is unlikely that losses of nitrate to ground water would be great.

REFERENCES

Edmonds, R.L. and J.G. McColl. 1989. Effects of forest management on soil nitrogen in *Pinus radiata* stands in the Australian Capital Territory. Forest Ecology and Management 29:199–212.

Frazer, D.W., J.G. McColl, and R.F. Powers. 1990. Soil nitrogen mineralization in a clearcutting chronosequence in a northern California conifer forest. Soil Science Society of American Journal 54:1145–52.

Parsons, W.F.J., D.H. Knight, and S.L. Miller. 1994. Root gap dynamics in lodgepole pine forest: nitrogen transformations in gaps of different size. Ecological Applications 4:354–62.

Prescott, C.E. 1996. Influence of forest floor type on rates of litter decomposition in microcosms. Soil Biology and Biochemistry 28:1319–25.

______. 1997. Effects of clearcutting and alternative silvicultural systems on rates of decomposition and nitrogen mineralization in a coastal montane coniferous forest. Forest Ecology and Management 95: 253–60.

Prescott, C.E., J.P. Corbin, and D. Parkinson. 1992. Immobilization and availability of N and P in the forest floors of fertilized Rocky Mountain coniferous forests. Plant and Soil 143:1–10.

Prescott, C.E. and G. Hope. 1997. Sicamous Creek Silvicultural Systems Trial: Litter input, decomposition, and nitrogen mineralization. *In* Sicamous Creek silvicultural systems trial: workshop proceedings. C. Hollstedt and A. Vyse (editors). B.C. Ministry of Forests, Victoria B.C. Working Paper No. 24. pp. 92–100.

Prescott, C.E., M.A. McDonald, and G.F. Weetman. 1993. Differences in availability of N and P in the forest floors of adjacent stands of western redcedar–western hemlock, and western hemlock–amabilis fir on northern Vancouver Island. Canadian Journal of Forest Research 23:605–10.

Prescott, C.E. and C.M. Preston. 1995. Nitrogen mineralization and decomposition in forest floors in adjacent plantations of western redcedar, western hemlock, and Douglas-fir. Canadian Journal of Forest Research 24:2424–31.

Smethurst, P.J. and E.K.S. Nambiar. 1990. Distribution of carbon and nutrients and fluxes of mineral nitrogen after clearfelling in a *Pinus radiata* plantation. Canadian Journal of Forest Research 20:1490–7.

Stark, J. and S.C. Hart. 1997. High rates of nitrification and nitrate turnover in undisturbed coniferous forests. Nature 385:61–4.

Thomas, K.D. 1997. Nitrogen mineralization in forest floors under different tree species and the influence of litter quality. MSc thesis, Faculty of Forestry, University of British Columbia, Vancouver, B.C.

Vitousek, P.M. 1981. Clear-cutting and the nitrogen cycle. *In* Terrestrial nitrogen cycles: processes, ecosystem strategies, and management impacts. F.E. Clark and T. Rosswall (editors). Ecological Bulletin (Stockholm, Sweden) 33:631–42.

The Potential for Woody Angiosperms and Advanced Regenerated Douglas-fir to Provide Refuge for Ectomycorrhizal Inoculum at Opax Mountain: Preliminary Results

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INTRODUCTION

Ectomycorrhizae are symbiotic mutualistic associations between specific fungi and the fine roots of temperate woody angiosperms and conifers. Their importance to plants includes their ability to convert unavailable forms of nitrogen to available forms (Abuzinadah and Read 1986) and to increase water uptake in plants (Parke et al. 1983). The number of fungal types and ectomycorrhizal fungal inoculum is often reduced in clearcuts as compared with undisturbed forests (Parke et al. 1984; Parsons et al. 1994) and this may, in turn, reduce the functional diversity of the ectomycorrhizal community in the disturbed site. Ectomycorrhizal fungi tend to remain active on a site for only 1–2 years following the removal of their conifer hosts (Harvey et al. 1980; Perry et al. 1987). In addition, the number of spores entering a clearcut after tree harvesting is usually not sufficient to maintain the ectomycorrhizal fungal inoculum (Perry et al. 1987); however, advanced regenerated conifers and small woody shrubs are often left on cutblocks in the Interior Douglas-fir (IDF) zone following tree harvesting. These plants, since they harbour ectomycorrhizal fungi commonly found on outplanted seedlings, have the ability to provide refuge for ectomycorrhizal fungal inoculum during the period between harvesting and outplanting (Molina and Trappe 1982; Amaranthus and Perry 1989; Hunt 1992). The objectives of this study were to determine:

1. the plant species that have a high potential to provide refuge for ectomycorrhizal fungal inoculum of outplanted and advanced regenerated Douglas-fir on 1.7-ha cutblocks at Opax Mountain; and
2. the difference between the ectomycorrhizal community in undisturbed forests and 1.7-ha cutblocks in terms of ectomycorrhizal fungal diversity.

METHODS

Site Description and Overall Design

The Opax Mountain site is located within the Lac du Bois planning area, approximately 20 km northwest of Kamloops, B.C. and is classified as IDFxh2 for the lower elevations and IDFdk1 for the upper elevations (Klenner and Vyse, these proceedings, p. 128). The silvicultural treatments used in our study included only the 1.7-ha cutblocks and the adjacent unharvested control areas.

Ectomycorrhizal Survey In July 1995, a survey was conducted of ectomycorrhizae appearing on different species of shrubs and conifers from the unharvested areas. Roots of 14 vascular plant species and 2 conifer species (*Juniperus communis* and Douglas-fir) were collected from the upper and lower Opax Mountain Mud Lake sites. Ten plants from both sites were sampled for each plant species, except when a limited number of plants were available for sampling. Root samples were analyzed in the lab and ectomycorrhizal fungi associated with 200 short roots (when present) from each plant were described in a manner similar to Agerer (1988) and Ingleby et al. (1990), and in accordance with the protocol adopted in Goodman et al. (1996). The percentage of live and dead root tips colonized with ectomycorrhizal fungi from all 10 plants was determined.

Ectomycorrhizal Diversity In September 1996, colonization and diversity of the ectomycorrhizal fungal community of woody shrubs and Douglas-fir were determined from both harvested and unharvested areas. Roots of advanced regenerated Douglas-fir and of five woody angiosperms (*Alnus viridis*, *Arctostaphylos uva-ursi*, *Betula papyrifera*, *Populus tremuloides*, and *Salix commutata*) were collected from 1.7-ha cutblocks and adjacent unharvested areas between September 9–15, 1996 to determine the percent colonization of ectomycorrhizae and the diversity of ectomycorrhizal fungi. Four individuals of each plant species were selected from each of three 1.7-ha cutblocks (C15, E6, E1) from the "C" and "E" areas (the lower Mud Lake site) and from each of three 1.7-ha cutblocks (J11, M1, M2) from the "J" and "M" areas (upper Opax Mountain site). Roots of each plant species were also sampled from undisturbed areas that were adjacent to each of the 1.7-ha openings. *Betula papyrifera* was not present at the upper site and *Alnus viridis* was not found at the lower site; therefore, root samples from *B. papyrifera* were sampled only at the Mud Lake site and those from *A. viridis* were sampled only at the Opax Mountain site. Overall, 240 plants were sampled in September 1996. When possible, 500 root tips per seedling were counted and categorized as either live or dead, or live ectomycorrhizal roots. Thus, approximately 2000 root tips were categorized for each treatment unit. Ectomycorrhizal fungi were characterized as described for the 1995 experiments. The percentage of total (live and dead) roots tips that had live ectomycorrhizae was calculated and expressed as percentage colonization. Both Shannon's and Simpson's diversity values per sampling location were calculated. Two-way and one-way analyses of variances were used to test for significant differences between plant species, harvesting treatments, and elevation treatments. Differences between plant species or treatments were determined using a Tukey pairwise comparison.

Greenhouse Bioassay A laboratory bioassay was established in October 1996. Soil from one 1.7-ha cutblocks (Mud Lake) was collected, brought back to the lab, sieved to remove big rocks and roots, and then sterilized. Five 2–3 cm root segments of a single woody angiosperm, grass species, or a Douglas-fir were mixed in enough soil to fill one leach tube. The soil containing the roots was put into a leach tube and a Douglas-fir seed was sown into the soil mixture. Seedlings were harvested in January 1997. Eight treatments were conducted:

- sterile soil alone;
- sterile soil plus Douglas-fir roots;
- sterile soil plus *Arctostaphylos uva-ursi* roots;
- sterile soil plus *Betula papyrifera* roots;
- sterile soil plus *Populus tremuloides* roots;
- sterile soil plus *Salix commutata* roots;
- sterile soil plus *Alnus viridis* roots; and
- sterile soil plus *Calamagrostis rubescens* roots.

Twenty leach tubes per treatment were sown with Douglas-fir seeds for a total of 160 leach tubes or 160 bioassays. Ectomycorrhizal roots were processed and described as discussed for the field studies.

RESULTS AND DISCUSSION

Ectomycorrhizal Survey

In July 1995, 28 ectomycorrhizal types were distinguished from 10 of the 16 woody plant species observed (Tables 1 and 2). Six of the 15 woody angiosperms, on average, had greater than 25% of their fine roots colonized with ectomycorrhizal fungi (Table 1). The percentage value of roots colonized in descending order was: *Betula papyrifera* (Mud Lake only), *Pseudotsuga menziesii*, *Populus tremuloides*, *Alnus viridis* (Opax Mountain), *Arctostaphylos uva-ursi*, *Salix commutata*, and *Amelanchier alnifolia*.

Ectomycorrhizal Diversity

Although not analyzed statistically, percent colonization of ectomycorrhizal fungi from both the undisturbed and disturbed areas was lowest on *Alnus viridis* roots as compared with all other plant species sampled (Table 3). In most of the plant species tested, the percent colonization of ectomycorrhizal fungi was lower on plant roots from the disturbed than from the undisturbed sites. Douglas-fir, *Arctostaphylos uva-ursi*, *Betula papyrifera*, and *Salix commutata* all had greater than 20% colonization of ectomycorrhizal fungi on the roots, irrespective of elevation or harvest treatment.

On the Mud Lake site, ectomycorrhizal diversity on Douglas-fir was significantly lower in the cutblocks than in the undisturbed areas (Table 3). These data demonstrate that a harvest of 1.7 ha can significantly reduce the diversity of the ectomycorrhizal fungal community on Douglas-fir roots. Other studies have found decreases in ectomycorrhizal fungal inoculum and ectomycorrhizal fungal richness following clearcutting (Parke et al. 1984; Parsons et al. 1994).

Greenhouse Bioassay

Arctostaphylos uva-ursi, *Betula papyrifera*, *Salix commutata*, and *Populus tremuloides* had a higher percent colonization and richness of ectomycorrhizal fungi than did the *Alnus viridis*, grass (*Calamagrostis rubescens*, a non-ectomycorrhizal plant), and control treatments (Figure 1). The only ectomycorrhizal fungus that was associated with the control treatment was *Thelephora*-like. It is most probable that this was a contaminant that survived autoclaving rather than a resident of the soil. The greenhouse study demonstrates that *A. uva-ursi*, *B. papyrifera*, *P. tremuloides*, and *S. commutata* roots have a high potential to provide ectomycorrhizal fungal inoculum to newly planted Douglas-fir seedlings.

TABLE 1 *Ectomycorrhizal types colonizing different plant species on the Opax Mountain site in July of 1995*

		Okanagan University College Opax (OUC Opax) ectmycorrhizal description numbers																											
Plant species	**% colonization**	**20**	**30**	**40**	**41**	**51**	**60**	**84**	**95**	**115**	**120**	**125**	**130**	**148**	**150**	**170**	**222**	**223**	**230**	**234**	**240**	**250**	**252**	**440**	**441**	**510**	**520**	**530**	**540**
Acer glabrum	0.00 ± 0.00																												
Alnus viridis	48.12 ± 7.60		•	•	•	•									•		•		•			•			•				
Amelanchier alnifolia	17.11 ± 9.53		•													•	•		•							•			
Arctostaphylos uva-ursi	34.99 ± 6.33	•	•	•	•	•	•	•	•	•			•	•	•	•			•			•	•				•		•
Betula papyrifera	56.01 ± 10.04		•	•	•		•				•	•	•	•	•	•	•		•		•	•							
Chimaphila umbellata	0.00 ± 0.00																												
Juniperus communis	0.00 ± 0.00																												
Mahonia aquifolium	0.00 ± 0.00																												
Paxistima myrsinites	1.03 ± 1.03																												
Populus tremuloides	44.52 ± 6.80	•	•	•	•	•		•		•			•	•	•	•	•		•	•	•	•	•						
Pseudotsuga menziesii	46.78 ± 9.24		•	•	•	•				•			•		•	•	•		•	•		•		•	•		•	•	
Salix commutata	25.38 ± 5.99	•	•	•	•			•			•		•	•			•		•		•	•				•	•		•
Shepherdia canadensis	15.16 ± 7.79	•	•		•			•					•					•	•				•						
Spiraea betulifolia	0.99 ± 0.99		•																•										
Vaccinium caespitosum	0.97 ± 0.97		•																										
Vaccinium membranaceum	0.00 ± 0.00																												

TABLE 2 *Okanagan University College Opax (OUC Opax) numbers and the corresponding tentative descripitve names*

OUC number[a]	Description[a]	References[b]
020	*Amphinema*-like	B
030	*Cenococcum*-like	B
040	*Cortinarius*-like I	A
041	*Cortinarius*-like II	A
051	*Dermocybe*-like	A
060	E - Strain-like I	B
084	*Hebeloma*-like	B
095	*Inocybe*-like	B
115	ITE. 4-like	B
120	ITE. 5-like	B
125	ITE. 6-like	B
130	*Laccaria*-like	A
148	*Lactarius pubescens*-like	
150	*Lactarius rufus*-like	B
170	Mycelium *radicis atrovirens* (MRA)-like	B
222	*Russula illota*-like	A
223	*Russula*-like	A
230	*Suillis*-like	A
234	*Suillis plorans*-like	A
240	*Thelephora terrestris*-like	B
250	*Tomentella*-like I	A
252	*Tomentella*-like II	A
440	unknown	
441	unknown	
510	unknown	
520	unknown	
530	unknown	
540	unknown	

a Descriptions of OUC Opax numbers (numbers refer to a specific morphological type) are on file at Okanagan University College, Kelowna, B.C.

b References: A = Agerer (1988); B = Ingleby et al. (1990).

TABLE 3 *Mean values ± SE for percent colonization and for Simpson's diversity of ectomycorrhizal types found on selected plant species from undisturbed and disturbed areas at two elevations on the Opax Mountain site in the fall of 1996*

Tree species	% Colonization[a]	Simpson's diversity[b]	Shannon's diversity[b]
Mud Lake			
Undisturbed			
Arctostaphylos uva-ursi	40.1 ± 3.3	0.73 ± 0.06a	1.08 ± 0.12
Betula papyrifera	33.7 ± 8.7	0.63 ± 0.07a	0.82 ± 0.13
Populus tremuloides	22.6 ± 7.3	0.64 ± 0.09a	0.89 ± 0.15
Pseudotsuga menziesii	37.7 ± 7.3	0.83 ± 0.05a	1.26 ± 0.11
Salix commutata	28.5 ± 11.5	0.55 ± 0.10a	0.68 ± 0.15
Harvested			
Arctostaphylos uva-ursi	39.9 ± 3.4	0.40 ± 0.07ab	0.75 ± 0.15
Betula papyrifera	34.7 ± 3.3	0.40 ± 0.05ab	0.68 ± 0.09
Populus tremuloides	32.6 ± 2.4	0.55 ± 0.05ab	1.08 ± 0.14
Pseudotsuga menziesii	24.8 ± 8.3	0.30 ± 0.01bc	0.62 ± 0.19
Salix commutata	25.7 ± 6.6	0.43 ± 0.07ab	0.75 ± 0.14
Opax Mountain			
Undisturbed			
Alnus viridis	13.0 ± 10.4	0.34 ± 0.13	0.40 ± 0.17
Arctostaphylos uva-ursi	39.2 ± 5.4	0.63 ± 0.09	0.88 ± 0.13
Populus tremuloides	34.0 ± 10.5	0.68 ± 0.07	0.91 ± 0.12
Pseudotsuga menziesii	24.9 ± 3.6	0.49 ± 0.12	0.63 ± 0.17
Salix commutata	34.8 ± 5.9	0.37 ± 0.07	0.45 ± 0.10
Harvested			
Alnus viridis	10.9 ± 3.2	0.14 ± 0.09	0.13 ± 0.07
Arctostaphylos uva-ursi	27.4 ± 9.6	0.57 ± 0.11	0.54 ± 0.13
Populus tremuloides	19.4 ± 2.3	0.41 ± 0.16	0.60 ± 0.24
Pseudotsuga menziesii	21.2 ± 4.9	0.53 ± 0.10	0.54 ± 0.11
Salix commutata	36.4 ± 9.0	0.63 ± 0.09	0.79 ± 0.15

a Percent colonization data (percentage of live and dead roots colonized with ectomycorrhizae) were not analyzed statistically.

b Values with different letters within a given elevation are significantly different at $p \leq 0.05$ as indicated by a one-way ANOVA. Differences between plant species or harvest treatement were determined using a Tukey pairwise comparison.

