WORKING PAPER

Sicamous Creek Silvicultural Systems Project: Workshop Proceedings

April 24–25, 1996

Kamloops, British Columbia, Canada

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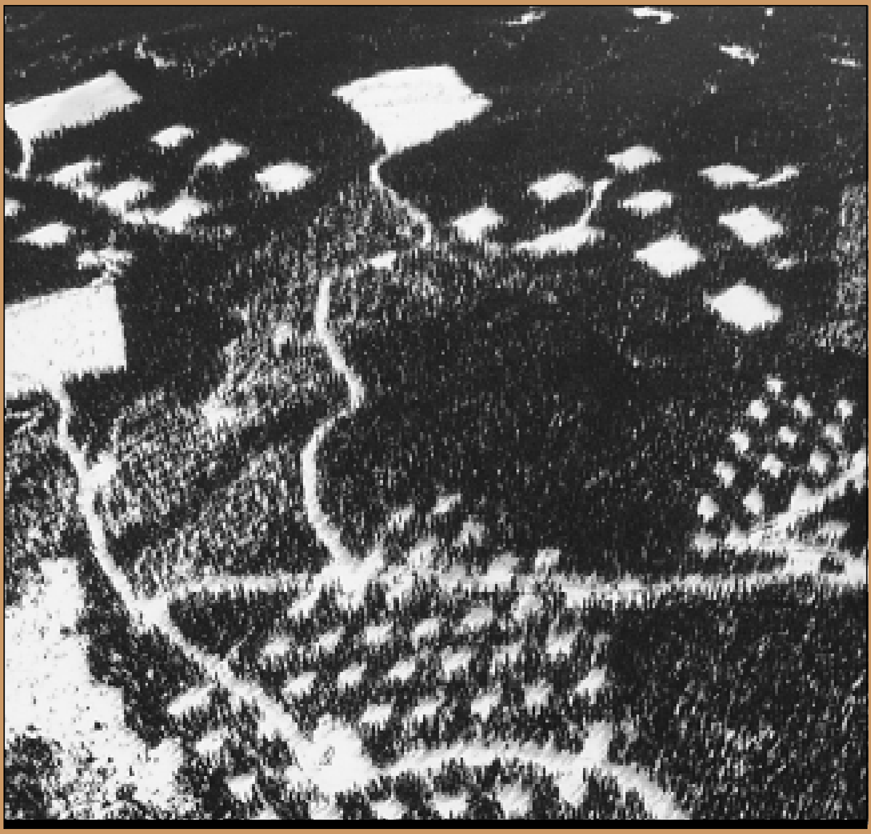

BRITISH COLUMBIA

Ministry of Forests Research Program

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Chris Hollstedt and Alan Vyse (editors)

British Columbia

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Sicamous Creek Silvicultural Systems Project: Workshop Proceedings

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Developing Silvicultural Systems for Sustainable Forestry in British Columbia

PASI PUTTONEN AND BRIAN MURPHY

ABSTRACT

In response to increasing public concerns over the practice of clearcutting, the British Columbia Ministry of Forests started a province-wide research and development program in 1990 to study alternatives to conventional clearcutting. The joint "Silvicultural Systems Program" involved several provincial and federal agencies, forest industry, universities, and communities, and now includes over 300 projects. The Sicamous Creek Silvicultural Systems Project and its many associated studies form part of this program.

The program's first objective is to improve our knowledge and under standing of various partial-cutting systems and their biological, operational, social, and economic implications. The second objective is to promote effective economic and appropriate application of alternatives to manage forests for multiple-use objectives. The program includes four main areas: (1) research, (2) operational trials, (3) social/economic studies, and (4) extension, training, and technology transfer. The four components of the program offer a unique opportunity to integrate new knowledge from ecosystems that have not been manipulated with partial-cuttings to a management for multiple-use objectives.

INTRODUCTION

In the last few years, public concern has increased over the traditional ways of managing forest resources in British Columbia. The public also demands a continued flow of benefits that result from the harvesting of timber in our forests. These benefits include high income, direct and indirect jobs, government revenues, and a myriad of forest products that are part of our everyday lives. As a response to increasing public concerns over the practice of clearcutting and its effects on biological diversity and visual aesthetics, the British Columbia Ministry of Forests with several collaborators started a province-wide research and development program (i.e., the Silvicultural Systems Program) in 1990 to study alternatives to conventional clearcutting. Alternative systems are often grouped under the common term "partial cutting," which means that some trees are left standing after the harvest. These include clearcuts with reserves; seed tree, shelterwood, and selection cuts; and intermediate commercial cuts such as thinnings. Silvicultural systems refer to forestry activities that are applied to a stand within a forest to meet specific resource management objectives over the life of the forest. Ideally, a silvicultural system must be environmentally sound, socially acceptable, economically feasible, and physically possible.

The program's objectives are:

- to improve our knowledge and understanding of various partial-cutting systems, and their biological, operational, social, and economic implications; and
- to promote effective, economic, and appropriate application of alternatives to manage forests for multiple-use objectives.

The shift from single resource use to multiple resource, ecologically based forest management conveys the intent of silvicultural systems research and development. The task has brought together and integrated the expertise of researchers and practitioners from many disciplines of forest management, including harvesting, regeneration, stand tending, forest health, growth and yield, biodiversity, fish and wildlife, recreation, and socio-economics. The program is a joint effort of several provincial and federal agencies, the forest industry, universities, and communities, and now involves almost 300 projects. The Sicamous Creek Silvicultural Systems Project and its many associated studies form part of this program.

DEVELOPING SILVICULTURAL SYSTEMS

The program to develop alternative silvicultural systems to manage forests for multiple-use objectives includes four main areas:

1. research;
2. operational trials;
3. social/economic studies; and
4. extension, training, and technology transfer.

The approach of multidisciplinary research supported by simultaneous demonstration, extension, training, and technology transfer is an attempt to gain new knowledge faster because the information needs about the use of partial-cutting systems in British Columbia are immediate and wide.

Research

This program area includes short- and long-term biological and ecological studies which examine the response of entire communities and species to alternative silvicultural systems. However, developing functional silvicultural systems is an ongoing process. The research will focus on increasing our understanding of ecosystem functions and thus help to maintain, or increase ecosystem structural integrity, while meeting various management requirements. The program's studies investigate the complexity of forest values through different scales of research at regional, landscape, stand, and process-oriented levels.

Landscape-level research Research at this level integrates stand- and process-oriented studies. This results in landscape-scale information that will be used in watershed planning which considers all resources. Landscape-level research has close links with issues of biological diversity, and will address questions concerning the effects of harvesting on water quality and quantity, fauna habitat needs, as well as the effect of silvicultural systems on future wood supply.

Stand-level research Research at this level addresses questions of operational forest management. The research will look at treatments that can be carried out, while still maintaining site productivity and biodiversity. It will focus on understanding stand structure and functions. It will also include studies of biophysical attributes of a particular ecosystem or site series and how those attributes affect stand dynamics and successional pathways after various disturbances.

Process-oriented research Research at this level provides an understanding of the responses of the plants and animals to micro- and meso-environmental changes brought about by various partial-cutting practices. Process-oriented research will provide the information needed to predict the outcome of a particular silvicultural treatment.

Operational Trials

These trials are generally short-term studies that test the practical implications of partial cutting or alternative silvicultural systems when applied in the field. The results will be used in conjunction with those from long-term studies to develop operational guidelines. The costs of different partial-cutting systems must be determined, including the costs of harvesting and silviculture. Analyses will include the costs and benefits of both timber and non-timber resource values. The initial assessment has concentrated on harvest planning, on testing equipment and their cost efficiencies in harvesting, and on evaluating the silvicultural system's ability to meet many resource values.

Socioeconomic Studies

These studies have an important role to play in assessing the costs and benefits of silvicultural systems and in developing a framework to compare alternative systems. The economic implications of using different silvicultural systems are being studied at the stand, landscape, and regional levels. The studies investigate the conservation of aesthetic values, site quality, and biodiversity at the lowest cost, with maximum timber production benefits, or maximum social benefits such as employment. Socioeconomic analysis also looks at the impacts of silvicultural systems on employment levels and government revenues.

Extension, Training, and Technology Transfer

These activities ensure effective communication of existing and new information to a wide range of audiences including foresters, forest workers, the public, researchers, and senior management of government and other agencies. This includes providing training courses; and producing scientific papers as well as technical documents and reports for foresters and workers, and information for the public audience.