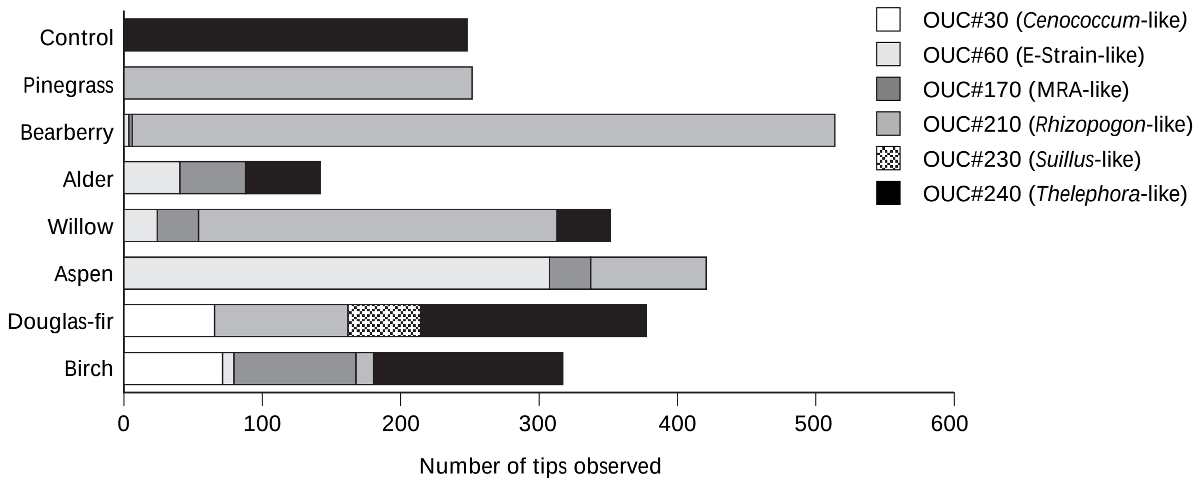

FIGURE 1 *Relative abundance of ectomycorrhizal types from a greenhouse bioassay with roots of different plant species used as inoculum.*

SUMMARY AND CONCLUSIONS

The field and greenhouse bioassay data support previous evidence (Molina and Trappe 1982; Amaranthus and Perry 1989; Hunt 1992) that woody angiosperms can provide refuge for ectomycorrhizal fungal inoculum after harvesting and that this inoculum can potentially colonize young roots of newly planted seedlings. The data also support the notion that disturbed ecosystems rarely establish "from scratch"; instead, residual soil microorganisms, especially mycorrhizal fungi, hold the system together and with their associations with the remaining plants, pull the system back up "by their bootstraps" (Perry et al. 1989b).

In the dry-belt IDF, the plants remaining following tree harvesting that are most likely to "bootstrap" the disturbed system are: advanced regenerating Douglas fir; *A. uva-ursi*, *B. papyrifera*, *S. commutata*, and *P. tremuloides*. At the Opax and Mud Lake sites, *A. uva-ursi* and advanced regenerated Douglas-fir were the most abundant of these species on the 1.7-ha cutblocks. Future studies will examine the ability of these two plant species to provide refuge for ectomycorrhizal fungal inoculum following tree harvesting.

ACKNOWLEDGEMENTS

We thank the B.C. Ministry of Forests, Research Branch, Kamloops Region for funding from 1994 to 1995 and Forest Renewal BC for funding in 1996. We acknowledge the technical contribution of Jody Sommerfeld, Melanie Walker, Kevin Glowa, and Tara Wiley.

Abuzinadah, R.A. and D.J. Read. 1986. The role of proteins in the nitrogen nutrition of ectomycorrhizal plants. I. Utilization of peptides and proteins by ectomycorrhizal fungi. New Phytologist 103:481–93.

Agerer, R. 1987–1995. Colour atlas of ectomycorrhizae. Schwäbisch Gmünd, Germany. Eihorn-Verlag Eduard Dietenberger.

Amaranthus, M.P. and D.A. Perry. 1989. Interaction effects of vegetation type and Pacific madrone soil inocula on survival, growth, and mycorrhiza formation of Douglas-fir. Canadian Journal of Forest Research 19:550–6.

Harvey, A.E., M.F. Jurgensen, and M.J. Larsen. 1980. Clear-cut harvesting and ectomycorrhizae: survival of activity on residual roots and influence on a bordering forest stand in western Montana. Canadian Journal of Forest Research 10:300–3.

Hunt, G.A. 1992. Effects of mycorrhizal fungi on quality of nursery stock and plantation performance in the Southern Interior of British Columbia. Final report, FRDA Project F52-41-006.

Goodman, D.M., D.M. Durall, J.A. Trofymow, and S.M. Berch (editors). 1996–1997. A manual of concise descriptions of North American Ectomycorrhizae. Mycologue Publications, Sidney, B.C.

Ingleby K., P.A. Mason, F.T. Last, and L.V. Fleming. 1990. Identification of ectomycorrhizas. HMSO, London, U.K. ITE Research Publication No. 5.

Molina, R. and J.M. Trappe. 1982. Lack of mycorrhizal specificity by the ericaceous hosts *Arbutus menziesii* and *Arctostaphylos uva-ursi*. New Phytologist 90:495–509.

Parke, J.L., R.G. Linderman, and C.H. Black. 1983. The role of ectomycorrhizae in drought tolerance of Douglas-fir seedlings. New Phytologist 95:83–95.

Parke, J.L., R.G. Linderman, and J.M. Trappe. 1984. Inoculum potential of ectomycorrhizal fungi in forest soils of southwest Oregon and northern California. Forest Science 30:300–4.

Parsons, W.F.J., S.L. Miller, and D.H. Knight. 1994. Root-gap dynamics in lodgepole pine forest: ectomycorrhizal and nonmycorrhizal fine root activity after experimental gap formation. Canadian Journal of Forest Research 24:1531–8.

Perry, D.A., M.P. Amaranthus, J.G. Borchers, S.L. Borchers, and R.E. Brainerd. 1989. Bootstrapping in ecosystems: internal interactions largely determine productivity and stability in biological systems with strong positive feedback. BioScience 39:230–7.

Perry, D.A., H. Margolis, C. Choquette, R. Molina, and J.M. Trappe. 1989. Ectomycorrhizal mediation of competition between coniferous tree species. New Phytologist 112:501–11.

Perry, D.A., R. Molina, and M.P. Amaranthus. 1987. Mycorrhizae, mycorrhizospheres, and reforestation: current knowledge and research needs. Canadian Journal of Forest Research 17:929–40.

Simard, S.W., R. Molina, J.E. Smith, D.A. Perry, and M.D. Jones. 1997. Shared compatibility of ectomycorrhizae on *Pseudotsuga menziesii* and *Betula papyrifera* seedlings grown in mixture in soils from southern British Columbia. Canadian Journal of Forest Research 27:331–42.

Spatial and Temporal Response of Vegetation to Silvicultural Practices in Interior Douglas-fir Forests

DAVID MIEGE, DENNIS LLOYD, AND ANDRÉ ARSENAULT

INTRODUCTION

Douglas-fir forests cover a substantial part of the Southern Interior. Historically, selective harvesting has dominated silvicultural practices in the lower Interior Douglas-fir (IDF) environments, and combined with wide-scale fire suppression, cattle grazing, and numerous recreational activities, has likely affected biodiversity, ecosystem sustainability, and numerous ecological processes.

Canopy removal changes the microclimate by affecting the amount of solar radiation that reaches the forest floor. This results in more extreme temperatures and higher light availability. Without canopy interception, greater amounts of moisture reach the soil, but the loss of cover reduces humidity, and reduces soil moisture through evaporation (Chen et al. 1993; Matlack 1994). The hot, dry climate of the IDF zone often results in substantial water deficits throughout the growing season. This can lead to poor forest regeneration after harvesting.

Both grazing and forest harvesting facilitate the introduction of weeds. Overgrazing can shift the plant community composition from the bunchgrasses to other less palatable species and weedy invaders (Hope et al. 1991).

Historic management practices in the IDF forests have resulted in stand structures with high levels of understorey and intermediate regeneration, which may contribute to abnormally high levels of infestation by insects such as spruce budworm. Concurrently, the abundance of ponderosa pine and associated understorey species has decreased.

The Opax Mountain Silvicultural Systems Project is evaluating the short- and long-term (rotation length) effects of various silvicultural treatments on the ecology and dynamics of interior Douglas-fir forests. Many ecological factors and their interactions are being studied through a series of complementary studies by scientists from the B.C. Ministry of Forests and provincial universities. The long-term commitment to monitor this site provides a unique opportunity to follow these interactions, including vegetation succession, over time.

Vegetation is the most visible biological indicator of environmental change. Silvicultural, harvesting, and site preparation practices affect the rates of vegetation recovery and alter successional pathways; changes also occur in species diversity, vigour, phenology, and biomass production. None of these effects is well documented. Additionally, interactions between vegetation and wildlife, nutrient cycling, regeneration success, and microclimate are poorly understood. In general, vegetation serves as hiding cover, shelter, and a food source for many small mammals, birds, insects, and microfauna.

Understorey vegetation often impedes tree regeneration through its influence on shading and competition for moisture and nutrients, or by physically damaging seedlings through whipping or vegetation press. Following catastrophic disturbances such as logging, root systems of many understorey species serve as refugia for microfauna and microflora, including mycorrhizae which are considered essential to the long-term productivity and health of our forests. Different logging and site preparation disturbances result in varying levels of forage production and palatability for cattle. Roots and surface vegetation protect the soil and provide a source of organic matter. Widespread weed invasion and loss of species diversity can also be attributed to resource management activities in the IDF. In summary, vegetation plays a significant role in our forests and warrants study at the Opax Mountain research site.

Several forest harvesting treatments ranging from an undisturbed control to 20% uniform removal by partial cutting to clearcuts ranging in size from 0.1 to 1.6 ha were established at the Opax Mountain site. The result is numerous stand openings ranging from small canopy gaps to 1.6-ha clearcuts, undisturbed leave-strips of varying width, and residual stands of varying size and density. The resulting mosaic of cut and leave conditions provides an opportunity to study vegetation response to a range of stand densities and canopy gap sizes that likely represent a gradient of light, temperature, humidity, and snow accumulation/melt conditions. The proximity to cut edges, level of ground disturbance, and type of ecosystem also affect soil and air temperature regimes, light and soil moisture availability, other microclimatic factors, and surface substrate conditions, which influence vegetation development.

A number of complementary vegetation studies have been initiated to quantify vegetation responses to the silvicultural practices at the Opax Mountain research site. Two studies are discussed here but several others are under development. The studies presented examine: 1) the effect of cutblock edges, and 2) the abundance, size, and distribution of canopy gaps and residual stands in each of the six harvesting treatments. Vascular plants, because of their stature and importance, are the focus of the edge effect study. However, the dominant non-vascular plants including mosses, lichens, and liverworts are also being monitored. The results presented are preliminary as both the level of study and data analysis are in their infancy.

STUDY SITE

The Opax Mountain site is 20 km northwest of Kamloops, B.C. The site extends over two biogeoclimatic subzone variants—the Interior Douglas-fir Dry Cool (IDFdkl) and the Interior Douglas-fir Very Dry Hot (IDFxh2). Both IDF subzones have sites characterized by Douglas-fir climax stands and a herb-rich understorey dominated by pinegrass; however, several key vegetation differences exist between the two subzones, with common snowberry (*Symphoricarpos albus*) and ponderosa pine present in the IDFxh2, and lodgepole pine, soopolallie (*Shepherdia canadensis*), and, to a lesser extent, kinnikinnick (*Arctostaphylos uva-ursi*) present or more common in the IDFdkl. The corded beard lichen (*Usnea* spp.) is generally absent from the IDFdkl, but is a very common epiphyte in the IDFxh2. The research area consists of two sites, Opax Mountain and Mud Lake. Opax Mountain is approximately 200 m higher in elevation than Mud Lake and falls within the IDFdkl, while Mud Lake occupies the IDFxh2 subzone. Each area has been divided into six study blocks of approximately 20 ha. The following treatments were applied in a randomized block design: 20% removal uniform partial cut, 50% removal uniform partial cut, 35% retained in uncut reserves, 20% removal in patch cuts of 0.1, 0.4, and 1.6 ha, 50% removal in patch cuts of 0.1, 0.4, and 1.6 ha, and uncut controls. Harvesting at the site occurred in the winter of 1993/1994.

THE EDGE-EFFECTS STUDY

Objectives

Large areas of the IDF zone are harvested each year. This results in the creation of edges between the cutblocks and the remaining forest. The edges become a zone of transition between the relatively stable forest interior and the centre of the cutblock. Matlack (1993) showed that significant edge effects were detected in light, temperature, moisture, vapour pressure deficit, humidity, and shrub cover in forest micro-environments up to 50 m from the edge.

The study at Opax Mountain attempts to quantify some of these edge effects for the IDF. We recorded tree seedling and vegetation response at increasing distances from the edges of cutblocks, both into the opening and into the adjacent forest. We also compared north with south edges and undisturbed surface substrates with bare mineral soil. Soil temperatures were also recorded relative to edge distance.

Methods

The edge-effects study was applied to nine cutblocks in the summer of 1995 at Opax Mountain. We intended to install replicate edge-effect sampling grids at the north- and south-facing edges of the 0.4- and 1.6-ha clearcuts at both the lower Mud Lake site and the upper Opax Mountain site (Figure 1). However, because of the ecological and stand irregularities of some edges, a few of the replicates at the Opax Mountain site were split over two cutblocks (e.g., J11 and J5, J7 and K3).

Vegetation plots were established in a grid pattern. Plot distances are 6, 12, 24, and 48 m from the baseline (0 m) at the cutblock's north and south edges. These distances are equivalent to approximately 0.25, 0.5, 1, and 2 tree heights, respectively. The grid provides a continuum of points from 48 m inside the stand to 48 m out into the cutblock (Figure 2). Ten plots were installed at each distance and plot centres were marked with a piece of 1 m tall rebar. During the summer of 1995, 1240 plots were installed and monitored. Sampling was completed from mid-May to mid-June. This corresponds to the peak of the vegetative season. Then, one-half of the plots in each grid were screefed (mineral soil exposed) with hand tools to remove all vegetation and organic material, while the other half were left undisturbed.

Vegetation was recorded within 1 m^2 circular plots. Data collected included specifics of plot location, plant species, percent cover, height, reproductive status, and surface substrates. All vascular species were recorded and difficult specimens were collected for identification.