These four program components offer a unique opportunity to integrate new knowledge from ecosystems that have not been manipulated with partial cuttings to develop new management practices for multiple-use objectives. The program has contributed to the development of the Forest Practices Code of British Columbia including its regulations and guidebooks.

The Sicamous Creek Silvicultural Systems Project: How the Project Came To Be and What It Aims To Accomplish

ALAN VYSE

ABSTRACT

The Sicamous Creek Silvicultural Systems Project was started in 1990 to address public and professional concerns about the effects of clearcut logging on high-elevation forests in the southern Interior of British Columbia. The project formed part of a wider B.C. Forest Service response to concerns about clearcutting forests throughout the province. The objective of the Sicamous Creek Project is to provide the forestry community with information on the ecology of high-elevation forests in the southern Interior and the probable reaction of the most common ecosystems to a wide range of disturbance. After a long period of planning by a team of scientists and foresters, the site was logged in the winter of 1994–95. Different levels of disturbance were created by cutting one-third of the forest in 30-ha experimental units using various opening sizes (single-tree selection, 0.1, 1, and 10 ha). Within these units, smaller areas were treated (e.g., by burning, complete scarification, and mounding) to create a range of soil disturbance. Many studies were conducted on the site by a team of scientists before and after treatment. Those studies are continuing.

INTRODUCTION

The Sicamous Creek Silvicultural Systems Project is designed to provide the forestry community and the public with information about the response of high-elevation forested ecosystems in the mountains of the southern Interior of British Columbia to various forest cutting and regeneration techniques. The project was originally part of a province-wide effort (B.C. Forest Service 1991) which was responding to general public concerns by encouraging the investigation of clearcutting and other cutting options. The Sicamous Project is now supported as part of a wider Forest Renewal BC research program.

Rationale for the Project

Support for the project was developed on the basis of the following arguments.

Subalpine forests are important to local communities Many communities in the southern Interior depend partly on subalpine forests for a supply of logs to keep their wood processing plants running and thus maintain a healthy local economy. The communities also depend on these forests for year-round supplies of clean water. The tree-clad mountains provide a scenic backdrop for a developing recreation industry, and habitat for a wide range of plants and animals. Not surprisingly, there is some publicly expressed concern about "what is going on up there," particularly in response to the common practice of clearcut logging 20- to 40-ha patches.

Nonsustainable harvests of subalpine forests are a concern Part of the community concern is based on fear that any form of timber extraction under such harsh conditions would lower the tree line either temporarily or permanently, and that current forestry operations are in some way "not sustainable." Because the subalpine forests have a visible limit (i.e., the tree line), they are thought to be at the "physiological edge" where low temperatures, high winds, and deep snowfalls combine to reduce biological activity above- and below-ground, and ultimately limit survival.

The past regeneration record in logged areas provides some support for this view. These forests are dominated by two species, subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) and Engelmann spruce (*Picea engelmanni* Parry ex Engelm.). Of the two, spruce is more highly favoured by the timber industry. Regeneration efforts have concentrated on this species with dismal results in many cases, especially at elevations above 1650 m in the southern Interior and in the areas with heaviest snowpacks (Mather 1987; Butt 1988; Farnden 1995). The lack of success may be due to the short growing season, rapid re-establishment of competing vegetation, poor spruce seed production, failure of planting stock, and damage to seedlings from snow and vegetation press (Coates et al. 1991). Recent changes in site preparation techniques and spruce nursery regimes and stock types have led to improved spruce planting success rates (Mather 1987, 1991; Butt and Vyse 1992). But concerns are still widespread among members of the public and professional foresters alike.

Subalpine forests are successfully managed elsewhere in North America Many records from other locations in North America demonstrate that silviculture systems, including clearcutting, can achieve regeneration success in high-elevation forests (e.g., Roe and DeJarnette 1965; Alexander 1977; Alexander and Edminster 1977; Alexander 1986). The climate and the vegetation and animal communities of subalpine forests of the British Columbia interior have many similarities with forests elsewhere in the North American Cordillera and the wider subarctic-subalpine forest region of the continent (Daubenmire 1978). There are some differences in forest stand structure, regeneration capability, and climate and climatic variability. For example, subalpine fir exist in higher proportions, particularly on northern aspects, and good seed crops in Engelmann spruce are produced less frequently. But none of these differences seem sufficient to argue that similar management successes would not be possible in British Columbia.

Information about the key effects of forest cutting is lacking The general concern about the loss of the subalpine forest in the longer term is buttressed by concerns about loss of resources and services that the existing forests supply in the short term. Loss of wildlife and wildlife habitat in old forests and changes in the amount and quality of water available to valley residents feature prominently in any discussions about the effect of clearcutting in high-elevation forests. Smith (1993) points out that very little information is available about the effects of silvicultural practices on the ecology of forest wildlife throughout North America, and that the information on watershed management in misunderstood.

Documented trials and studies of alternative cutting practices are lacking Very little information is available on the relative merits of alternative silvicultural practices at high elevation in British Columbia. Until recently, only one documented silvicultural systems trial had taken place in high-elevation forests in the province (Smith and Clark 1974; Mather 1991), but this has long been discontinued. Two trials have just begun in northern regions under the Silvicultural Systems program (e.g., Cariboo Forest Region 1992; Jull et. al. 1996). These trials will provide a useful source of ecological comparisons along with the long-established trials in northern low-elevation spruce-fir stands at Aleza Lake (Jull and DeLong 1996).

Project Objectives

The broad objective of the Sicamous Creek research project is to provide the forestry community with information on the ecology of high-elevation forests in the southern Interior. The project will study the responses of the most common ecosystems to a wide range of disturbance levels created by tree and stand removal and subsequent site preparation for regeneration. With this information, operational foresters should be able to test and then choose among alternative methods of logging and regenerating the common forest types. If the project is successful, a much wider range of silvicultural systems will be in use in the high-elevation forests of the southern Interior by the year 2000. Alternatives to clearcutting will be routinely prescribed with the confidence that wood supply targets can be met with minimal damage to forested ecosystems.

RESEARCH DESIGN AND METHODS

Project Approach

Our project design was based on the following propositions:

- Stand-level experiment: an experiment that examines the effect of different levels of disturbance through varying patch size, and therefore edge effect, has more chance of meeting diverse and changing management needs than one based on the application of some widely recognized "off the shelf" reproduction method or silvicultural system (see Smith [1986:331], Bradshaw [1992], and Smith [1993]).
- Spatial and temporal constraints: some management information needs can not be met by the limited spatial scale of a stand-level experiment (e.g., forecasting forest-level biomass production, or landscape-level

information on natural stand dynamics and wildlife diversity); resources must be directed to support associated studies in these areas. In addition, the conflict between the long time scale of any experiment and the urgent demand for operationally relevant information must be resolved by substituting space for time and devoting project resources to conduct retrospective studies across sites in the ESSF zone.

- Team approach: no single agency in British Columbia has the resources or staff to carry out such an ambitious ecosystem-based silvicultural systems project and the only possible route to success is to develop a co-operative network of researchers from various agencies.
- Extension: the best way to ensure management support of an inherently risky research venture is to provide information to the forestry community and interested public on project findings *and on existing findings* in a readily accessible form and on a regular basis.

Project Location

We selected the Engelmann Spruce – Subalpine Fir wet cold subzone (ESSFwc2) described by Lloyd et al. (1990) as the focus of our initial research efforts. This subzone is the largest of the seven ESSF subzones in the Kamloops Forest Region and is very similar to the colder, wetter ESSF subzones in the Nelson and Cariboo forest regions. Within this subzone, old-growth (age classes 8 and 9) Engelmann spruce – subalpine fir stands are the target of current logging operations. We therefore set out to find a site large enough to implement a range of treatments at an operational scale with replication. We determined that an area of 600–1000 ha would be required to carry out treatments at an operational scale and to avoid treatment-confounding effects.

A reconnaissance of high-elevation stands throughout the Kamloops Forest Region was conducted in the summer of 1991 by Dennis Lloyd and Bob Stathers. A site that best met all of the research requirements was located in the Salmon Arm Forest District, south of the north fork of Sicamous Creek and north of Mount Mara, near the town of Sicamous. The site is located in the ESSFwc2 subzone which grades into the ESSFvv subzone before alpine tundra is reached on Mount Mara and the adjacent summit (see Figure 1).

The stand is classified as B(S)841M6 with a small portion of SB841G6. The standing volume of live timber was estimated at 264 m^3/ha based on standard cruise procedures. Of this volume, approximately 35% was spruce and 65% subalpine fir. The cruise also reported that 30% of the standing trees were snags.