The vegetation assessments included trees, shrubs, herbs, lichens, and bryophytes. The percent cover was estimated for all plants except for the epiphytic (tree-dwelling) lichens, where an abundance scale from 1 to 5 was used. Average plant height was estimated by measuring several specimens of

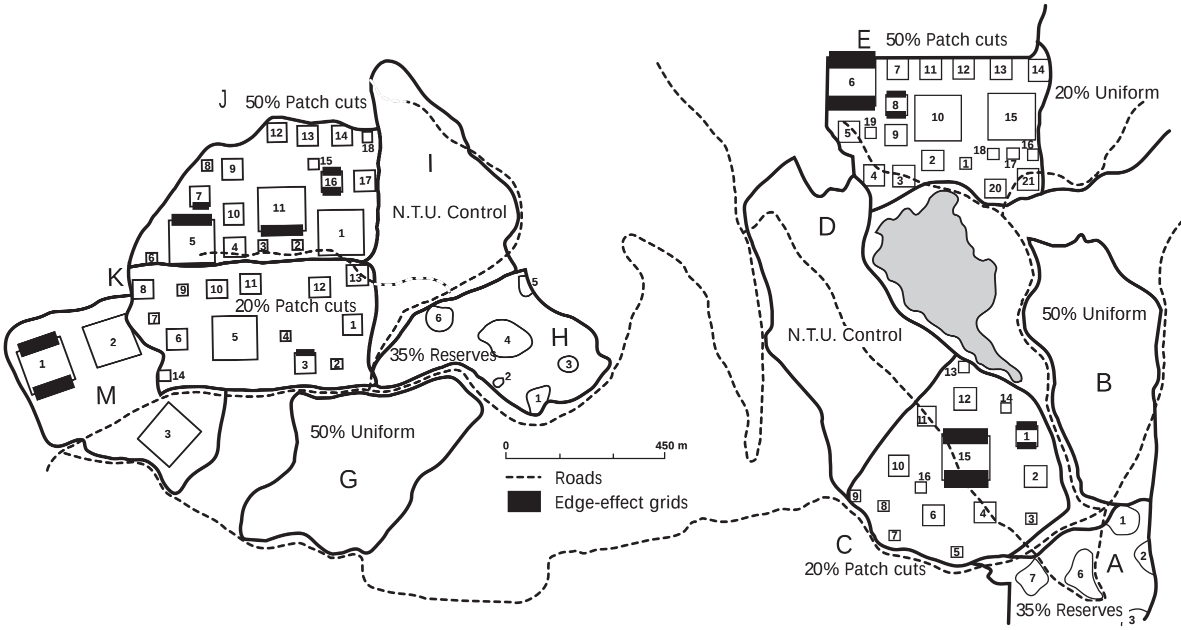

FIGURE 1 *Opax research area with edge-effect grids highlighted.*

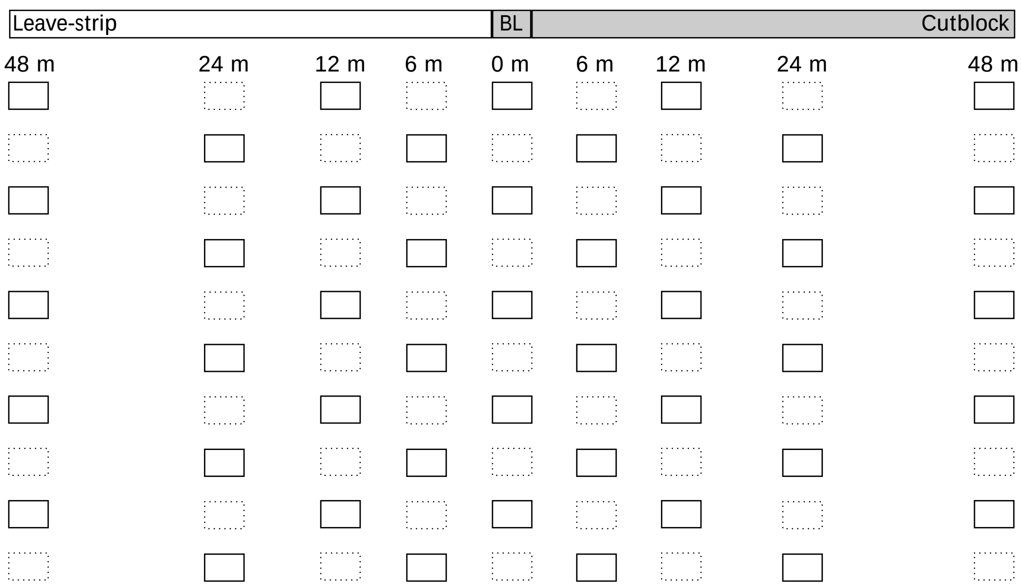

FIGURE 2 *Schematic drawing showing the sampling regime used to study the effects of forest edges on vegetation response (dotted line = screefed; solid line = undisturbed).*

each species with a metre stick. Tree height was not measured, but trees were divided into three categories: seedling (0–15 cm), sapling (15 cm–3 m), and mature (> 3 m). The reproductive status of vascular plants was classified as juvenile, nonflowering, flowering, or seedling.

Soil temperature studies were initiated in 1996. Thermocouples were installed. Four probes per plot were used: two at a 15 cm depth and two at a 1 cm depth. The wires of the 1 cm deep probes were installed to avoid large coarse woody debris and rock. Hand-held digital thermometers (OMEGA Thermocouple Thermometer Type E [450 AETD]), which were calibrated before use, were used to collect temperature data. This collection occurred between 11:00 am to 2:30 pm to minimize the time-of-day effect. Weather conditions, time of day, and date were recorded. For each cutblock edge, a fixed set of temperature sensors were sampled at the beginning and end of each sampling period to permit standardization of measurements.

Results and Discussion

Both the upper and lower areas of the Opax Mountain research area exhibited changes in plant community structure within 0–6 m of the stand edge: several trends in species abundance were observed. In the cutblock, the abundance of lichens and mosses was markedly reduced, while herb abundance increased. Shrub abundance was generally unaffected (Figure 3).

Observations of species richness suggest that the diversity of herbs and shrubs is not affected by edge distance, but lichen and bryophyte diversity is severely reduced outside of the leave strip (Figures 4a and 4b). The observed trend was similar for both Opax Mountain and Mud Lake.

Many strong trends were evident when individual species abundance and edge distance are compared. Figure 5 shows the response of 11 common species at Mud Lake. Some species are more frequent in the leave strip than in the cutblock (e.g., rattlesnake-plantain [*Goodyera oblongifolia*]). The frequency of other species (e.g., heart-leaved arnica [*Arnica cordifolia*] and pinegrass [*Calamagrostis rubescens*]) appeared equal in the leave strips and cutblocks. Finally, some species occurred more frequently in the logged area (e.g., yarrow [*Achillea millefolium*] and fireweed [*Epilobium angustifolium*]). While the frequency of occurrence of pinegrass did not vary significantly between cutblocks and leave strips, its abundance and density was significantly greater in the cutblocks.

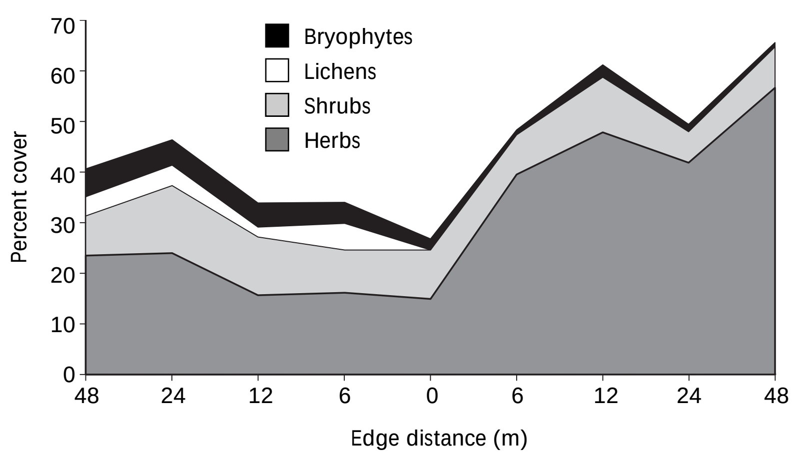

FIGURE 3 *Total average percentage cover for vegetation at the north edge of Opax Mountain cutblocks (non-screefed plots only).*

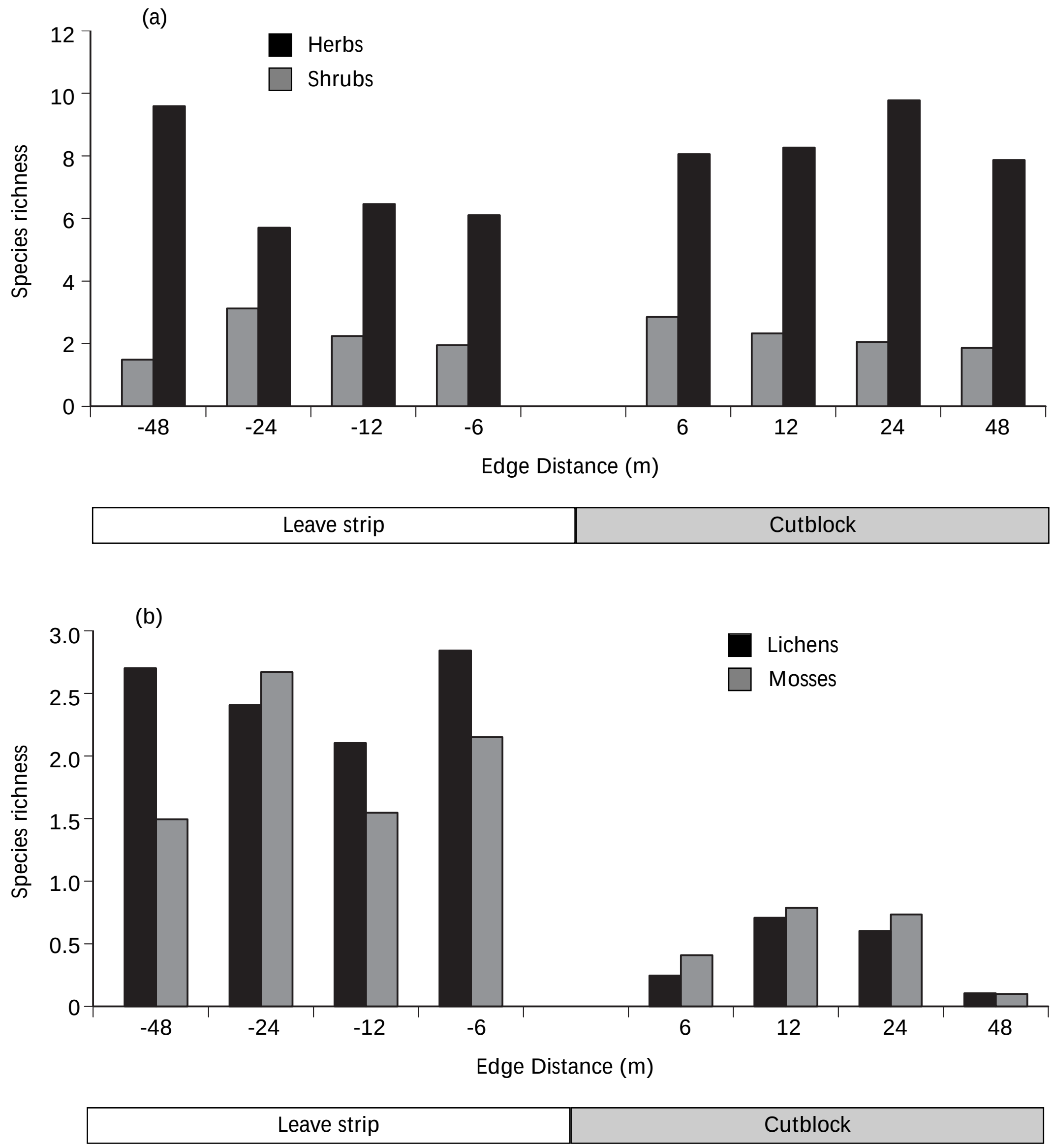

FIGURE 4 *Vascular (a) and nonvascular (b) plant species richness relative to distance from Opax Mountain cutblock north edge.*

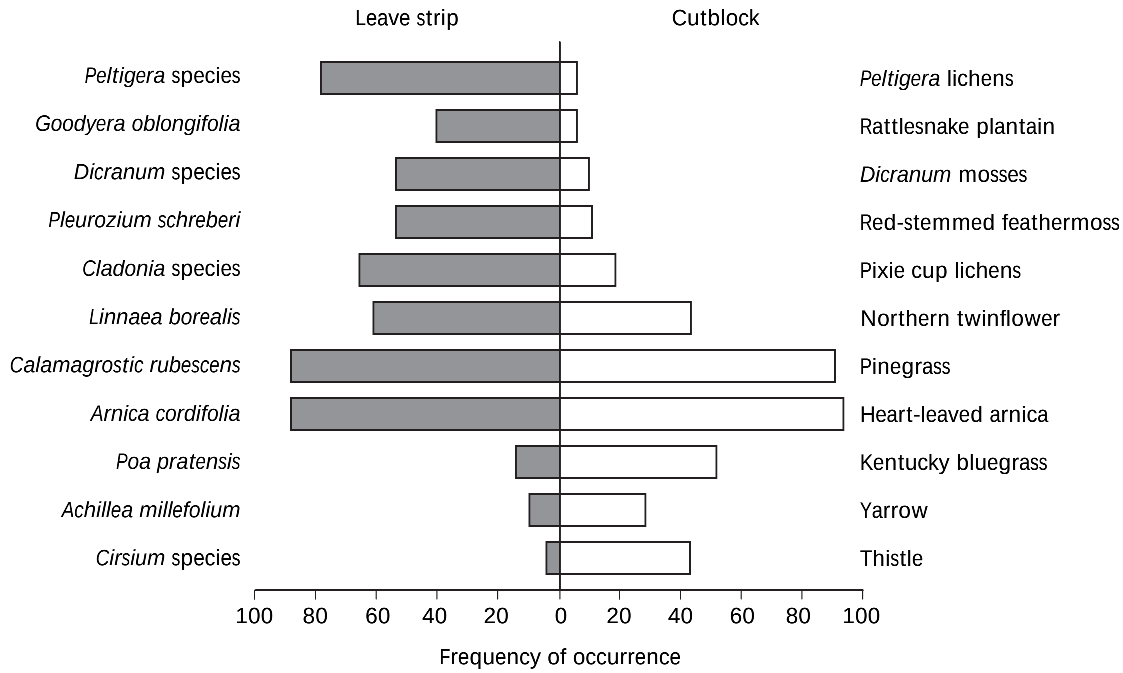

FIGURE 5 *Frequency of selected plants in the leave strips and cutblocks at Mud Lake.*

A brief analysis of the site preparation data indicated some differences in vegetation response to screefing (Figure 6). One year after site disturbance, the average percent cover of heart-leaved arnica in screefed plots was still 68% higher in undisturbed plots, while pinegrass was 90% more abundant on untreated than screefed plots. Part of this difference may be attributed to differences in the plant's root structure, seed production, seed dispersal, and ability to germinate and grow on mineral soil.

Phenological studies across the edges also yielded some interesting trends (Figure 7). Generally, flowering plants fell into three classes:

1. Species that tend to flower more frequently in the cutblock;
2. Species that flower more frequently in the leave strip; and
3. Species that flower regardless of their position relative to the edge.

During the sampling period, arnica was more likely to flower in the cutblock than in the leave strip. This effect was true for both the IDFxh2 and dkl. In contrast, pinegrass flowered in the cutblock at the Opax Mountain site, but did not flower with the same frequency or abundance at the lower-elevation Mud Lake site. Some species, such as rattlesnake plantain, black huckleberry (*Vaccinium membranaceum*), and northern twinflower (*Linnaea borealis*), flowered more often in the leave strip. The final group, including blue-eyed Mary (*Collinsia parviflora*), Oregon grape (*Mahonia aquifolium*), Kentucky bluegrass (*Poa pratensis*), dandelion (*Taraxacum officinale*), and dwarf blueberry (*Vaccinium caespitosum*), produce flowers with equal regularity in the cutblocks and the leave strips. It is expected that all annuals should produce flowers, although flowering may be restricted to cutblocks because of light requirements; for example, narrow-leaved collomia (*Collomia linearis*) flowered in the cutblocks, but did not in the leave strips or at the baseline.

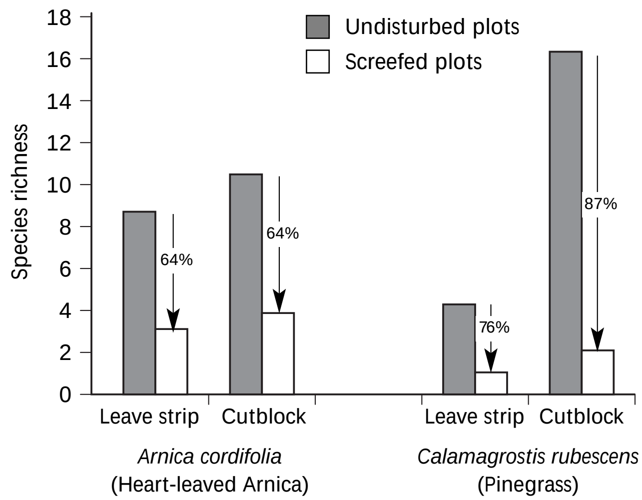

FIGURE 6 *Vegetation response to site preparation.*

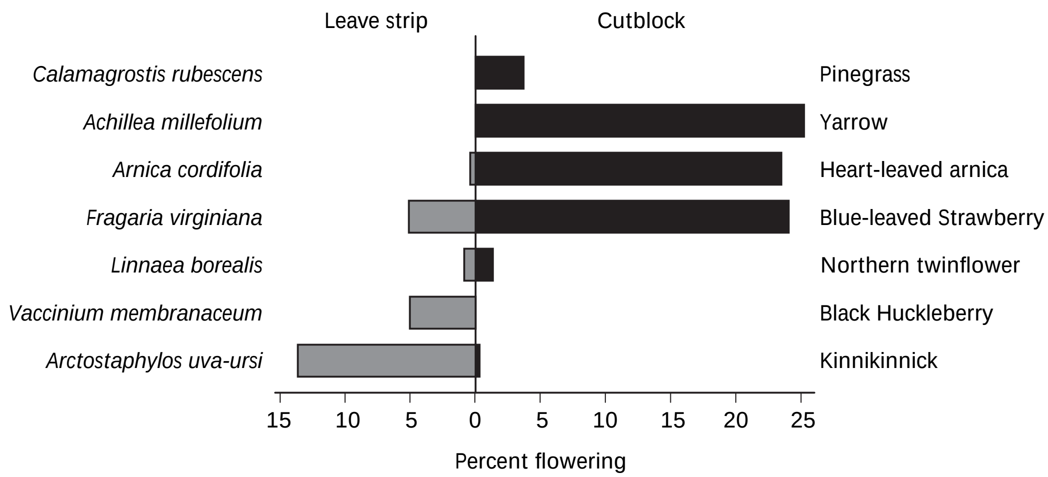

FIGURE 7 *Flowering frequency of selected species at Opax Mountain on non-screefed plots.*

Openings in the forest canopy may also affect the average height growth of some species. For example, the heights of purple-leaved willowherb (*Epilobium ciliatum*), dandelion, and Kentucky bluegrass at the Mud Lake site are all greater in the cutblock (Figure 8). This trend is observed for several other species, including pinegrass, white hawkweed (*Hieracium albiflorum*), creamy peavine (*Lathyrus ochroleucus*), and American vetch (*Vicia americana*), which are taller in the cutblocks than adjacent uncut areas. The increased height growth of hawkweed can be attributed primarily to the higher proportion of flowering individuals in the cutblock; the flowering structure of this particular species can add significantly to its height. Creamy peavine and American vetch consistently flower more often in the cutblock, but because of their morphology, it is unlikely that reproductive status adds significantly to their heights. Trends in height growth are similar for species at both the Opax Mountain and Mud Lake sites.

Differences may exist in the edge effect between the north and south edges, but we have not examined these statistically. However, because of the

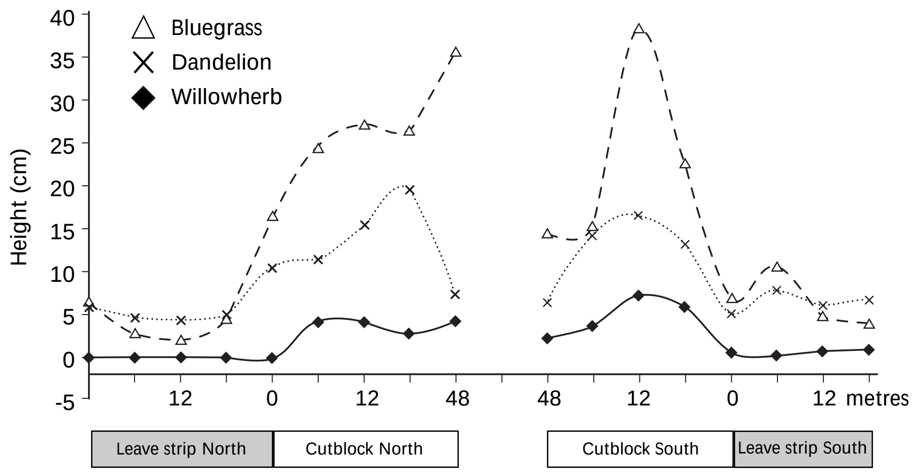

FIGURE 8 *Edge effect on height of selected species at Opax Mountain on non-screefed plots.*

increased light infiltration to the south-facing (north) edge, it is likely that the influence of added light, reduced humidity, and decreased soil moisture availability would extend further into the north leave strip than the south.

Plant communities differed between the leave strip and cutblock, and between the IDFdkl and IDFxh2. Principal Components Analysis (PCA) of plant abundance displays differences in community composition (Figure 9). The difference in species composition between cutblock plots ("1") and the leave-strip plots ("2") are well illustrated on the ordination. The differences between cutblock plots and leave-strip plots on the first two PCA components were statistically significant. Note also the different pattern generated for Mud Lake (left) compared to Opax Mountain (right).

There is a significant increase in the total cover of weeds in cutblocks when compared to leave strips. Also, Mud Lake has a higher cover of weeds than does Opax Mountain (Figure 10). Species considered as weeds for this analysis were: thistle (*Cirsium arvense* and *C. vulgare*), dandelion, yellow salsify (*Tragopogon dubius*), common plantain (*Plantago major*), fireweed, horsetail (*Equisetum arvense*), mullein (*Verbascum thapsus*), sow thistle (*Sonchus arvensis*), shepherd's purse (*Capsella bursa-pastoris*), and little buttercup (*Ranunculus uncinatus*).

Mean soil temperatures across the cutblock edge showed a temporal change. In addition, a strong temperature gradient occurred from the forest interior out into the cutblock. Soil temperature response tended to lag at the 15 cm depth (Figure 11). Trends at Opax Mountain were similar to those observed at Mud Lake.

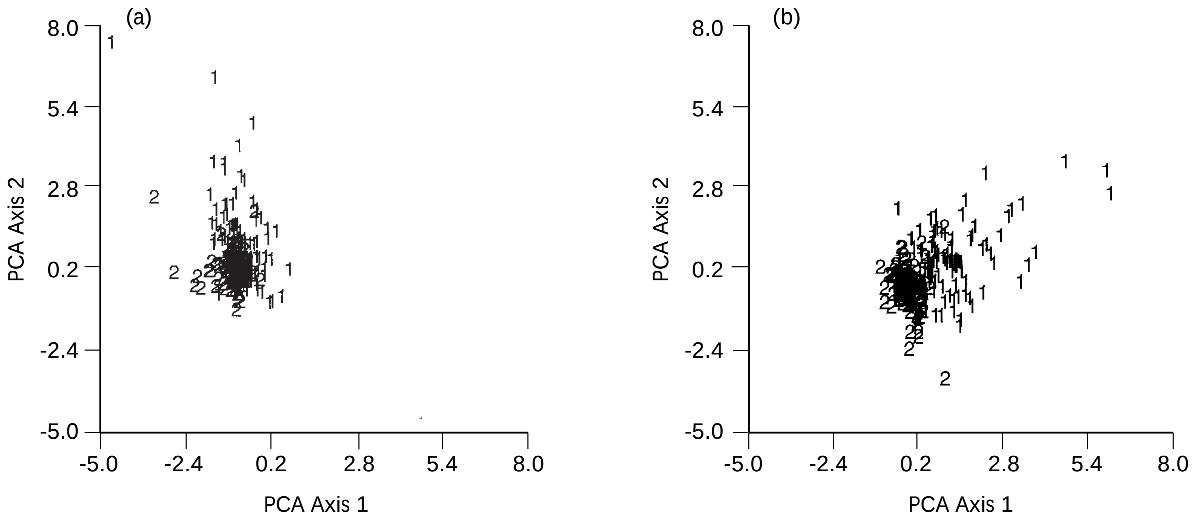

FIGURE 9 *Principle components analysis for (a) Mud Lake and (b) Opax Mountain comparing cutblock (1) and leave strip (2).*

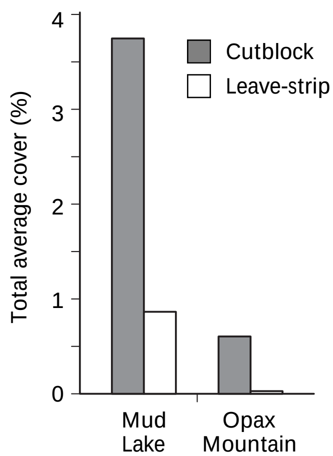

FIGURE 10 *Weed abundance at Mud Lake and Opax Mountain.*

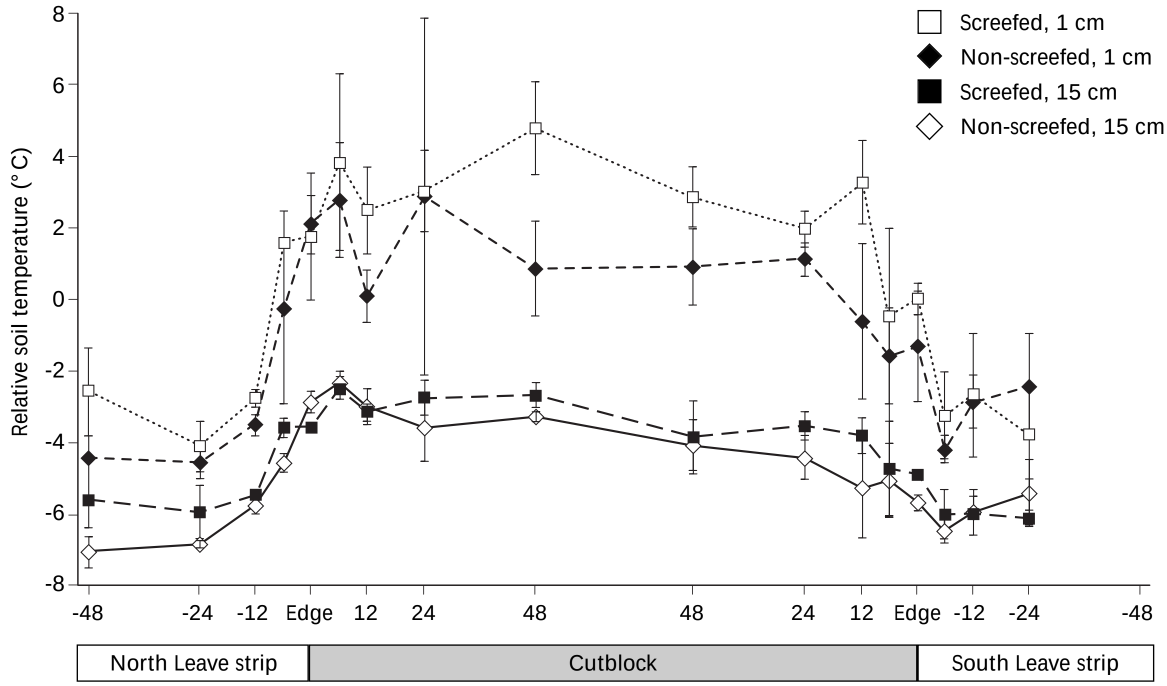

FIGURE 11 *Edge effect on soil temperature at Mud Lake, Block E6.*

THE STAND DENSITY AND CANOPY GAP STUDY

Objectives

The stand density and canopy gap studies initiated at the Opax Mountain Research Site complement the edge-effect studies and provided additional insight into the gap-phase dynamics of interior Douglas-fir forests. Several studies have been initiated, each with their own objectives, most notably:

1. to assess the effect of the silvicultural treatments on the forest canopy structure and the distribution of canopy gaps and various residual forested patches;
2. to monitor the development of vegetation along a gradient of canopy openings and residual stand densities to determine how each contributes to the variation in plant species composition, abundance, and vigour;
3. within the framework of sampling sites selected in 1 and 2 above, to determine the effect of site preparation (burning, screefing, and control) on the response of vegetation, planted seedlings, deposited tree seed of known germination quality, tree seed predation, and forage biomass production; and
4. to assess the relationship between direct and diffuse light regimes and the canopy gaps and residuals stands sampled in 2 and 3 above.

For the purpose of this paper, only objective 1 above is discussed, because the other studies have only just been initiated.

Methods

Transects were laid out in all treatment areas of the Opax Mountain site in the spring of 1996. Three or more transects running north or south were installed 50 m apart in each treatment block. Each transect was approximately 300 m long and all treatments contained about 1000 m of reconnaissance transects. Data sheets were used to describe gaps and stands found along each transect. Gaps within the stands were considered as "openings" within the canopy if they were a minimum of 25 m^2. The edge of the gap occurred where the crowns of living trees were not more than 5 m apart. Canopy gaps were measured as the vertical projection of the canopy opening as defined by the canopy of trees bordering the gap. Expanded gap measurements included the area delineated by the the boles of the trees forming the border of the canopy gap. The area of each gap was roughly estimated from the gap's length and width measurements. Six classes of gaps based on size were determined.

Results and Discussion

Silviculture treatments affect the ratio of canopy gaps to the residual stand (Figure 12). The difference between treatments is greatest when comparing the control to the 50% patch cuts. When the gaps are broken into their respective classes (Figure 13), differences in gap-size distributions between treatment units become apparent. Generally, 20% removal logging strategies tend to maintain a higher frequency of small gap sizes than the 50% removal. This also appears to be true of uniform selection cuts when compared to the patch cuts. To test these observations, ANOVA tests were performed on the ranges of gap sizes found in each harvested treatment; that is, the gap sizes for the two 20% removal treatments (patch and uniform) were grouped and tested against those of the control and the 50% removals. Similarly, data for patch cuts (20 and 50%) were grouped and tested against the uniform cuts. The ANOVA tests confirm the observations: 50% removal treatments (in general) are significantly different in gap distribution from the control ($p < 0.01$), while the 20% removal units are not. The same is true for patch

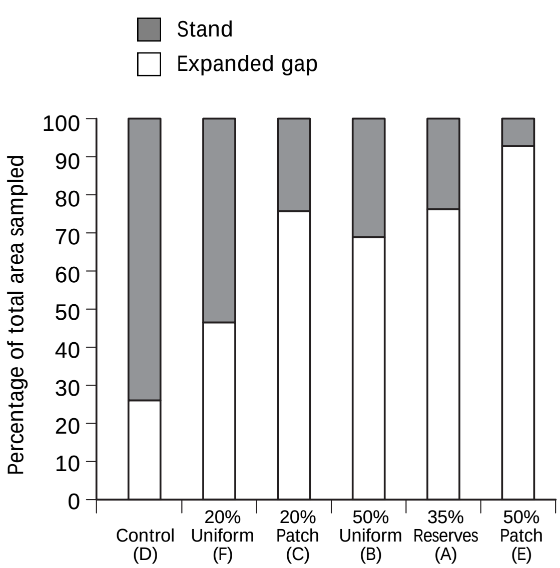

FIGURE 12 *Proportion of treatment units composed of expanded gap at Mud Lake.*

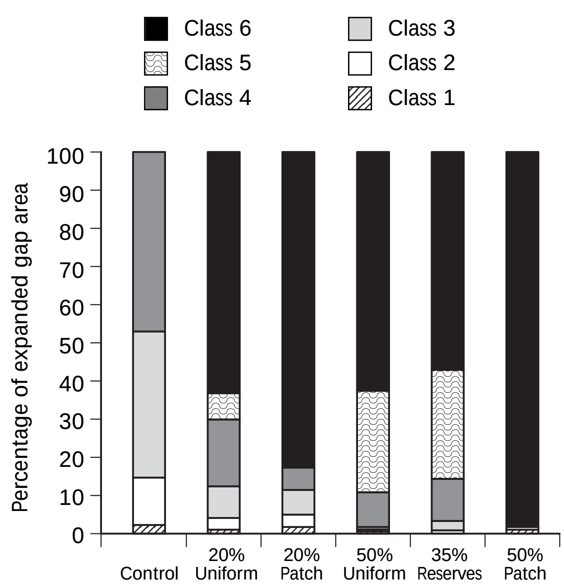

FIGURE 13 *Proportion of gap classes in each treatment unit at Mud Lake.*

cuts (significantly different from control [$p < 0.003$]) and uniform cuts (not significantly different).

Because of the volume of timber removed, the leave strips are quite narrow, and many of the natural gaps observed in the leave strips were not actually closed off by canopy trees from the cutblock. Thus, small leave-strip gaps that might have changed the character of the gap-size distribution could actually only be recorded as extensions of the cutblock openings.

The uniform cut with 35% of the forest retained as reserves was unique in that the gaps here appear to form the matrix of the landscape with the forested reserves as "islands." The very large sizes and irregular shapes of gaps made them extremely difficult to quantify with our sampling protocol; for some of the canopy gaps, it was nearly impossible to define their beginning and end. The combination of large gaps with the reserves, which retained a wide range of smaller gap sizes, makes this treatment unit the one with the greatest variation in gap sizes among the treatments.

Lichen and bryophyte species richness decreased with increasing canopy gap size, but herb and shrub diversity is not affected (Figures 14 and 15).