Experimental Treatments

Stand regeneration is a fundamental requirement of a successful silvicultural system. We therefore chose experimental treatments that would create a wide variety of conditions, from an undisturbed canopy with natural canopy gaps to areas with no overhead canopy which were at least 150 m from a forest edge. These conditions were to be created at an operational scale. Our aim was to provide opportunities to study logging practices, regeneration, and also to assess the effects on the spatial distribution of such ecosystem features as ground vegetation, snow accumulation and melt, and wildlife habitat structures.

Microsite disturbance was selected as the secondary treatment. Considerable experimental evidence suggests that some degree of forest

FIGURE 1 *Map of the ESSF zone in British Columbia and the general location of the Sicamous Creek Project.*

floor disturbance will significantly improve regeneration success (e.g., Smith and Clark 1960; Butt and Vyse 1992). Knapp and Smith (1982) found that mineral soil exposure reduced the mortality that results from drought stress, especially for Engelmann spruce germinants, which exhibit slower initial root growth than subalpine fir.

The five main treatments were (see Figure 2):

1. Control: No removal.
2. Single-tree selection: 33% of the volume was removed over a 30-ha area on the first pass by cutting every fifth tree using faller's choice (no marking) and marked skid trails. This treatment is the same as a single-tree selection cut, but with no attempt to "improve the stand." The original plan called for removal of 0.01-ha groups, but this was rejected by the loggers as impractical.
3. 0.1 ha: 33% of the volume was removed over a 30-ha area by cutting approximately sixty 0.1-ha openings, each about 31 m^2, with 60 m between centres and skid trails linking all groups.
4. 1 ha: 33% of the volume was removed over a 30-ha area by cutting nine groups of 1-ha, each about 100 m^2, with 200 m between centres and skid trails linking all groups.
5. 10 ha: 33% of the volume was removed over a 30-ha area by cutting one 10-ha opening approximately 330 m^2.

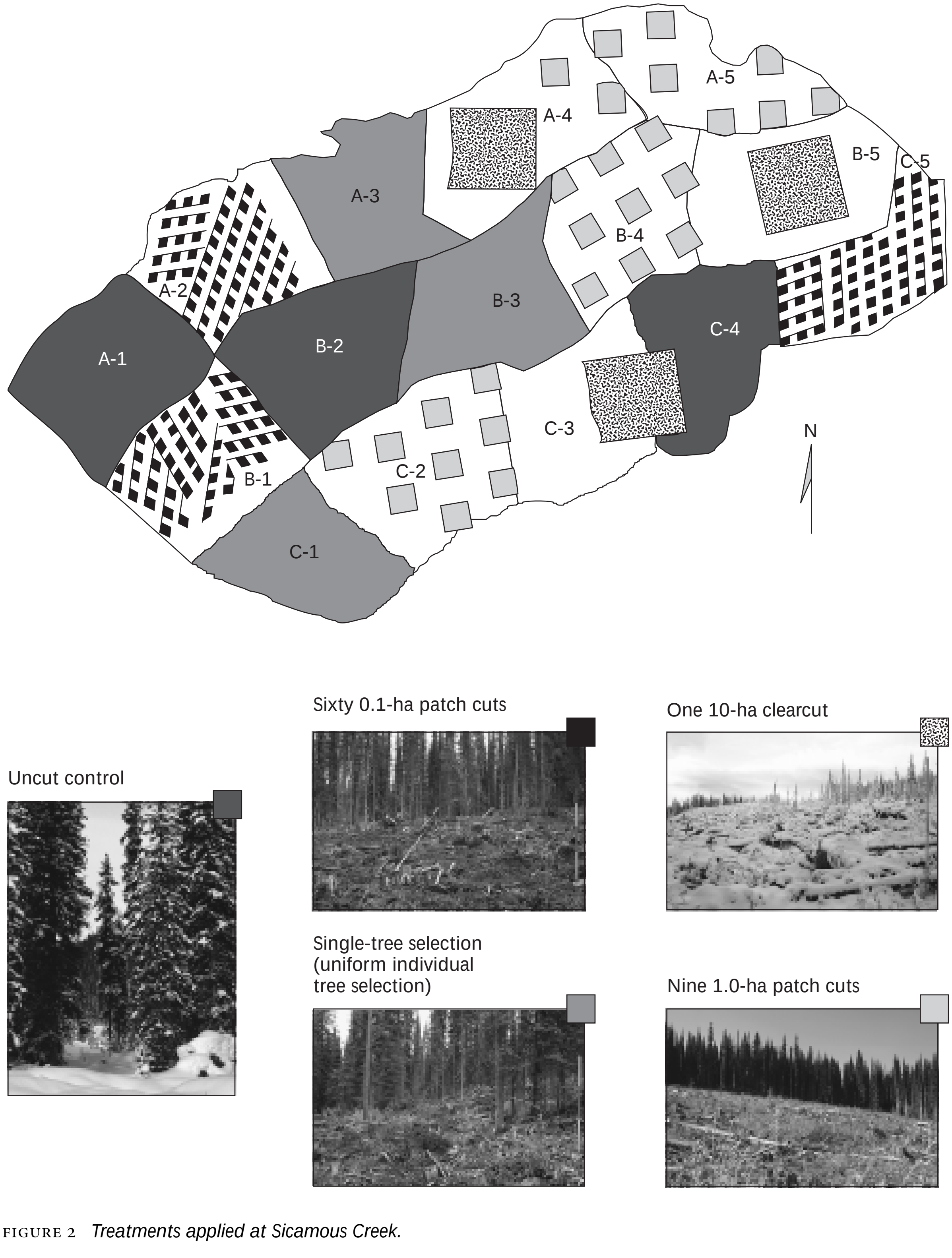

FIGURE 2 *Treatments applied at Sicamous Creek.*

All treatments were logged in winter to minimize soil disturbance. The largest opening size of 10 ha (320 × 320 m) was surrounded by a 100 m wide buffer strip of mature forest. This helps to ensure that treatments are independent. All treatment areas are therefore 520 × 520 m (about 27 ha), or approximately 30 ha with access roads included. The long-term silvicultural treatment assumes that about 33% of the volume within each treatment unit will be removed in the first pass. The remaining volume will be removed in two further entries. The remaining second pass volume will not be removed until:

- adequate regeneration is established in the first openings;
- concerns about water quality and quantity in Sicamous Creek are resolved; and
- experiments are completed in the initial openings.

Four microsite or soil disturbance treatments were applied to 0.1-ha plots in each opening size treatment. A mixture of Engelmann spruce and subalpine fir seedlings were planted in each. The soil disturbance treatments were:

- no site preparation;
- mechanical site preparation by scalping;
- mechanical site preparation by mounding; and
- burning of downed wood, vegetation, and organic soil horizon.

Three replicates of each experimental treatment were arranged in a randomized block design. A nested plot design will be used to assess the effect of the four canopy opening sizes and the 12 site preparation and regeneration strategies.

RESEARCH TEAM

The nucleus of the research team was established in 1991. Scientists from various specialties were invited to join Forest Service team members. The team has since grown to include those listed in Table 1. The studies supported by the project are also shown in Table 1. Membership is not restricted: additional research scientists are welcome to participate. However, base funding is limited and will be allocated to the highest scientific priorities once minimum administrative needs have been met. The team has met on an informal basis on many occasions to make basic decisions on project design and to identify project connections and gaps (ESSA 1995).

TABLE 1 *Research projects and investigators at Sicamous Creek*

Study	Title	Investigator(s)
1	Wind tunnel and field studies of wind regimes and windthrow in forest clearcuts	M.D. Novak, A.L. Orchnasky, T.A. Black
2	A hazard rating scheme for windthrow in high-elevation forests of the Kamloops Forest Region	R.J. Stathers
3	Effects of alternative silviculture practices on the song bird communities in ESSF forests	T.E. Dickinson, N.J. Flood, E. Leupin
4 and 5	Effects of silvicultural systems and habitat and landscape relationships: mice and voles, small carnivores, cavity-nesting birds, ground-dwelling arthropods, spruce grouse, insectivores Snow depth, density, and duration Persistence rates of snags: implications for worker safety and habitat supply	W. Klenner, D. Huggard
6	Effects of silvicultural systems on carbon cycling and storage by ESSF forests	D. Sachs
7	Effects of silvicultural systems on soil productivity in the ESSF: impacts on organic matter and nutrient dynamics	G. Hope, C. Prescott
8	Effects of silvicultural systems on soil microarthropods in an ESSF forest	H. Nadel, S. Berch, G. Hope
9 and 10	Effects of different silvicultural practices on ectomycorrhizal diversity in ESSF forests	D. Durall, M. Jones
11	Effects of silvicultural systems on nutrient budgets in ESSF forests	M. Feller
12	Effects of silvicultural systems on fine-root biomass and nutrient dynamics in ESSF forests	G. Hunt
13	Effects of clearcut size on microclimate	R. Adams
14	Effects of opening size on snow accumulation and melt	R. Winkler
15	Patterns of regeneration in mature subalpine forests in ESSFwc2 and ESSFvv Planting microsite selection for silvicultural systems at high elevations	R. Parish
16	Growth processes of interior spruce and subalpine fir	D. Simpson
17 and 18	Spatial and temporal response of vegetation to silvicultural practices in the ESSF	D. Lloyd, K. Yearsley, A. Arsenault, T. Goward, M. Ryan
19	Regeneration of Engelmann spruce	M. Feller
20	Advanced regeneration in ESSF and IDF forests	P. Puttonen
21	Effects of single-tree selection cutting on growth and yield of ESSF forests	A. Vyse
22	Effects of opening size on diseases of ESSF forests	H. Merler
23	Spatial and temporal analysis of attack and impact of *Dryocoetes* in natural and managed ESSF forests	L. MacLauchlan, L. Harder
24	Relationships between small mammals and downed wood	W. Klenner, T. Sullivan, V. Craig
25	Coarse woody debris dynamics in wet and dry ESSF forests	C. Prescott

TABLE 1 *Continued*

Study	Title	Investigator(s)
26	Pathology of conifer seed and seedlings in natural and disturbed forest floor seedbeds in ESSF stands	B. Van der Kamp
27	Ecosystem mapping	A. Inselberg
28	Logging costs	J. Mitchell
29	Effects of timber harvesting and road construction on stream invertebrates in the Sicamous Creek watershed	B. Heise
30	Modelling the role of alternative silvicultural systems on landscape-level biodiversity	W. Klenner
31	Seedfall of Engelmann spruce and subalpine fir in the ESSF at Sicamous Creek	A. Vyse
32	Evaluating the effects of site preparation treatments on vegetation, soils, and biodiversity	D. Lloyd, G. Hope, W. Klenner
33	Effects of silvicultural treatments on the soil food web and nitrogen dynamics in ESSF study sites	G. Hope, S. Simard, K. Johnson
34	Ecological niche for seedling establishment in high-elevation forests in wet subzones of the ESSF	P. Puttonen, P. Brang
35	Structure and condition of high-elevation forests 40 years after harvesting	P. Puttonen, A. Vyse

ACCOMPLISHMENTS AND ACKNOWLEDGEMENTS

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, permanent camp and trail facilities constructed, pre-logging studies carried out with results presented to the forestry community, the logging carried out and monitored, and post-logging impact studies begun. The early results of this work are reported in the remainder of this volume.

None of this would have been possible without a remarkable display of co-operation among project investigators; nor would any of the this co-operative spirit have led to anything more than "castles in the air" without efforts well beyond normal duties on the part of many other forestry colleagues. The forest licensee in the area, Riverside Forest Products, the Salmon Arm Forest District, and the Kamloops Forest Region all worked together to ensure that the project could go ahead as planned and within a reasonable time frame.

Much remains to be done.

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Harvesting Costs and Logistics at Sicamous Creek

JANET MITCHELL

ABSTRACT

In 1994, the Kamloops Forest Region of the British Columbia Ministry of Forests, in co-operation with the Lumby Division of Riverside Forest Products Ltd., carried out a study to examine the effects of five canopy opening sizes (including uncut control, single-tree selection, 0.1-ha group selection, 1.0-ha small patch cut, and 10-ha clearcut) on the regeneration of a high-elevation forest near Sicamous, B.C. The Forest Engineering Research Institute of Canada (FERIC) monitored the harvesting phase. Productivity and cost were affected more by the harvesting system than by the opening size. Mechanical harvesting systems were more productive than conventional systems and the 10-ha opening had the highest productivity of the mechanical systems. The overall harvesting cost was highest for the conventional harvesting systems on the single-tree selection units.

Detailed results from this study can be found in the following technical note:

Mitchell, J.L. 1996. Trial of alternative silvicultural systems in southern British Columbia: summary of harvesting operations. Forest Engineering Research Institute of Canada, Vancouver, B.C. FERIC Technical Note No. TN-240.

Age and Size Structure of the Forest at Sicamous Creek

ROBERTA PARISH

ABSTRACT

Dendrochronological techniques and stem analysis are used to develop a history of the subalpine fir – Engelmann spruce forest at Sicamous Creek. The forest originated in the 1650s. A forest of dense patches with moderately growing dominants and a suppressed subcanopy probably existed until the mid-1800s when a large number of trees were removed from the canopy. A period of recruitment and release of suppressed trees occurred from 1850 to 1900. Small-scale disturbances by insects (most likely western balsam bark beetle) and diseases (e.g., *Armillaria* and Indian paint fungus), which are all prevalent in the current stand, are likely responsible for this pattern of release. Growth rates for the post-1850 cohort were considerably higher than previous growth rates, possibly in response to the open canopy and the increased temperatures associated with the end of the "Little Ice Age."

INTRODUCTION

Current theory emphasizes the role of disturbance as often the dominant influence on stand development pattern. Stands become more susceptible to disturbance as they age (White 1979). The resultant forest is then a mosaic of different patch sizes and age structures which reflects the history of the size and frequency of past disturbances within the stand. At some point in the renewal cycle a steady state may be achieved as the forest as a whole regenerates in patches.

Fire has historically been the most important form of natural disturbance to provide regeneration sites. In wet subalpine forests, however, fires are generally considered infrequent, but of a severe and stand-replacing nature (Peet 1988; Baker and Veblen 1990). Fire intervals of 300–400 years are typical of Yellowstone National Park (Romme and Knight 1982). However, studies of fire history in the Canadian Rockies and southeast British Columbia indicate that large fires were frequent (every 60–130 years), generally stand replacing, and that most trees came from the post-fire cohort (Johnson and Fryer 1989; Johnson et al. 1990; Johnson and Larsen 1991). Jull (1983) found that post-disturbance recruitment could take 30–50 years at high elevations, which means that fire-initiated stands are not necessarily very even-aged.

In forests where no or only infrequent, intense, large-scale disturbances occur, small-scale disturbances such as insect attack, disease, and blowdown play an important role in creating patchiness and spatial heterogeneity within the forest landscape (e.g., in many tropical rain forests: Brokaw 1985, Denslow 1987; in some deciduous forests of eastern North America: Runkle 1990, Runkle and Yetter 1987; in very wet conifer forests of western North America: Stewart 1986, Spies and Franklin 1989, Lertzman and Krebs 1991, Lertzman 1992; and in temperate rain forests of the southern hemisphere: Stewart et al. 1991).

Engelmann spruce (*Picea engelmannii* Parry) and subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) are the major tree species in the subalpine forests of the Sicamous area. In general, Engelmann spruce is longer lived than subalpine fir, although it is susceptible to windthrow in stands with poorly drained or shallow soils (Alexander 1986). Engelmann spruce is also susceptible to the spruce beetle (*Dendroctonus rufipennis*), which attacks mature trees. Usually, populations of this insect build to epidemic levels only after extensive windthrow (Alexander and Shepperd 1990; Safranyik et al. 1990). Spruce is also attacked by some of the root, butt, and stem fungi, which also affect subalpine fir (e.g., *Armillaria mellea*, *Haematostereum sanguinolentum*, or *Phellinus pini*).

Subalpine fir is susceptible to western balsam bark beetle (*Dryocoetes confusus*), as well as various root and butt rots (e.g., *A. mellea*, *Polysporus tomentosus*, and *Coniophora puteana*) and stem rots (e.g., *Echinodontium tinctorium*, *H. sanguinolentum*, *P. pini*, and *Amylostereum chailletii*) (Alexander et al. 1990). The high mortality of subalpine fir is offset by its high fecundity which enables it to maintain its abundance in the stand (Veblen 1986).

This paper describes the age and size structure of the forest at Sicamous and uses height and radial growth patterns of Engelmann spruce and subalpine fir to develop a stand history.