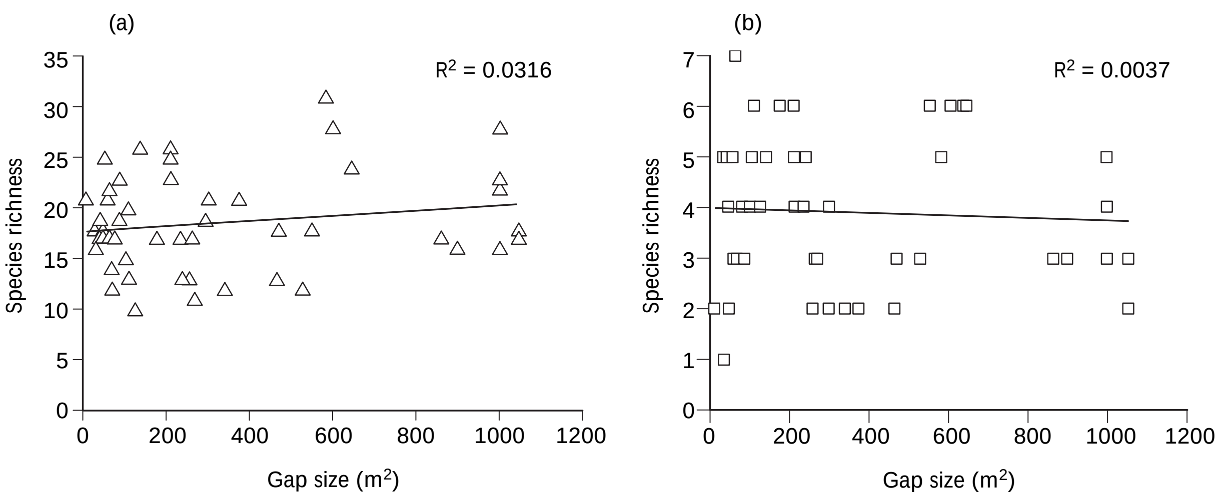

FIGURE 14 *Vascular species richness with increase in gap size at Mud Lake: (a) herb species, (b) shrub species.*

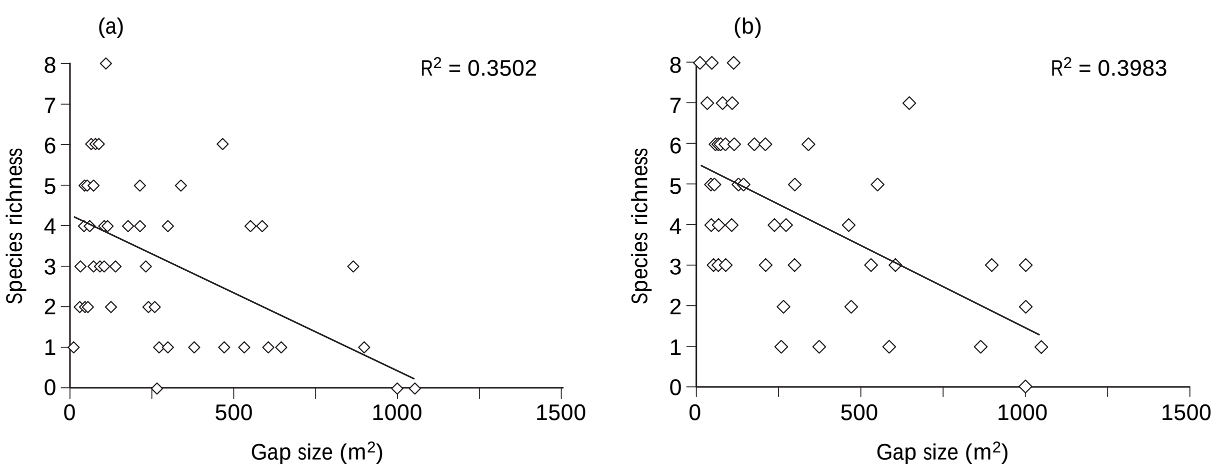

FIGURE 15 *Nonvascular species richness with increase in gap size at Mud Lake: (a) lichen species, (b) bryophyte species.*

The high degree of variability between gaps of the same class size will need to be examined further. Possible causes of the variation are:

- variation in stand densities on the southern edge;
- edaphic characteristics;
- aspect;
- height of the trees creating the boundary to the gap;
- proximity to other gaps; and
- gap etiology (i.e., partial or patch cutting, root rot, windthrow, fire).

A gap recently created by logging activity may have experienced more disturbance to the forest floor than a gap formed by the natural death of one or two trees; consequently, some of the variation may diminish after the vegetation recovers.

SUMMARY OF RESULTS

- Vascular plants (herbs and shrubs) have a greater abundance in large canopy gaps and cutblocks compared to small gaps and leave strips; species richness (diversity), however, is not greater.
- Nonvascular plants (lichens and mosses) are less abundant and suffer a loss of diversity in the larger canopy gaps and in leave strips when compared to cutblocks. However, only the dominant nonvascular plants were monitored and epiphytic lichen species were not included when calculating species diversity. The actual loss of species diversity in the cutblock is undoubtedly much greater than expressed here.
- Plant communities in the leave strip are different from those in the cutblock. Community differences also exist between Mud Lake and Opax Mountain.
- Weed species are more abundant in cutblocks. After logging, some plant species do not thrive in cutblocks, while others (often weeds) do.
- Edge effects are recognized in plant phenology and height.
- Soil temperatures (at midday) are greater in the cutblock than in the leave strip. Screefed areas are warmer, in general, than those left undisturbed.
- The larger the volume of wood removed, the more canopy gap structure is changed from a range of small- and medium-sized gaps to one dominated by very large gaps.
- Our data suggest that lichens and bryophytes might be more sensitive indicators of environmental change than the vascular plants. It is important to improve our understanding of their distribution and ecology to ensure that interpretations from nonvascular plant data are accurate and complete.

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REFERENCES

Chen, J., J.F. Franklin, and T.A. Spies. 1993. Contrasting microclimates among clearcut, edge, and interior of old-growth Douglas-fir forest. Agriculture and Forest Meteorology 63:219–37.

Hope G.D., W.R. Mitchell, D.A. Lloyd, W.R. Erickson, W.L. Harper, and B.M. Wikeem. 1991. Interior Douglas-fir Zone. *In* Ecosystems of British Columbia. D. Meidinger and J. Pojar (editors). B.C. Ministry of Forests, Victoria, B.C. Special Report Series No. 6, pp. 153–66.

Matlack, G.R. 1993. Microenvironment variation within and among forest edge sites in the eastern United States. Biological Conservation 66:185–94.

______. 1994. Vegetation dynamics of the forest edge: trends in space and successional time. Journal of Ecology 82:113–23.

Faunal Biodiversity Studies at Opax Mountain: Background and Rationale for the Choice of Monitored Indicator Groups

WALT KLENNER AND DAVE HUGGARD

INTRODUCTION

Maintaining biodiversity is now recognized as an important land-use management objective, both provincially and globally (Wilson 1988; Salwasser 1990; Fenger et al. 1993). Initiatives to protect endangered species or species of management concern have co-evolved with forest management in British Columbia over the last 50 years, but with the implementation of the Forest Practices Code, the maintenance of biodiversity has become an important obligation for forest managers. To maintain species, as well as genetic and ecosystem diversity, biodiversity guidelines (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1995) recommend targets for seral stages, habitat patterns (patch size and connectivity), and habitat structures, such as green trees, snags, and downed wood. This "coarse filter" approach to habitat management (Hunter 1990) represents an interim and manageable surrogate for maintaining biodiversity, in contrast to the unwieldy alternative of managing each of the thousands of species and ecological processes on an individual basis. However, there is little direct evidence to indicate whether the habitat targets specified in the biodiversity guidelines are adequate for maintaining biodiversity, or whether they are too conservative and thus represent an excessive constraint to the timber industry. These are important issues because the consequences of not achieving biodiversity objectives or of exacting unnecessary constraints on industry both have a high cost.

Before our forests were managed for timber commodities, a wide range of natural disturbances (e.g., wildfire, insect attack, windthrow)(Canham and Marks 1985; Runkle 1985; Agee 1991) and aboriginal burning (Kay 1995) influenced forest ecosystems. These events created a diverse mosaic of seral stages and patch sizes across the landscape (Lehmkuhl et al. 1991; Mladenoff et al. 1993; Spies and Franklin 1991). Within stands, an abundance of habitat structures such as snags and downed wood was maintained (Franklin and Spies 1991 a, b; Spies and Franklin 1991). This complex and heterogeneous mosaic of habitat structures and patterns is important in maintaining high levels of biodiversity (Miller 1982; Denslow 1985; Karr and Freemark 1985; Shugart and Seagle 1985). Many species use or depend on these habitat features for food, breeding sites, or cover. In British Columbia, for example, more than 90 species of vertebrates use large declining trees or snags for nesting, foraging, or cover (Backhouse and Lousier 1991). Other habitat structures such as downed wood (Anderson 1986; Barnum et al. 1992);

a deep layer of forest floor litter, terrestrial moss, and lichens (Seastedt and Crossley 1987); grasses and forbs (van Horne 1981); and arboreal lichens (Stevenson 1979; Stevenson and Hatler 1985; Rominger and Oldemeyer 1989; Waterhouse et al. 1991) provide important habitat for other biota.

The abundance and distribution of several habitat structures will likely differ between managed and natural forests (Cline et al. 1980; Zarnowitz and Manuwal 1985; Spies and Cline 1988; Hansen 1990; Franklin and Spies 1991 a, b; Spies and Franklin 1991; Lee et al. 1995 a, b; Roy et al. 1995; Schieck and Nietfeld 1995). The numerous large declining green trees, snags, and abundant downed wood characteristic of old-growth stands will decline in managed forests unless special practices are implemented to maintain these features (Swanson and Franklin 1992). Largely unknown are the effects of canopy removal through partial or clearcutting, or the influence and duration of site preparation disturbances on the forest floor litter layer and its associated fauna. In addition, the effect of reducing several habitat structures may be offset by the enhancement of others. For example, increased grass, forb, and shrub production in large openings may compensate for reduced levels of downed wood. Alternatively, the effect of forest harvesting on some forest-dependent species may be exacerbated by the biota (e.g., nest predators) that flourish in early seral habitats (Wilcove 1985; Lehmkuhl and Ruggiero 1991).

Along with changes in habitat structures, forest harvesting will modify the temporal and spatial distribution of habitat types. Increased amounts of edge, a decrease in the complexity of edges, and an increase in the interspersion of early and late seral habitats may have both short- and long-term implications for maintaining biodiversity (Franklin and Forman 1987; Lehmkuhl and Ruggiero 1991; Mladenoff et al. 1993; Saunders et al. 1991). These changes will benefit some species and be detrimental to others, as the foraging, breeding, or cover capabilities of the habitats are modified.

RESEARCH APPROACH

Assessing the effects of forest management practices on each of the diverse array of species and processes that comprise biodiversity is an impossible task. To reduce the complexity of the problem, our approach was:

1. to identify the important habitat structures and patterns that are likely to diminish or change as a result of forest management activities;
2. to document the magnitude of these habitat changes at the Opax Mountain forest harvesting trials site;
3. to monitor the effects of these changes on a range of indicator species, or groups of species that are likely to respond to the habitat structures and patterns being modified; and
4. to develop predictive models that incorporate the results from the Opax Mountain research trials and other related studies into temporally and spatially explicit habitat supply models (e.g., Klenner et al. 1995, 1997).

The heterogeneity of the post-harvest treatments at the Opax Mountain site allows for two complementary analyses. The first uses an analysis of variance (ANOVA) approach to assess the overall treatment effects on the monitored taxa. The second uses a regression-prediction approach to analyze the variation in habitat structures and patterns within and across treatments in relation to the monitored fauna to develop more detailed habitat suitability models.

Eight faunal indicator groups were chosen for monitoring at Opax Mountain. These particular groups were chosen for two reasons:

- Information on their natural history suggests that they require specific habitat structures or patterns that will likely diminish or change in managed forests; and
- The groups or species within a group are likely to perceive habitat patterns at different scales (see Table 1) (e.g., a home range of 0.1 ha for shrews vs. 100 ha for small carnivores).

For some of the species being monitored (e.g., terrestrial arthropods, amphibians, shrews), the spatial scale of the experimental treatment units (approximately 25 ha each) is suitable for studies that focus on demographic changes. However, to assess the response of wide-ranging species (e.g., hairy woodpeckers [*Picoides villosus*], coyote [*Canis latrans*], and pine marten [*Martes americana*]) to the habitat changes arising from the experimental treatments, the foraging and nesting behaviour of individuals was monitored with radiotelemetry or snow tracking. This approach focuses on the behavioural changes (e.g., time budgets, foraging, nest site selection) of individuals, and the use of habitat structures and patterns and their availability. This approach represents the necessary and workable compromise inherent in any fixed-scale experimental study that addresses a range of species with different spatial requirements, or where wide-ranging species are studied in relation to specific harvesting treatments.

The eight monitored groups span a diverse range of taxa and reflect various trophic interactions (plant–herbivore–predator). Shifts in the population density, demography, home range, habitat use, and community structure of these groups should provide reliable indicators of changes in overall faunal biodiversity at Opax Mountain. Ongoing studies by T. Dickinson on passerine birds, D. Lloyd and A. Arsenault on flora, and L. Maclauchlan on forest insects complement our approach and will provide a more complete picture of changes in biodiversity that arise from the experimental treatments. Large, wide-ranging vertebrates such as black bears (*Ursus americanus*), cougars (*Felis concolor*), or goshawks (*Accipiter gentilis*) are not frequently encountered at the Opax Mountain site, and hence the effects of the research treatments on these and other species can only be inferred from changes in the preferred habitat and prey populations that we monitor.

The experimental harvesting and site preparation treatments at the Opax Mountain site present an opportunity to evaluate the responses of various fauna to changes in canopy structure (e.g., the abundance of live trees or snags) or understorey vegetation. Either through design or operational necessity (e.g., the removal of snags in the individual-tree selection, partial-cut treatments to comply with worker safety requirements), the treatment areas represent a gradient of opening sizes, habitat structure conditions, and edge patterns (Figure 1). Responses to these conditions by the monitored faunal groups can then be extrapolated to a wider range of habitat structure and pattern conditions associated with operational management practices in Interior Douglas-fir zone forests.