METHODS

In 1994, six 1-ha cutblocks were chosen with fairly uniform aspects and vegetation and away from the patches of saturated soils near the toe of the slope. Within each block, a 50 × 50 m-plot was located. The distance and bearing from the centre of the plot of all live and dead trees taller than 1.3 m were recorded. The trees were tagged and species and condition recorded. The diameter of those greater than 4.0 cm at breast height was measured. After logging in the summer of 1995, discs were cut from the base of each tagged tree.

A group of trees within each plot was selected for stem analysis. Because of the large amount of rot in subalpine fir, preference was given to groups with a high percentage of Engelmann spruce to maximize the potential for sound stems. The trees were felled in 1994 and discs removed at height intervals of 0 cm, 30 cm, 70 cm, 1.3 m, 2 m, and every subsequent 2 m to the top of the tree. Total height, height to lowest live branch, diameter, and bark thickness were recorded in the field.

The discs were planed or sanded and annual ring widths along one radius measured to the nearest 0.01 mm with the Measu-Chron device at Pacific Forestry Centre. Two radii were measured on a subsample of discs and the computer program COFECHA (Holmes 1983) was run to detect measurement errors.

Discs from trees over 15-cm dbh were selected for chronology development. Ring-width chronologies for both species were produced from cross-dated discs by the ARSTAN program (Cook and Holmes 1984). The COFECHA program was also used to cross-date the discs of standing dead trees against the master chronology to determine the year of death.

RESULTS AND INTERPRETATION

Stand Characteristics

Engelmann spruce accounts for 10% and subalpine fir for 90% of the stems at Sicamous. Figures 1a and 1b show the diameter distribution of each tree species in 10-cm classes. Class 0 contains all the trees under 5 cm diameter at breast height and accounts for about 20% of both species. Spruce was well represented in all size classes, although only 6%

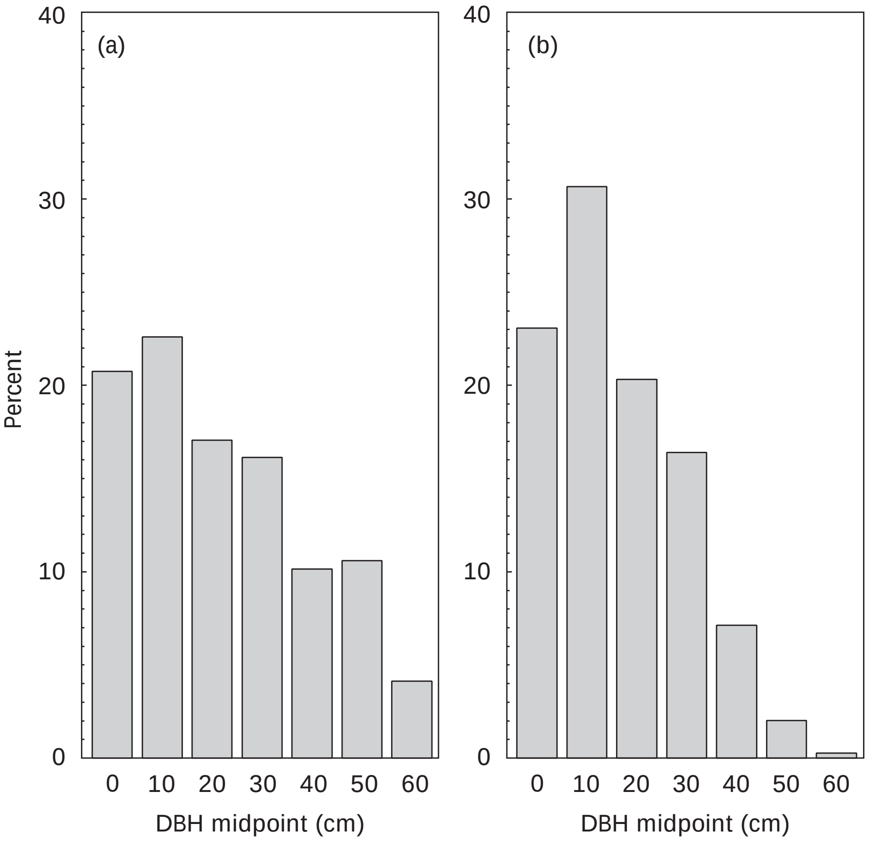

FIGURE 1 *Diameter classes of major tree species at Sicamous Creek: (a) Engelmann spruce, and (b) subalpine fir.*

were in the over 55-cm class. This, however, was a considerably higher percentage than subalpine fir for which only 2.3% were over 45 cm diameter and 0.3% over 55 cm.

The size structure of the spruce and subalpine fir populations at Sicamous did not coincide with the age structure (Figure 2a and 2b). For both species, the youngest age classes were not the most abundant. The majority of trees were 100–150 years old (55% of both spruce and subalpine fir). Spruce showed a bimodal age distribution with a second, smaller peak (16%; *n* = 21) of trees over 300 years old. No spruce were in the 150–200 age class. Subalpine fir over 150 years of age constituted 22% of the population and only 3% were over 300 years old (*n* = 27). The larger size of Engelmann spruce when compared to the subalpine fir did not necessarily mean it was older, unlike in subalpine forests of the southern Rockies (Oosting and Reed 1952; Veblen 1986; Veblen et al. 1991).

The current spruce population appears to originate from two major recruitment periods, one through the late 1600s and the second from 1850 to 1900. The empty age class in the early 1800s does not necessarily mean that no spruce germinated in this interval, but it is evident none survived to reach the canopy. The recruitment of subalpine fir has occurred

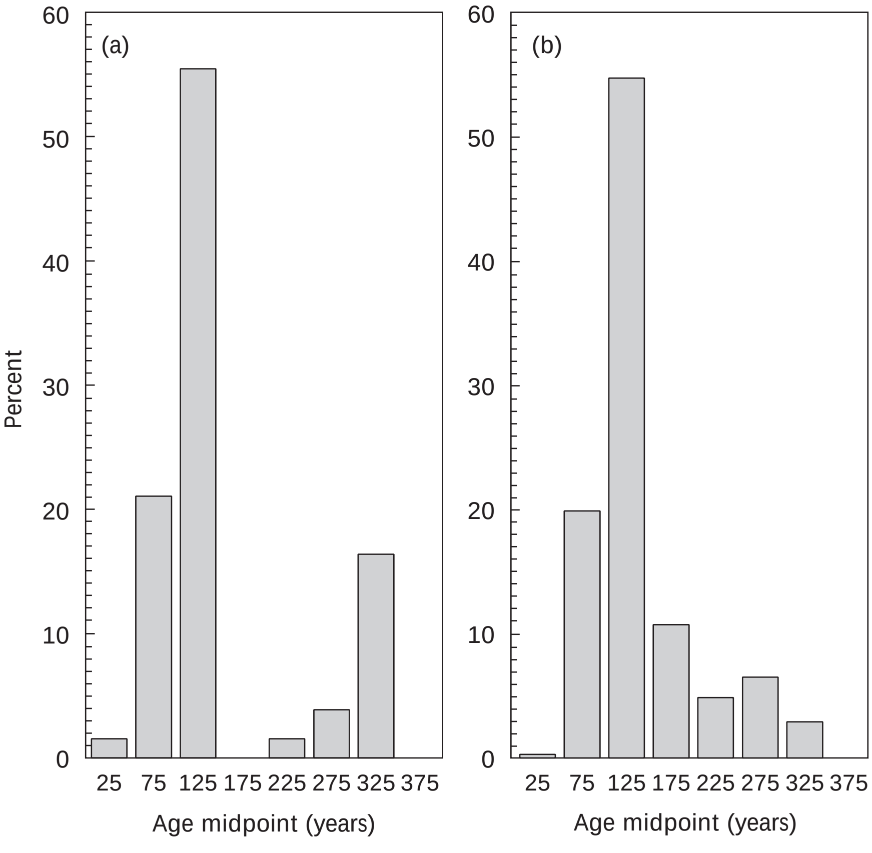

FIGURE 2 *Age class frequency of major tree species at Sicamous Creek: (a) Engelmann spruce, and (b) subalpine fir.*

throughout the past 300 years, although a major recruitment period also occurred around 1850–1900.

No strong relationships exist between age and diameter at breast height (Figures 3a and 3b). Whereas the largest spruce were always the oldest, this was not the case for subalpine fir. Linear, exponential, logarithmic, and power function regressions were fitted to the diameter at breast height and age data for each species. A linear regression gave the best estimate for spruce ($r^2 = 0.58$) and the power function for subalpine fir ($r^2 = 0.32$). Stewart (1986) noted a similar inconsistent relationship between age and size in coastal montane species, the result of periods of release from suppressed growth.

The mean density of stems for the six plots was 1526 stems per hectare of which 720 were greater than 15-cm dbh, the minimum size to reach the canopy. Density varied widely among the plots. The least densely stocked had 736 stems per hectare of which 376 were greater than 15-cm dbh. The most densely stocked had 1884 stems per hectare of which 800 were greater than 15-cm dbh. Twenty percent of the stand were dead trees, which were almost exclusively subalpine fir. In the sample of 2290 mapped trees, only two were dead spruce.

Figure 4 shows the condition classes of the standing dead subalpine fir. The majority of dead trees were in class 3, with loose bark and branches reduced to stubs. The mean age at death of subalpine fir was remarkably consistent at 150–160 years old for all condition classes (Figure 5), which probably reflects the predominant age within the stand. Class 4 trees were younger than the other classes, but this may be an error from the sample size ($n = 3$). Because of the high amount of internal rot, measurable basal discs were rarely recovered from class 4 trees and never from snags.

Ring measurements from dead trees were compared to a master chronology for subalpine fir (discussed below) to estimate the time since death (Figure 6). On average, dead trees retained their needles for four years, but this ranged from one to nine years. Although it is possible that

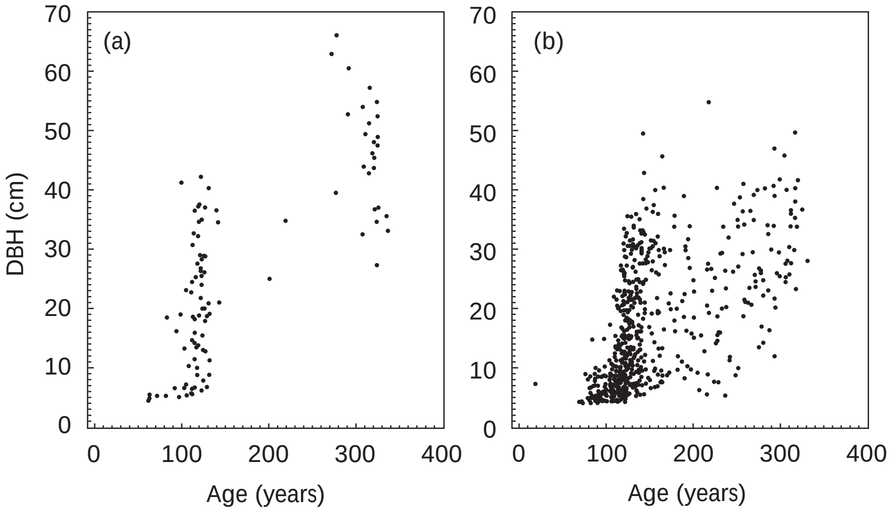

FIGURE 3 *Relationship of age to diameter at breast height for: (a) Engelmann spruce, and (b) subalpine fir.*

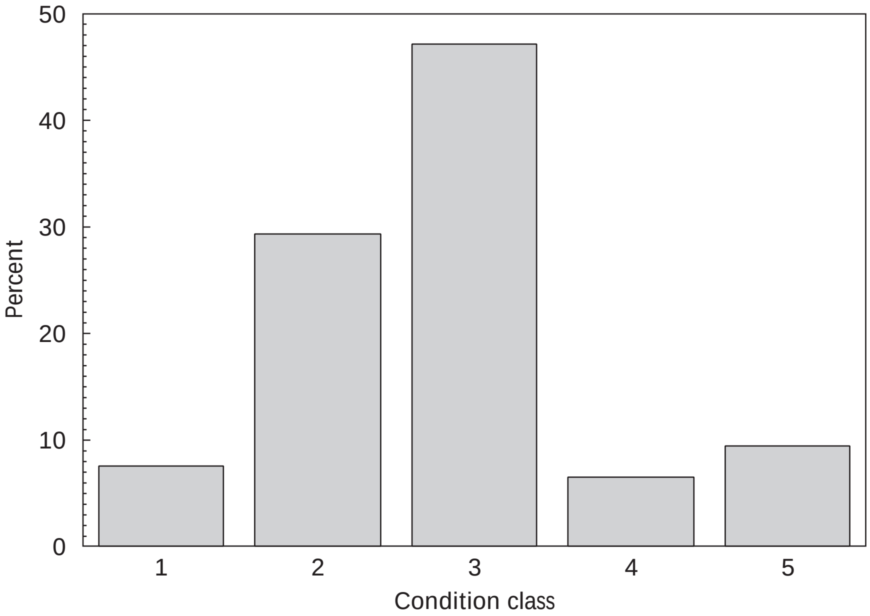

FIGURE 4 *Condition classes of dead subalpine fir (class 1 = trees with intact bark and retained needles; class 2 = trees with intact bark and retained branches; class 3 = trees with loose bark and retained branches; class 4 = trees with no bark and branch stubs; and class 5 = visibly decaying, snags over 3 m tall).*

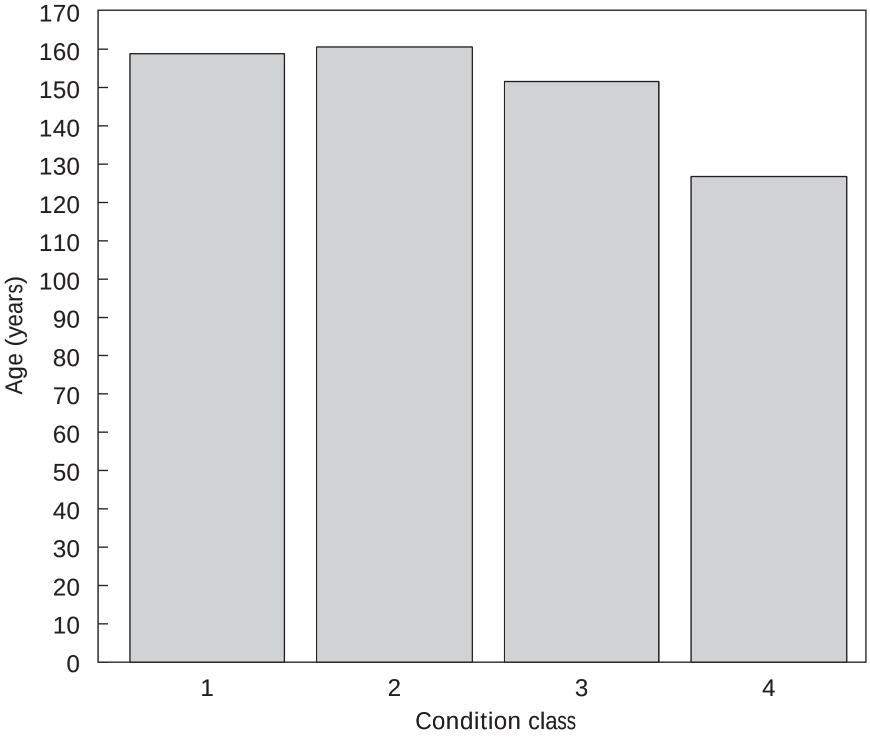

FIGURE 5 *Age of dead subalpine fir in each condition class (for a description of condition classes, see Figure 4).*

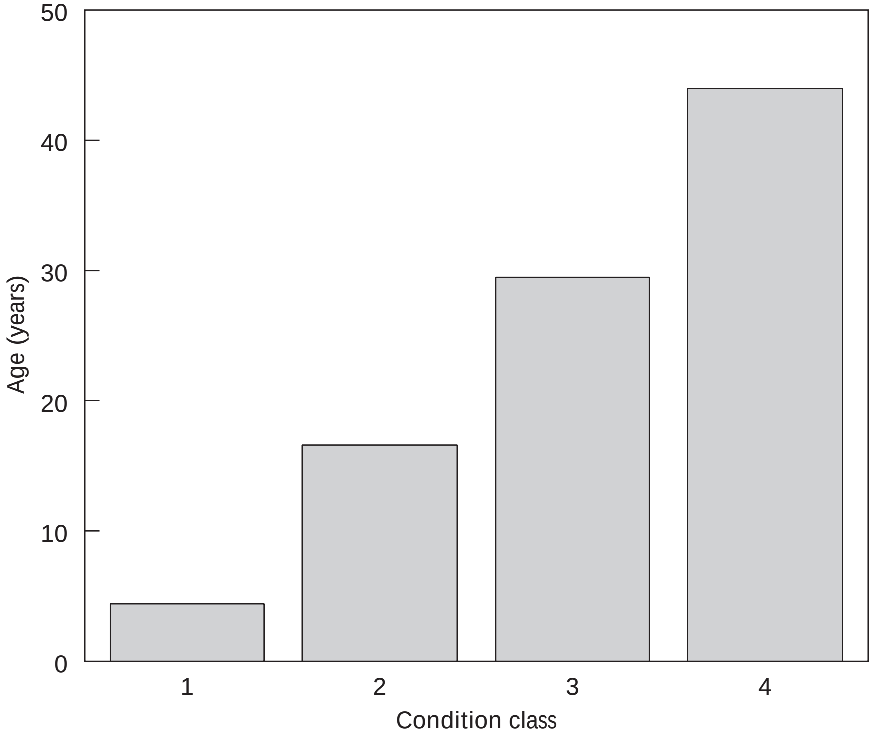

FIGURE 6 *Time since death of subalpine fir in each condition class (for a description of condition classes, see Figure 4).*

nine years is an overestimate of time since death, because the trees may not have put on any basal increment during the last years of their lives, subalpine fir retaining needles for up to nine years have been monitored in other parts of the province (T. Ebata, B.C. Forest Service, pers. comm., 1996). Some of the class 3 stems were still standing after 60 years.