TABLE 1 *Indicator groups being monitored to assess the effects of forest harvesting treatments on faunal biodiversity at Opax Mountain*

Taxon	Home range size (ha)	Possible important habitat structures	Possible important habitat patterns	Possible ecological interactions
Ground-dwelling arthropods (many orders, families, and species)	0.001– 0.01	– terrestrial moss, lichens, and forest floor litter – grass, forbs, and shrubs – downed wood	– habitat edges	– competition and predation within the group – predation by shrews and amphibians
Amphibians: Long-toed salamander Western toad Pacific tree frog Spotted frog	0.01– 0.1	– riparian habitat – terrestrial moss, lichens, and forest floor litter – downed wood – grass, forbs, and shrubs	– habitat edges – use of forested corridors	– predation on arthropods
Shrews: Masked shrew Pigmy shrew Montane shrew Vagrant shrew	0.01– 0.1	– downed wood – grass, forbs, and shrubs – terrestrial moss, lichens, and forest floor litter – canopy cover	– habitat edges – use of leave strips	– competition and predation within the group – predation on arthropods
Mice and voles: Red-backed vole Long-tailed vole Meadow vole Deer mouse	0.1–1.0	– downed wood – grass, forbs, and shrubs – terrestrial moss, lichens, and forest floor litter – canopy cover	– habitat edges – use of forested corridors – habitat dilution within home range	– competition within the group – predation by small carnivores
Sciurids: Yellow-pine chipmunk Flying squirrel Red squirrel	1.0 – 10	– mature hardwoods and conifers – snags – canopy cover – downed wood	– habitat edges – use of forested corridors – habitat dilution within home range	– competition within the group – predation by small carnivores
Passerine birds: Many species	1.0 – 10	– conifers – canopy cover – grass, forbs, and shrubs	– habitat edges – use of forested corridors – habitat dilution within home range	
Cavity-nesting birds: Boreal chickadee Red-breasted nuthatch Black-backed woodpecker Three-toed woodpecker Common flicker	1.0 – 100+ 1.0 – 10 10 – 100+	– snags – hardwoods and conifers – downed wood	– habitat edges – use of forested corridors – habitat dilution within home range	
Small carnivores and ungulates (winter only): Short-tailed weasel Coyote Marten Mule deer Moose	10–1000+	– canopy cover – downed wood – hardwoods and conifers – snags – grass, forbs, and shrubs	– habitat edges – use of forested corridors – habitat dilution within home range	– predation and competition within the group – predation on small mammals

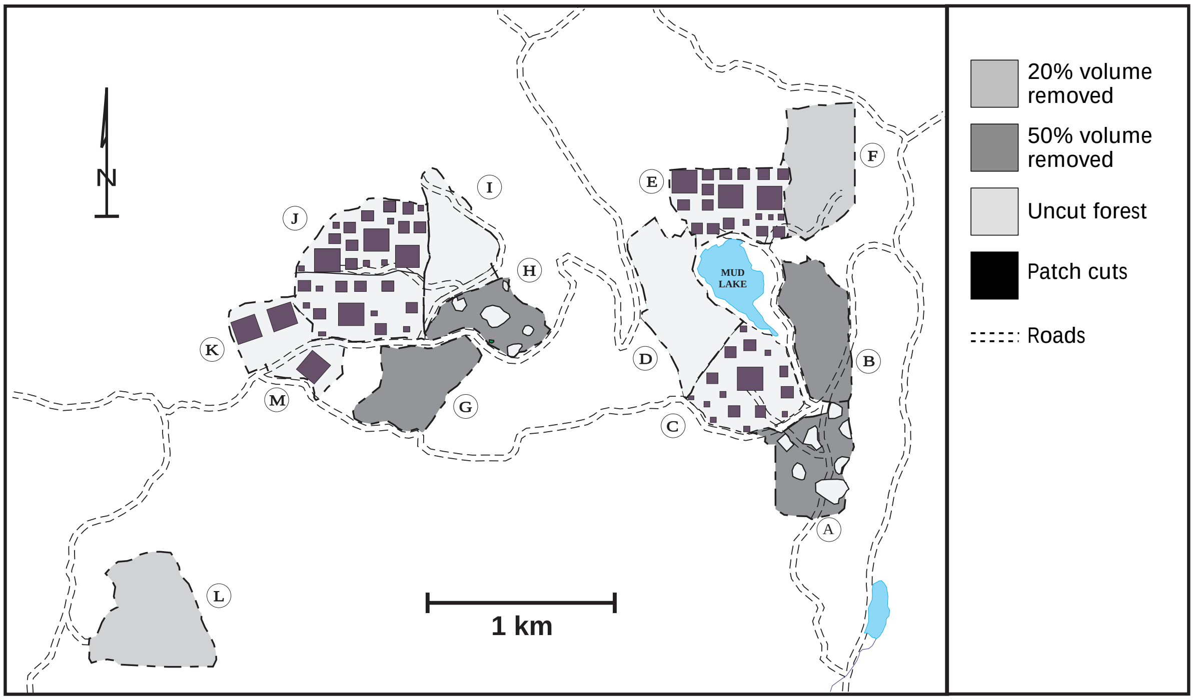

FIGURE 1 *Harvesting treatments at the Opax Mountain research site: 20% volume removal using individual-tree selection (units F, L); 35% volume removal, consisting of 75% of the treatment unit area harvested as 50% volume removal using individual-tree selection, and 25% of the treatment unit area retained as uncut reserves (units A, H); 50% volume removal using individual-tree selection (units B, G); patch cuts of 0.1, 0.4, and 1.6 ha on 20% of the treatment unit area (units C, K); patch cuts of 0.1, 0.4, and 1.6 ha on 50% of the treatment unit area (units E, J); uncut controls (units D, I).*

These silvicultural systems trials are important from another perspective as well. The project infrastructure (e.g., roads, GPS locations of study sites, co-ordination of treatments and studies) will be important in the long-term monitoring of these sites, and for integrating the numerous interdisciplinary research projects into prescriptions that can be applied on an operational basis. Harvesting and subsequent management actions, such as site preparation and planting, initiate a series of seral changes, which eventually lead to the establishment of a mature coniferous forest. The short-term responses of various biota will likely diminish as forest cover again dominates the site, and the habitat structures found in the original forest develop. The duration of these effects is not well known, but is probably species-specific. For example, shrew abundance and diversity may not be affected; it may change, but then return to pre-harvest levels on the treated sites within 10 years; or, the greatest effects may not occur until several decades after harvesting. The study site infrastructure of the Opax Mountain silvicultural systems trials will facilitate and encourage the long-term monitoring of habitat changes and the testing of habitat suitability models over time.

The rationale for choosing each monitored indicator group is outlined in the following series of reports. Two approaches are being used to study these species. Initially, the monitoring effort is focusing on the development of habitat suitability models based on use of the habitat structures or patterns being measured. If species abundance or other parameters show a high correlation with specific habitat features, a more detailed hypothesis-testing project may be initiated. For example, for the mouse and vole indicator group, the habitat suitability monitoring program is being conducted along with a hypothesis-testing manipulation of downed wood because evidence from previous studies suggests that the abundance and distribution of downed wood strongly influences these populations. A similar approach will be applied to the other indicator species as habitat suitability models are developed.

REFERENCES

Agee, J.K. 1991. Fire history of Douglas-fir forests in the Pacific Northwest. *In* Wildlife and vegetation of unmanaged Douglas-fir forests. K.B. Aubry, M.H. Brookes, J.K. Agee, R.G. Anthony, and J.F. Franklin (editors). U.S. Department of Agriculture, Forest Service, Portland, Oreg. pp. 25–33.

Anderson, P.K. 1986. Foraging range in mice and voles: the role of risk. Canadian Journal of Zoology 64:2645–53.

Backhouse, F. and J.D. Louiser. 1991. Silviculture systems research: wildlife tree problem analysis. B.C. Ministry of Forests and B.C. Ministry of Environment, Wildlife Tree Committee, Victoria, B.C.

Barnum, S.A., C.J. Manville, J.R. Tester, and W.J. Carmen. 1992. Path selection by *Peromyscus leucopus* in the presence and absence of vegetative cover. Journal of Mammalogy 73:797–801.

British Columbia Ministry of Forests and B.C. Ministry of Environment, Lands and Parks. 1995. Biodiversity guidebook. B.C. Forest Practices Code, Victoria, B.C.

Canham, C.D. and P.L. Marks. 1985. The response of woody plants to disturbance: patterns of establishment and growth. *In* The ecology of natural disturbance and patch dynamics. S.T.A. Pickett and P.S. White (editors). Academic Press Inc., San Diego, Calif. pp. 197–215.

Cline, S.P., A.B. Berg, and H.M Wight. 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management 44:773–86.

Denslow, J.S. 1985. Disturbance-mediated coexistence of species. *In* The ecology of natural disturbance and patch dynamics. S.T.A. Pickett and P.S. White (editors). Academic Press Inc., San Diego, Calif. pp. 307–23.

Fenger, M.A., E.H. Miller, J.F. Johnson, and E.J.R. Williams (editors). 1993. Our living legacy: proceedings of a symposium on biological diversity. Royal British Columbia Museum, Victoria, B.C.

Franklin, J.F. and R.T.T. Forman. 1987. Creating landscape patterns by forest cutting: ecological consequences and principles. Landscape Ecology 1:5–18.

Franklin, J.F. and T.A. Spies. 1991a. Ecological definitions of old-growth Douglas-fir forests. *In* Wildlife and vegetation of unmanaged Douglas-fir forests. K.B. Aubry, M.H. Brookes, J.K. Agee, R.G. Anthony, and J.F. Franklin (editors). U.S. Department of Agriculture, Forest Service, Portland, Oreg. pp. 61–9.

______. 1991b. Composition, function and structure of old-growth Douglas-fir forests. *In* Wildlife and vegetation of unmanaged Douglas-fir forests. K.B. Aubry, M.H. Brookes, J.K. Agee, R.G. Anthony, and J.F. Franklin (editors). U.S. Department of Agriculture, Forest Service, Portland, Oreg. pp. 71–80.

Hansen, A. 1990. Managing species diversity on forest lands. *In* Symposium proceedings, The forests—wild and managed: differences and consequences. A.F. Pearson and D.A. Challenger (editors). Students for Forestry Awareness, University of British Columbia, Vancouver, B.C. pp. 109–14.

Hunter, M.L. 1990. Wildlife, forests, and forestry: principles of managing forest for biological diversity. Prentice-Hall Canada Inc., Toronto, Ont.

Karr, J.R. and K.E. Freemark. 1985. Disturbance and vertebrates: an integrative perspective. *In* The ecology of natural disturbance and patch dynamics. S.T.A. Pickett and P.S. White (editors). Academic Press Inc., San Diego, Calif. pp. 153–67.

Kay, C.E. 1995. Aboriginal overkill and Native burning: implications for modern ecosystem management. Western Journal of Applied Forestry 10:121–6.

Klenner, W., G. Sutherland, and D. Daust. 1995. Managing the spatial and temporal distribution of habitats across landscapes to maintain biodiversity. *In* Implementing forest ecosystem management: an industrial perspective. Alberta-Pacific Industries, Hinton, Alta. pp. 66–77.

Klenner, W., W.A. Kurz, and T.M. Webb. 1997. Projecting the spatial and temporal distribution of forest ecosystem characteristics. *In* Proceedings, GIS '97. GIS World, Fort Collins, Colo. pp. 418–21.

Lee, P.C., S. Crites, and J.B. Stelfox. 1995a. Changes in forest structure and floral composition in a chronosequence of aspen mixedwood stands in Alberta. *In* Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. J.B. Stelfox (editor). Alberta Environmental Centre and Canadian Forest Service, Edmonton, Alta. pp. 29–48.

Lee, P.C., S. Crites, K. Sturgess, and J.B. Stelfox. 1995b. Changes in understory composition for a chronosequence of aspen mixedwood stands in Alberta. *In* Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. J.B. Stelfox (editor). Alberta Environmental Centre and Canadian Forest Service, Edmonton, Alta. pp. 63–80.

Lehmkuhl, J.F. and L.F. Ruggiero. 1991. Forest fragmentation in the Pacific Northwest and its potential effects on wildlife. *In* Wildlife and vegetation of unmanaged Douglas-fir forests. K.B. Aubry, M.H. Brookes, J.K. Agee, R.G. Anthony, and J.F. Franklin (editors). U.S. Department of Agriculture, Forest Service, Portland, Oreg. pp. 35–46.

Lehmkuhl, J.F., L.F. Ruggiero, and P.A. Hall. 1991. Landscape-scale patterns of forest fragmentation and wildlife richness and abundance in the southern Washington Cascade Range. *In* Wildlife and vegetation of unmanaged Douglas-fir forests. K.B. Aubry, M.H. Brookes, J.K. Agee, R.G. Anthony, and J.F. Franklin (editors). U.S. Department of Agriculture, Forest Service, Portland, Oreg. pp. 425–42.

Miller, T.E. 1982. Community diversity and interactions between the size and frequency of disturbance. The American Naturalist 120:533–6.

Mladenoff, D.J., M.A. White, J. Pastor, and T.R. Crow. 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. Ecological Applications 3(2):294–306.

Pough, F.H., E.M. Smith, O.H. Rhodes and A. Collazo. 1987. The abundance of salamanders in forest stands with different histories of disturbance. Forest Ecology and Management 20:1–9.

Powers, R.F. 1989. Maintaining long-term forest productivity in the Pacific Northwest: defining the issues. *In* Maintaining the long-term productivity of Pacific Northwest forest ecosystems. D.A. Perry, R. Meurisse, B. Thomas, R. Miller, J. Boyle, J. Means, C.R. Perry, and R.F. Powers (editors). Timber Press, Portland, Oreg. pp. 3–16.

Rominger, E.M. and J.L. Oldemeyer. 1989. Early-winter habitat of woodland caribou, Selkirk Mountains, British Columbia. Journal of Wildlife Management 53:238–43.

Roy, L.D., J.B. Stelfox, and J.W. Nolan. 1995. Relationships between mammal biodiversity and stand age and structure in aspen mixedwood forests in Alberta. *In* Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. J.B. Stelfox (editor). Alberta Environmental Centre and Canadian Forest Service, Edmonton, Alta. pp. 159–89.

Runkle, J.R. 1985. Disturbance regimes in temperate forests. *In* The ecology of natural disturbance and patch dynamics. S.T.A. Pickett and P.S. White (editors). Academic Press Inc., San Diego, Calif. pp. 17–33.

Salwasser, H. 1990. Conserving biological diversity: A perspective on scope and approaches. Forest Ecology and Management 35:79–90.

Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5:18–32.

Schieck, J. and M. Nietfeld. 1995. Bird species richness and abundance in relation to stand age and structure in aspen mixedwood forests in Alberta. *In* Relationships between stand age, stand structure, and biodiversity in aspen mixedwood forests in Alberta. J.B. Stelfox (editor). Alberta Environmental Centre and Canadian Forest Service, Edmonton, Alta. pp. 115–57

Seastedt, T.R. and D.A. Crossley, Jr. 1987. Soil arthropods and their role in decomposition and mineralization processes. *In* Forest hydrology and ecology at Coweeta. W.T. Swank and D.A. Crossley, Jr. (editors). Springer-Verlag, Berlin, Germany. pp. 233–44.

Shugart, H.H. and S.W. Seagle. 1985. Modeling forest landscapes and the role of disturbance in ecosystems and communities. *In* The ecology of natural disturbance and patch dynamics. S.T.A. Pickett and P.S. White (editors). Academic Press Inc., San Diego, Calif. pp. 353–68.

Spies, T.A. and S.P. Cline. 1988. Coarse woody debris in forests and plantations of coastal Oregon. *In* From the forest to the sea: a story of fallen trees. C. Maser, R.F. Tarrant, J.M. Trappe, and J.F. Franklin (editors). U.S. Department of Agriculture, Forest Service, Portland, Oreg. General Technical Report PNW-GTR-229. pp. 5–24.

Spies, T.A., and J.F. Franklin. 1991. The structure of natural young, mature, and old-growth Douglas-fir forests in Oregon and Washington. *In* Wildlife and vegetation of unmanaged Douglas-fir forests. K.B. Aubry, M.H. Brookes, J.K. Agee, R.G. Anthony, and J.F. Franklin (editors). U.S. Department of Agriculture, Forest Service, Portland, Oreg. pp. 91–109.

Stevenson, S.K. 1979. Effects of selective logging on arboreal lichens used by Selkirk caribou. B.C. Ministry of Environment and B.C. Ministry of Forests, Victoria, B.C. Fish and Wildlife Report No. R-2.

Stevenson, S.K. and D.F. Hatler. 1985. Woodland caribou and their habitat in southern and central British Columbia. B.C. Ministry of Forests, Victoria. B.C. Land Management Report No. 23.

Swanson, F.J. and J.F. Franklin. 1992. New forestry principles from ecosystem analysis of Pacific Northwest forests. Ecological Applications 2(3):262–74.

van Horne, B. 1981. Demography of *Peromyscus maniculatus* populations in seral stages of coastal coniferous forest in southeast Alaska. Canadian Journal of Zoology 59:1045–61.

Waterhouse, M.J., H.M. Armleder, and R.J. Dawson. 1991. Forage litterfall in Douglas-fir forests in the central Interior of British Columbia. B.C. Ministry of Forests, Victoria, B.C. Research Note No. 100.

Wilcove, D.S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. Ecology 66:1211–4.

Wilson, E.O. 1988. The current state of biological diversity. *In* Biodiversity: Papers from the National Forum on Biodiversity. September 21–25, 1986, Washington, D.C. E.O. Wilson (editor). National Academy of Sciences, Washington, D.C.

Zarnowitz, J.E. and D.A. Manuwal. 1985. The effects of forest management on cavity-nesting birds in northwestern Washington. Journal of Wildlife Management 49(1):255–63.

Effects of Harvest Type and Edges on Shrews at the Opax Mountain Silvicultural Systems Site

DAVID HUGGARD AND WALT KLENNER

INTRODUCTION

Shrews play several roles in forested ecosystems. They can consume a substantial proportion of the larvae and pupae of some forest insect pests and may reduce the likelihood and severity of insect outbreaks (Hanski 1987). As predators of forest insects, shrews may also help maintain this highly diverse community. Shrews are also prey for larger forest carnivores, sustaining their populations when other small mammals are at low densities (Korpimäki and Norrdahl 1989).