Dendrochronology

Trees growing in areas limited by climate show greater variability in ring width than those in more favourable conditions. The cold, wet climate of the subalpine provides an environment with sufficient variability to plot growth trends.

The COFECHA computer program was used to cross-date the ring-width series. Strong correlations existed among samples dating back to the late 1800s, but the correlations became gradually weaker with older material. Radii from the same tree showed strong correlations, but cross-correlation among trees in the period before 1800 was not observed. Poor cross-correlation is seen in trees not affected by major climatic factors (cf., Fritts 1976), such as subcanopy individuals whose growth is dominated by neighbouring trees. Because of this very low inter-tree correlation, ring widths before 1800 were deleted from the record.

The resulting short-term chronologies were based on 48 ring-width series from 40 spruce trees and on 66 ring-width series from 51 subalpine firs (Table 1). The analysis indicated an overall series correlation of 0.527 for Engelmann spruce and 0.522 for subalpine fir (Table 1). Both series had high autocorrelation values which suggest that growth in any given

TABLE 1 *Dendrochronologic characteristics of the ring-width chronologies at Sicamous Creek*

	Engelmann spruce	**Subalpine fir**
Number of trees	40	51
Number of radii	48	66
Interval (years)	201	183
Years with prominent narrow rings	1814, 1815, 1826, 1836, 1838, 1854, 1867, 1869, 1870, 1894, 1916, 1927, 1929, 1951, 1954, 1955, 1956, 1971, 1972, 1982, 1991	1812, 1819, 1832, 1838, 1861, 1864, 1879, 1880, 1892, 1893, 1894, 1899, 1916, 1917, 1921, 1931, 1932, 1951, 1954, 1956, 1971, 1972, 1974, 1980, 1983, 1989
Years with prominent wide rings	1802, 1803, 1804, 1811, 1818, 1825, 1834, 1848, 1859, 1860, 1925, 1936, 1940, 1944, 1958, 1965, 1966, 1987	1813, 1814, 1822, 1825, 1837, 1839, 1877, 1888, 1903, 1905, 1906, 1908, 1926, 1938, 1940, 1941, 1942, 1944, 1958, 1960, 1969, 1970, 1976, 1977, 1987
Series correlation	0.527	0.522
Standard deviation	0.660	0.660
Autocorrelation	0.818	0.856

year is strongly influenced by growth in the preceding years. This is typical of other studies on the two species (e.g., Parker and Henoch 1971; Colenutt and Luckman 1991).

Tree growth on the site was very slow in the period before 1850–1860 (e.g., Figure 7). Both spruce and subalpine fir showed evidence of release from around 1848 to 1900. A sample of 245 ring-width series from trees of both species that were over 150 years old and over 10 cm at breast height were examined to identify periods of release. Release was defined as a 200% increase in growth over a 10-year period. Half of the trees examined (123) had released between 1840 and 1900 (Figure 8). The majority of those releases were between 1864 to 1874.

A small number (8) released from 1927 to 1937. Standing dead trees about 60 years old support the view that a period of death and release occurred in the late 1920s and early 1930s.

The computer program ARSTAN was used to produce the chronologies shown for Engelmann spruce (Figure 9) and subalpine fir (Figure 10). Both chronologies have similar overall growth trends. This includes periods of enhanced growth in the late 1930s to around 1950 when growth dropped off sharply, although 1958 was an exceptionally good year for both species. Reduced growth continued through the 1960s and 1970s for Engelmann spruce and only recovered in 1987. Subalpine fir showed enhanced growth from around 1870 to 1916, although a number of poor years occurred during the 1890s (Table 1). Although growth was moderately better in subalpine fir than spruce during the 1960s and 1970s, it had a very similar pattern of good and poor growth years (Table 1). Prominent narrow rings in 1938 and 1971 were found by Colenutt and Luckman (1991) in Engelmann spruce and subalpine fir in Alberta, indicating that the climatic influence in those years must have been very widespread.

Stem Growth

A total of 65 trees were felled and sectioned for analysis: 41 subalpine fir and 23 Engelmann spruce. Thirty-five of these trees were over 15-cm dbh, the minimum size to reach the canopy (16 subalpine fir and 19 spruce). The range in age was 95–325 years old.

Diameter at breast height was a good predictor of total height. Linear, exponential, logarithmic, and power function regressions were fitted to