Despite their potentially important role in forest ecosystems, we know very little about the ecology of shrews in British Columbia forests, or how they may be affected by forest management (Nagorsen 1996). The effects of forest harvesting on shrews may be of particular concern in interior dry-belt forests because the species found in these systems may be near their ecological limits in dry, hot forests. Shrews have high metabolic rates, and suffer rapid water loss, and are therefore thought to require moist microsites and moderate temperatures on the ground (Getz 1961; Hawes 1977). The diversity of shrew species increases locally and regionally with increased moisture (Wrigley et al. 1979; Kirkland 1991). Removing or reducing the forest canopy through harvesting creates more extreme high temperatures on the ground and may lower ground moisture through increased evaporation (Chen et al. 1993), while reducing large old coarse woody debris that provides moist microsites (Harmon et al. 1986). These changes and the rarity of shrews in adjacent open grasslands (Nagorsen 1996) suggest that harvesting dry-belt forest might lead to reduced shrew abundance and diversity. This negative effect should increase with the degree of canopy removal. Forests adjacent to cutblocks also experience higher ground temperatures and reduced humidity (Chen et al. 1995), which could have an additional negative effect on shrews.

The Opax Mountain Silvicultural Systems Project (see Klenner and Vyse, these proceedings, p. 128) provides an ideal site to test these predictions. Four species of shrews (*Sorex cinereus*, *S. monticolus*, *S. hoyi*, and *S. vagrans*) are found in this dry Interior Douglas-fir (IDF) forest. Uncut forest, two levels of partial cutting, and clearcut patches at the Opax Mountain site provide a gradient of canopy removal, while the systematic location of cutblock edges allows measurements of edge effects unconfounded by the changes in habitat type or topography often associated with operational cutblock locations. The shrew study at the Opax site also allows tests of two critical assumptions used in the *Biodiversity Guidebook* (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1995), the basis of biodiversity conservation in British Columbia forests:

1. Partially cut forests with less than 30% volume removal are equivalent to uncut forests, while partial cuts with greater removal are equivalent to clearcuts (the "30% rule").
2. Cutblocks affect adjacent forest for a distance of up to 200 m.

Both assumptions represent "educated guesses" that require testing in several forest types for a wide range of species that could be sensitive to forest management.

METHODS

Pitfall Sampling Technique

We trapped shrews and ground-dwelling arthropods using arrays of small pitfall traps (400 ml plastic cups with opening diameter of 9.5 cm) set flush with the ground surface. A 30 × 30 cm board was held 15 cm above the pitfall trap on three stakes to keep rain and debris out of the trap. To set the trap, we poured 100 ml of 66% propylene glycol, a nontoxic and nonvolatile liquid, into the cup. We collected the contents of the trap after 14 days. A trapping "session" consisted of two consecutive 14-day collections. In winter, 1.3 m tall chimneys of either plywood or PVC piping were held above the traps on raised cover boards. Winter traps were set through the chimneys and collected 2 months later without disturbing the snow cover.

Sampling Treatment Blocks

We used a triple-nested sampling design. Five individual pitfall cups were set in a 4 m radius circle to form a "trap circle." Five trap circles, 35 m apart in a cross pattern, formed a "set." Two sets of five circles each were located in each study block, with the sets at least 200 m apart (Figure 1). In winter, one chimney was used in the centre of each circle of five summer traps. Trapping sessions were conducted in the autumn of 1993 (pre-treatment); the summer of 1994 (post-treatment); spring, summer, and fall of 1995 and 1996; the spring of 1997; and in three post-treatment winters.

Sampling across Harvested Edges

An additional series of trap circles was used to sample across the edges of the 0.1-, 0.4-, and 1.6-ha cutblocks at the following distances (Figure 2):

- 0.1 ha: 15 m into the cutblock, at the forest edge, and 15 m into the forest;
- 0.4 ha: 15 and 30 m into the cutblock, at the edge, and 15 and 30 m into the forest;
- 1.6 ha: 15 , 30, and 60 m into the cutblock, at the edge, and 15, 30, and 60 m into the forest.

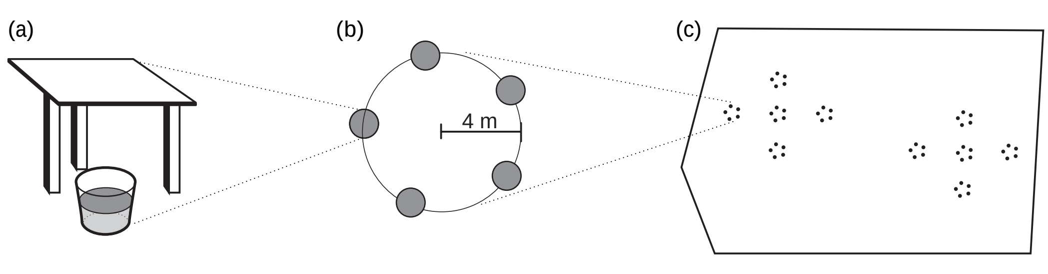

FIGURE 1 *Design of pitfall sampling and vegetation plots in treatment blocks at Opax Mountain: (a) individual pitfall cup; (b) trap circle of five cups; (c) two sets of five circles per block.*

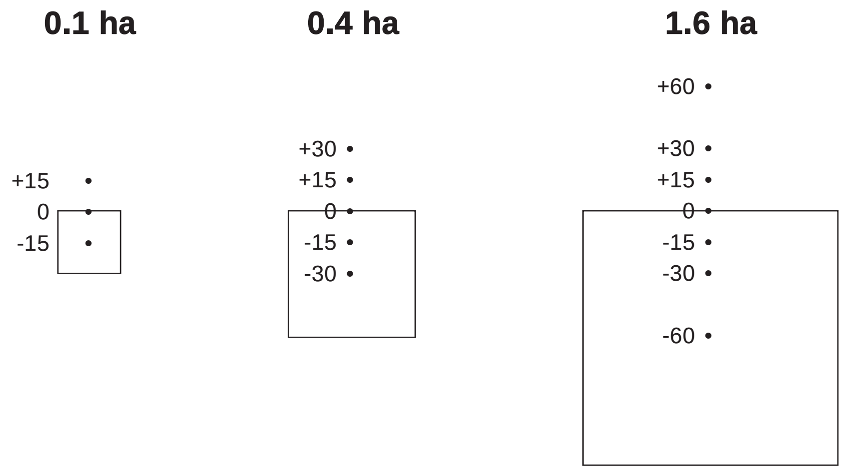

FIGURE 2 *Sampling across edges in the three sizes of patch cuts at Opax Mountain. A circle of five traps is located at each point (distance [m] into forest).*

These transects sampled only the north edges (south facing) of eight 0.1-ha openings, four 0.4-ha openings, and six 1.6-ha openings.

Identification of Specimens

Mammal species were identified on the basis of dental characteristics, except for old *Sorex monticolus* and *S. vagrans.* The worn incisors of these species required cranial measurements (Woodward 1994) and soft palate morphological characters for identification. Rodents were sexed and given a reproductive class based on external characteristics. Dissection was required to determine the sex and reproductive class of insectivores. Insectivores were also assigned to an "age" class based on tooth wear, from 1 (erupting teeth) to 3 (teeth completely worn down). All specimens were weighed.

Data Analysis

For most analyses, we combined the five cups within a circle and used the circle as the sample unit because different cups within a circle were obviously not independent; that is, an individual caught in one cup was not available to be caught in other cups in the circle. However, different circles were probably far enough apart to be sampling areas used by different individuals. Each trap circle was categorized into one of six harvest types:

- uncut contiguous forest (in control blocks);
- uncut leave strips (in patch-cut blocks);
- 20% partial cut;
- 50% partial cut;
- clearcut (within a patch cut); or
- edge of clearcut and uncut (within 5 m of the edge).

To summarize seasonal trends in numbers of each species in the different forest types, we standardized the data by calculating residual numbers after the seasonal mean was subtracted. This eliminated the annual and seasonal variability in overall numbers caught, and showed how each forest type differed from the overall seasonal mean. The nested sampling design allowed us to measure the relative importance of the variation in shrew numbers caused by harvest type, compared to three sources of natural spatial variation: differences between replicate blocks, between sets of traps within blocks, and between circles within sets. Variance partitioning based on nested analysis of variance (following Sit 1995) was used to determine the percentage of observed variance attributed to each of the four sources.

RESULTS

Total Numbers Caught

A total of 2 496 mammals were caught in the main treatment unit samples, and an additional 760 in the edge traps, of which 83 and 14, respectively, were caught in the winter sessions. The 2 413 mammals caught in the main traps, excluding those caught during the winters, represent an average of 1.59 captures per 100 trap-days, which is lower than many pitfall trapping studies report. This reflects our spring and autumn sampling when numbers were much lower than in the typical summer sampling season reported in other studies. It probably also demonstrates the relatively low densities of small mammals in dry-belt forest. In winter, the capture rate was 0.29 per 100 trap-days.

Red-backed voles were the most common species captured (40.1%), followed by *Sorex cinereus* (30.1%), *S. monticolus* (11.4%), *S. hoyi* (8.2%), meadow voles (4.7%), and *S. vagrans* (2.5%). The capture value for *S. hoyi* represents an exceptionally high relative abundance for this generally rare species. Red-backed voles were rarely captured in winter, while the relative abundances of the shrews were the same.

Annual and Seasonal Variability in Abundance and Maturity

Shrews at the Opax Mountain site showed large seasonal variation, with peak numbers in summer (August) and few captures in the spring or autumn (Figure 3). Red-backed voles were very rarely caught in spring, but were abundant in summer and autumn. Annual variation in abundance of the species was less pronounced, with 1.5- to 3-fold differences between the highest and lowest years at a particular season. Comparable annual variability and seasonal fluctuations have been reported in other studies (Henttonen et al. 1989; Sheftel 1989; Marcström et al. 1990).

The increase in summer abundance was attributed to the large increase in numbers of immature, young-of-the-year animals appearing at this time. Mature animals, presumably those born the previous summer, declined in abundance from spring to summer and were nearly absent by autumn. Seventy percent of the specimens in spring were mature, while by August only about 10% were mature. In winter samples, all animals were reproductively immature, but had worn teeth and appeared to be the young of the preceding summer. Captures of red-backed voles and other rodents were almost exclusively immature specimens, probably because adults avoided the traps or could escape from them.

The change from predominantly old, mature animals in spring to almost exclusively young animals in autumn, combined with the peak of abundance in August, suggests that spring samples represent overwintered adults, August samples represent primarily dispersing young-of-the-year, and autumn samples represent post-dispersal young.

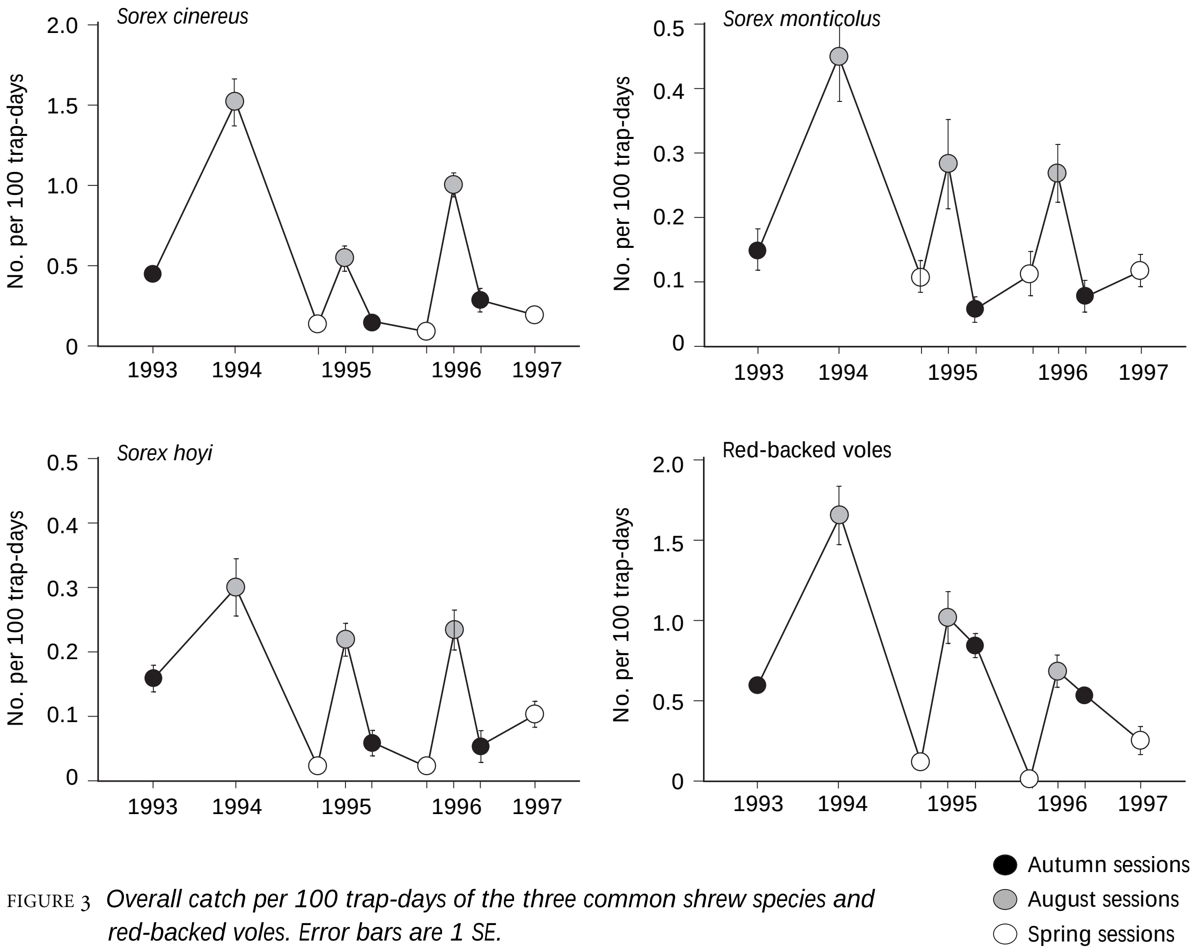

FIGURE 3 *Overall catch per 100 trap-days of the three common shrew species and red-backed voles. Error bars are 1 SE.*

Effects of Harvest Types

All main study species were found in all six harvest types, and no extreme differences existed between the harvest types (Figure 4). Statistically significant differences between harvest types were found only for *S. monticolus* in the spring and fall seasons (ANOVA: $F = 3.56$, $p = 0.025$, and $F = 2.74$, $p = 0.05$, respectively). The residual numbers (i.e., the percentage difference from the seasonal mean) suggest that some subtler effects of the harvest types were exerted on the different species.

***Sorex cinereus* (Figure 4a):** Trap circles in clearcuts and 50% partial cuts tended to have above-average numbers, while circles in leave strips and on edges had a lower abundance. Species abundance at the edges decreased from spring through fall, while numbers increased in adjacent clearcuts. Contiguous uncut forest consistently had a higher abundance than uncut leave strips.

***Sorex monticolus* (Figure 4b):** Trap circles in contiguous uncut forest, leave strips, and light partial cuts tended to have fewer than average captures, while edges, clearcuts, and 50% partial cuts tended to be above average. As with *S. cinereus*, abundance at the edges and in adjacent clearcuts showed opposite seasonal patterns, although the actual patterns differed for the two species.

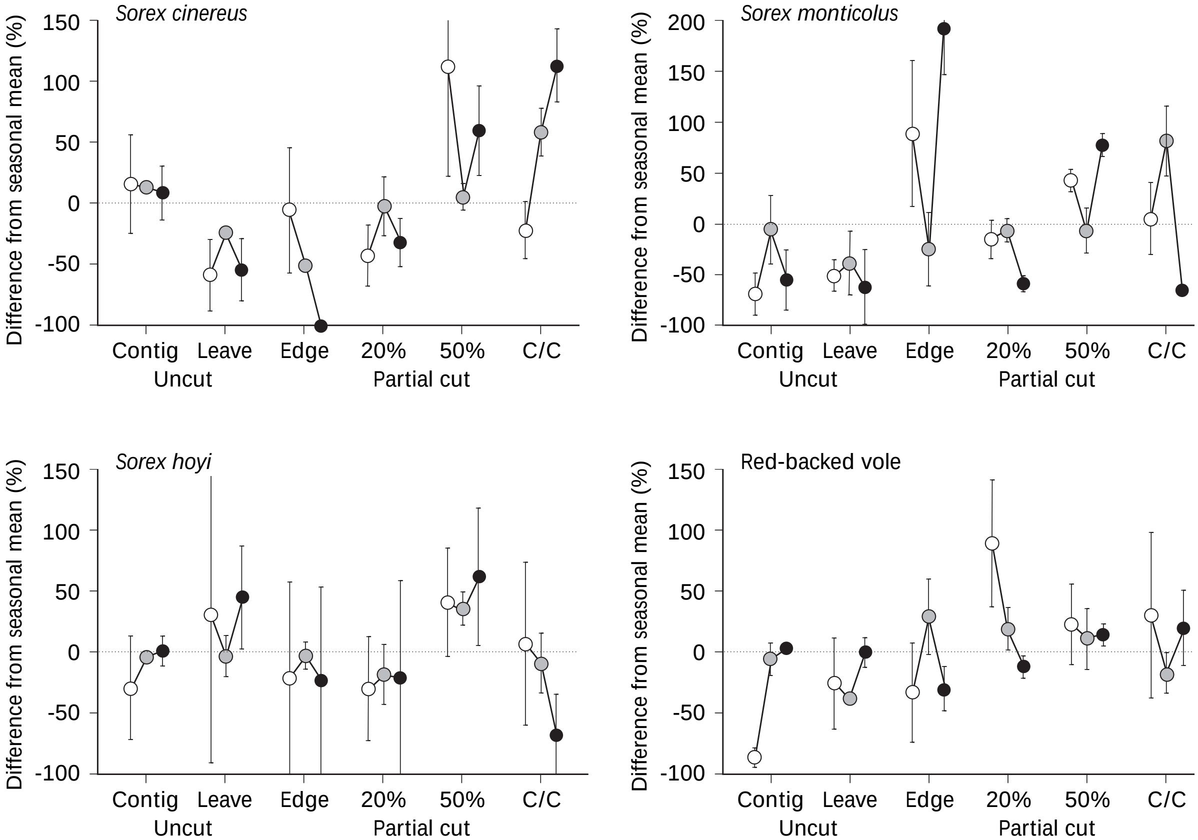

FIGURE 4 *Percentage difference from overall seasonal mean in six harvest types for three common shrew species and red-backed voles. Error bars are 1 SE.*

***Sorex hoyi* (Figure 4c):** This species was distributed very evenly between the different harvest types, except for possibly elevated numbers in the 50% removal partial cuts, and a reduction in clearcuts in autumn.

***Sorex vagrans*:** This species was not abundant in the study area, but seemed to be more common in the more disturbed circles, and least common in contiguous uncut forest and light partial cuts.

Red-backed voles (Figure 4d): This species showed an even distribution across the different harvest types, except in spring, when overall numbers were very low. Because only young juvenile red-backed voles were caught, these numbers may not reflect overall population differences for this species (see Klenner, these proceedings, p. 255).

In winter, *S. cinereus* was most commonly captured in clearcuts (0.29 per 100 trap-days), followed by contiguous uncut forest (0.23 per 100 trap-days), and 20% partial cuts (0.15 per 100 trap-days). *Sorex monticolus* was also most common in clearcuts in winter (0.16 per 100 trap-days), followed by 20% partial cuts (0.08 per 100 trap-days), and 50% partial cuts (0.07 per 100 trap-days). Clearcuts have the deepest snowpack in winter (Huggard et al., these proceedings, p. 186) and did not have the frozen ground that was observed in sites with a denser canopy and lower snowpack.

Differences between harvest types accounted for a substantial portion of overall variance in spring, summer, and autumn abundance only for *S. monticolus*, and for *S. cinereus* in spring (Table 1). For *S. hoyi* and red-backed voles, the different harvest types were actually more similar than would be expected given other sources of variation (negative variance components in Table 1). For all species, the difference between trap circles accounted for over 50% of variation in all seasons. This fine-scale variability emphasizes the importance of developing circle-level habitat relationships for these species; these relationships are being measured as part of this study, but are not reported here. Difference between sets was also an important source of variation in some cases, particularly for red-backed voles, which suggests that larger-scale habitat or topographic features influenced the abundance of these small mammals. Differences between replicate blocks were less often important sources of variation, and this mainly reflected the seasonal differences between the low-elevation Mud Lake replicates and the higher Opax Mountain blocks. These results indicate that the effects of the harvest types are not large compared to the natural spatial variability of small mammal populations.

Edge Effects

Edge effects were assessed in this study in two ways:

1. Using the main trap circles to compare cutblock edges with clearcut or uncut forest, and to compare leave strips (near edges) with contiguous uncut forest (> 100 m from cut edges).
2. Directly for distances up to 60 m with the edge transects.

Sorex hoyi, *S. vagrans*, and red-backed voles showed little difference between contiguous forest, leave strips, and edges, although considerable spatial variability existed for the two relatively uncommon shrew species. *Sorex monticolus* was much more abundant in some edge circles in spring and fall, but no difference in abundance existed between contiguous uncut

TABLE 1 *Relative importance of harvest effects and three sources of natural spatial variation in explaining variation in abundance of shrew and juvenile red-backed voles*

			% variation in abundance[a]		
Species	**Season**	**Harvest types**	**Blocks**	**Sets**	**Circles**
S. cinereus	Spring	16.4	-11.8	26.9	68.5
	Summer	7.2	10.5	-9.6	91.9
	Autumn	1.8	11.2	13.6	73.4
S. monticolus	Spring	23.0	-19.7	29.7	67.0
	Summer	16.6	3.0	-0.8	81.2
	Autumn	29.2	5.4	15.2	50.2
S. hoyi	Spring	-22.2	18.2	-9.8	113.9
	Summer	-1.6	-13.8	17.0	98.4
	Autumn	-16.0	-0.6	20.6	96.0
Red-backed vole	Spring	-11.5	15.9	29.9	65.7
	Summer	-25.8	45.6	26.8	53.4
	Autumn	-9.0	-7.2	60.2	56.1

a Negative percentage variations indicate that abundance was more similar than expected by chance.

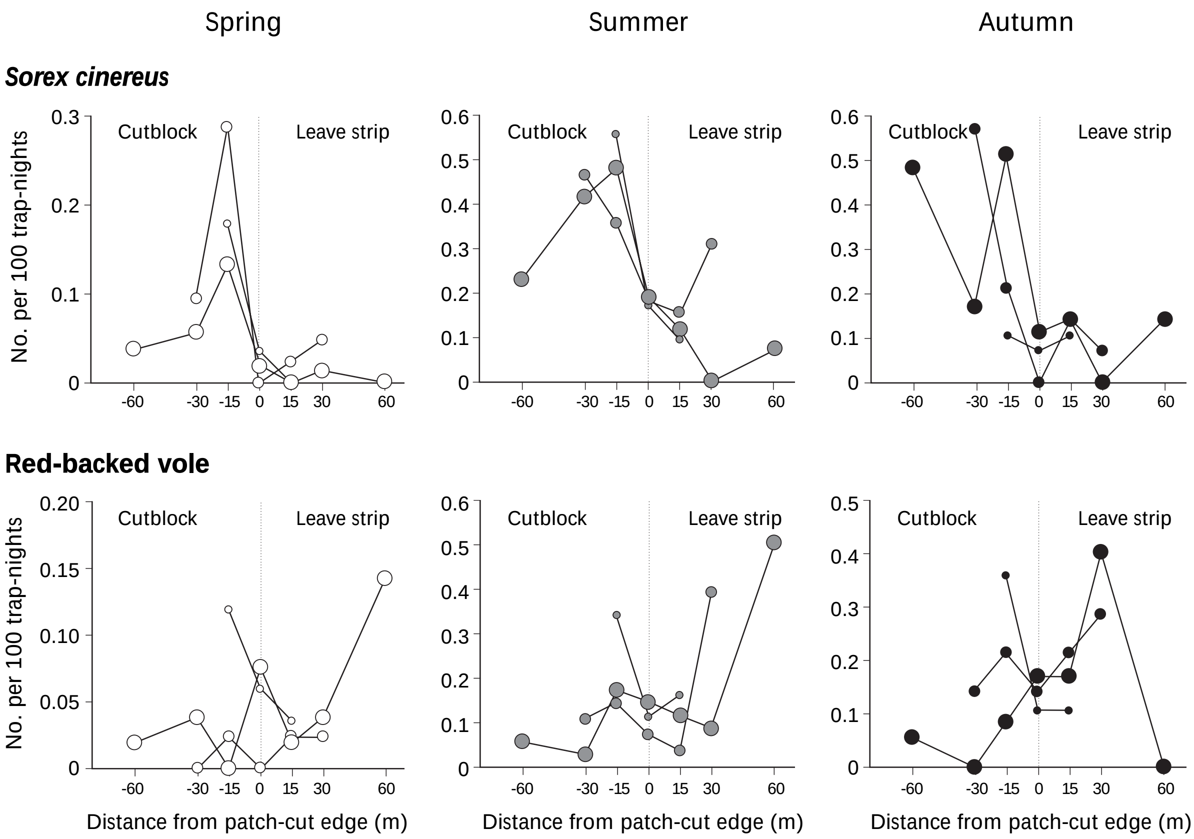

FIGURE 5 *Edge effects for* Sorex cinereus *and red-backed voles in spring, summer, and autumn.* S. monticolus *showed the same pattern as* S. cinereus.

forest and leave strips. This suggests that *S. monticolus* responded positively to edges, but this response did not extend far into the leave strips. *Sorex cinereus*, on the other hand, was more abundant in contiguous uncut forest than in leave strips or at edges, indicating the possibility of a more extensive negative edge effect. The species, however, responded positively to the clearcuts themselves.

The direct measurements of edge effects with the edge transects showed different patterns. In spring, both *S. cinereus* and *S. monticolus* were caught most frequently 15 m into the cutblock, and least commonly anywhere in the leave strips (Figure 5a). The same pattern continued in the summer sessions, except that the difference between 15 m and further into the cutblock was less pronounced. In autumn, with fewer animals caught, the pattern was more variable, but the centres of the larger cutblocks seemed to be favoured. Red-backed voles reached their highest abundance in the middle of the wider leave strips (Figure 5b) in spring (when they were rarely caught) and in summer (when they were common). In autumn, traps 30 m into the leave strips caught many voles, while the traps 60 m in caught none.

Study Limitations

Any study that relies on density to assess habitat quality risks misidentifying habitat with high densities, but poor productivity, as "good" habitat. This is particularly true when dealing with strongly territorial species or species with social systems that force many young animals into poorer habitat (Van Horne 1983). Because mature shrews are territorial (Hawes 1977), and we tend to catch mainly young animals, this could be an important limitation of this study. Samples taken in midsummer have the greatest chance of inaccurately reflecting habitat quality because, while our samples include mainly juvenile animals, territorial mature animals are still present in the population, and can potentially force the young into poorer habitats. As well, midsummer samples are likely to indicate local habitat quality poorly, if the animals caught are mainly dispersing juveniles that might have originated in a different habitat type. However, by sampling in spring (when we catch mainly overwintered adults) and in autumn (when the older generation has disappeared), we reduce the possibility of mistaking juvenile "sink" habitats as good-quality, high-density areas, and we sample animals that are less likely to be dispersing. Because we see the same general patterns of harvest effects in spring, summer, and fall, we are more confident that our density indices accurately represent habitat quality for shrews.

Harvest Effects Compared to Expectations

Based on accepted information about their general habitat needs, we expected that shrew numbers would be reduced in more disturbed low-elevation sites. However, this was not the case in our study; in fact, three of the four species appeared to benefit from more extreme harvest disturbances and the fourth was not affected. The only substantial negative effect of the harvesting was an apparent decrease in *Sorex cinereus* in leave strips when compared to contiguous uncut forest, but this species was common in patch cuts and heavier partial cuts. The lack of negative response to forest disturbance might be attributed to the naturally open, dry forest. This habitat requires shrews to use particular sites with suitable microclimates, even in uncut forest. Additionally, the more disturbed sites at Opax Mountain showed a rapid increase in forb and grass cover (Miege et al., these proceedings, p. 211); this increased cover could improve the ground habitat for shrews, and also provide increased food for the arthropod prey of shrews. A final possibility is that the four shrew species at Opax Mountain simply tolerate a wider range of microclimatic conditions than previously believed.

Effect of Light versus Heavy Partial Cutting: The "30% Rule"

The *Biodiversity Guidebook* recommends that partial cuts with less than 30% of the timber volume removed should be treated as uncut forest, while more heavily harvested partial cuts should be considered equivalent to clearcuts. This "30% rule" predicts that the partial cuts with 20% of volume removed at Opax Mountain should show similar patterns to uncut forest, while the partial cuts with 50% of volume removed should be similar to clearcuts. For *S. cinereus*, *S. monticolus*, and *S. vagrans*, the prediction seems valid, with only minor differences in seasonal patterns between uncut forest and 20% removal sites, and between 50% removal sites and clearcuts. *Sorex hoyi* differed somewhat with elevated numbers in the 50% removal sites when compared to other harvest types. Juvenile red-backed voles showed different seasonal patterns in the 20% removal sites and in the uncut forest. However, the "30% rule" appears to work well for this group of species.

Edge Effects

Somewhat conflicting results were produced by the different ways we used to measure edge effects. It was surprising to see any edge effects at all, as the differences between the harvest types themselves were not pronounced. However, the higher abundance of *S. cinereus* in contiguous uncut forest when compared to the leave strips or edge sites suggests that this common shrew species might be negatively affected by adjacent cutblocks, even though these cutblocks appear to be favourable habitat. Additionally, the edge transects showed a high abundance of this species and *S. monticolus* 15 m into the cutblocks, especially in spring and summer. All the edge transects were located across south-facing edges. Therefore, the observed edge effect results might reflect a warmer microclimate and earlier snowmelt (see Huggard et al., these proceedings, p. 186), which produces earlier growth of grass and forbs, and consequently more protective cover and insect prey near the cutblock edge. Sites right on the edge do not benefit from the southern exposure because forb and grass growth in the forest or within 5 m of the forest edge is very limited (Miege et al., these proceedings, p. 211), probably because of root competition with conifer trees. Edges with other aspects, including most of the edges sampled by the main traps, do not show early snowmelt, and often retain snow longer than any other sites. These other edges are therefore less likely to show the positive edge effect seen on the south-facing edge transects. Results from the edge transects implied that red-backed voles might be less common in the leave strips near the cutblock edge, although no evidence suggested that this species was more common in contiguous forest than in leave strips or edge sites. Overall, this study did not produce any strong evidence of negative edge effects. This is not surprising because these species are not negatively affected by harvesting itself, and live in an open, variable forest.

ACKNOWLEDGEMENTS

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REFERENCES

British Columbia Ministry of Forests and B.C. Ministry of Environment, Lands and Parks. 1995. Biodiversity guidebook. B.C. Forest Practices Code. Victoria, B.C.

Chen, J., J.F. Franklin, and T.A. Spies. 1993. Contrasting micro-climates around clearcut, edge, and interior old-growth Douglas-fir forest. Agricultural and Forest Meteorology 63: 219–37.

______. 1995. Growing-season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests. Ecological Applications 5:74–86.

Getz, L.L. 1961. Factors influencing the local distribution of shrews. American Midland Naturalist 65:67–88.

Hanski, I. 1987. Pine sawfly population dynamics: patterns, processes, problems. Oikos 50:327–35.

Harmon, M.E., J.F Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack Jr., and K.W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15:133–302.

Hawes, M.L. 1977. Home range, territoriality and ecological separation in sympatric shrews, *Sorex vagrans* and *Sorex obscurus*. Journal of Mammalogy 58:354–67.

Henttonen, H., V. Haukisalmi, A. Kaikusalo, E. Korpimäki, K. Norrdahl, and U.A.P. Skáren. 1989. Long-term population dynamics of the common shrew *Sorex araneus* in Finland. Annales Zoologica Fennici 26:349–56.

Kirkland, G.L. 1991. Competition and coexistence in shrews (Insectivora:Soricidae). Special Publication of the Museum of Southwestern Biology, University of New Mexico Press, Albuquerque, New Mexico. 1:15:22.

Korpimäki, E. and K. Norrdahl. 1989. Population biology of Eurasian shrews. Annales Zoologica Fennici 26:331–479.

Marcström, V., N. Högland, and C.J. Krebs. 1990. Periodic fluctuations in small mammals at Boda, Sweden from 1961 to 1988. Journal of Animal Ecology 59:753–61.

Nagorsen, D.W. 1996. Opossums, shrews and moles of British Columbia. Royal British Columbia Museum and University of British Columbia Press, Vancouver, B.C.

Sheftel, B.I. 1989. Long-term and seasonal dynamics of shrews in Central Siberia. Annales Zoologica Fennici 26:357–70.

Sit, V. 1995. Analyzing ANOVA designs. B.C. Ministry of Forests, Victoria, B.C. Biometrics Information Handbook No. 5.

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47:893–901.

Woodward, S.M. 1994. Identification of *Sorex monticolus* and *Sorex vagrans* from British Columbia. Northwest Science 68:277–83.

Wrigley, R.E., J.E. Dubois, and H.W.R. Copland. 1979. Habitat, abundance, and distribution of six species of shrews in Manitoba. Journal of Mammalogy 60:505–20.