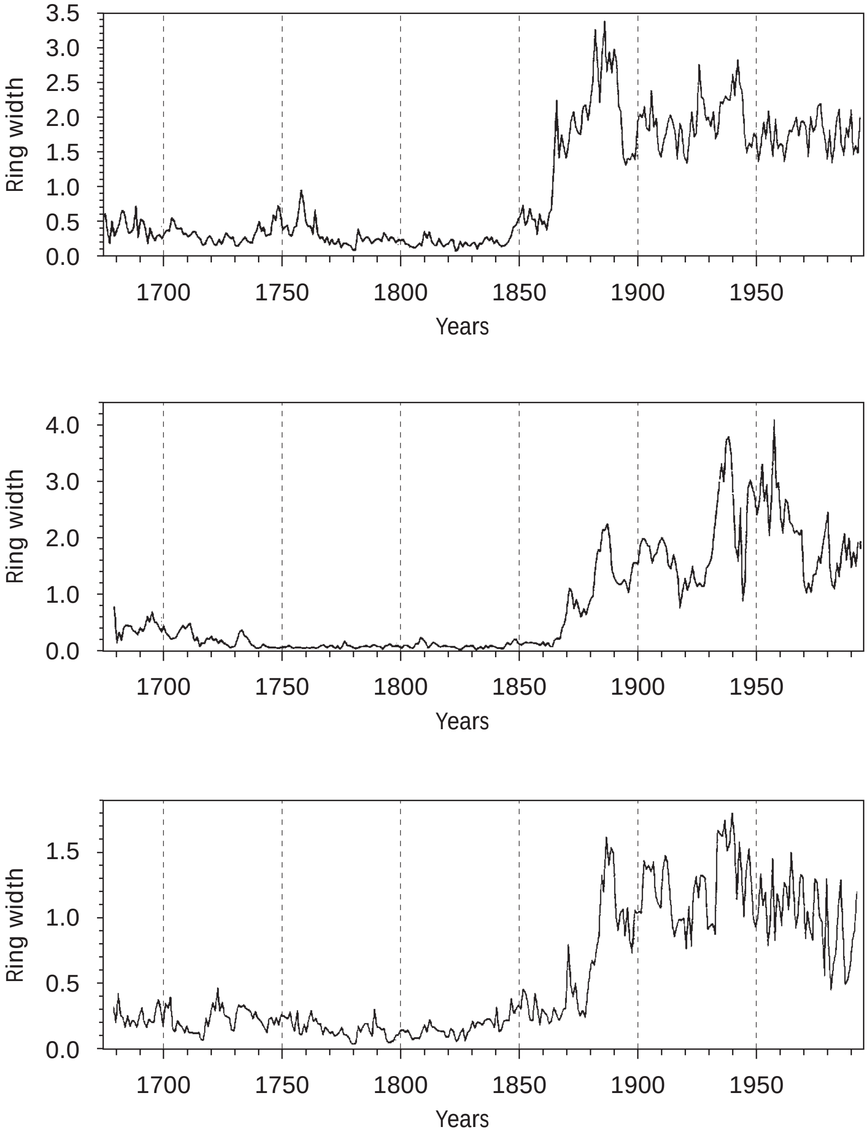

FIGURE 7 *Examples of suppressed trees released after 1860.*

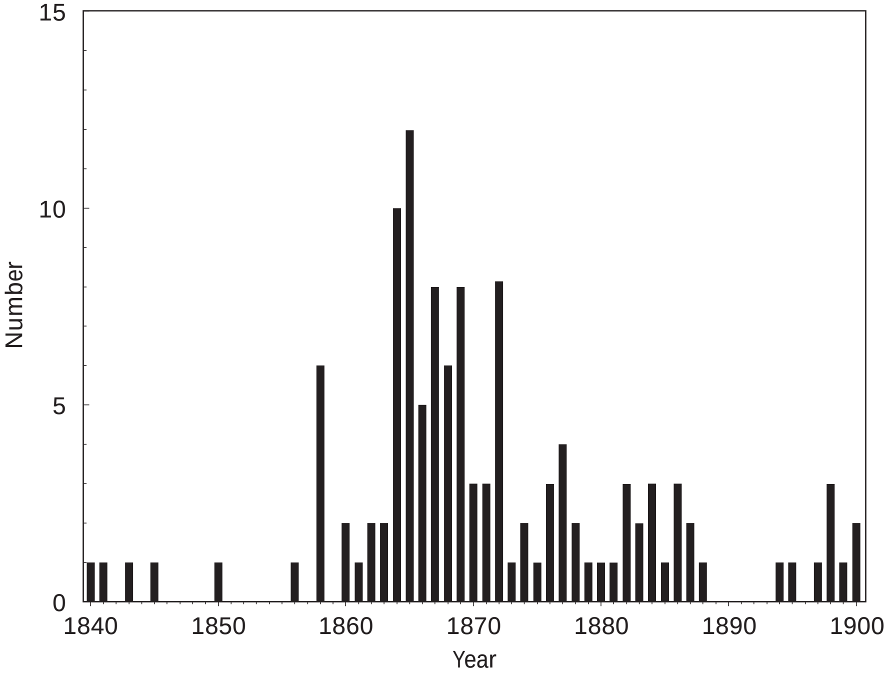

FIGURE 8 *Number of trees released in the late 1800s.*

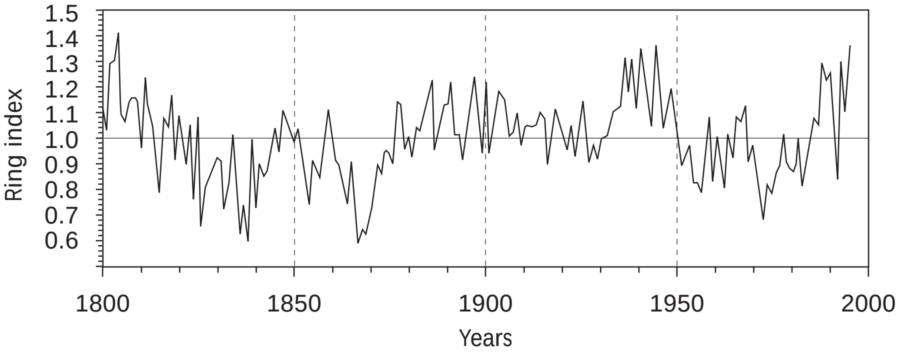

FIGURE 9 *Engelmann spruce ring-width chronology produced with ARSTAN.*

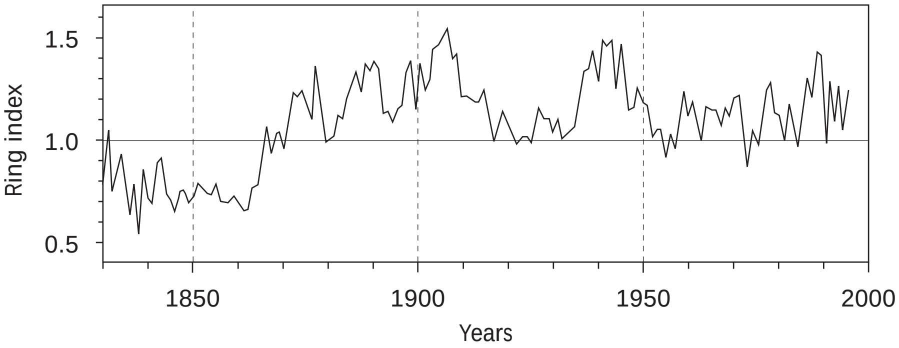

FIGURE 10 *Subalpine fir ring-width chronology produced with ARSTAN.*

the diameter at breast height and height data for each species. The power function provided the best fit for subalpine fir height (r^2 = 0.90) and the linear or power function for spruce (r^2 = 0.86).

Both Engelmann spruce and subalpine fir showed a wide range of growth rates and patterns (Figures 11a and 11b). Some Engelmann spruce grew very quickly and by 130 years had reach similar heights to trees over 250 years old. Diameter growth patterns showed close parallels to height growth (Figures 12a and 12b), with some spruce reaching diameters over 45 cm by 130–140 years of age.

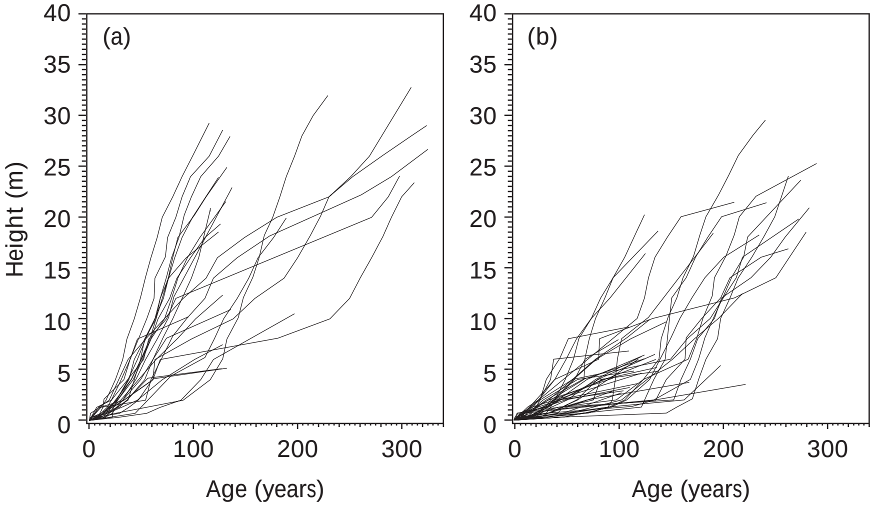

FIGURE 11 *Growth rates for major tree species at Sicamous Creek: (a) Engelmann spruce, and (b) subalpine fir.*

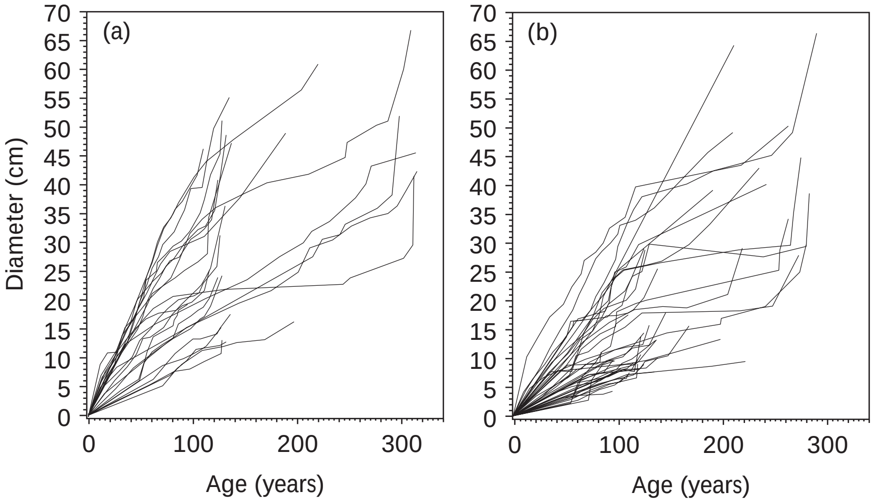

FIGURE 12 *Diameter growth rates for major tree species at Sicamous Creek: (a) Engelmann spruce, and (b) subalpine fir.*

Trees were separated into two broad cohorts: those over 200 years old representing the oldest trees on the site and those under 150 years old, most of which germinated after the release period in the 1860s and 1870s. Two patterns of growth emerged in the old trees (Figure 13). In one pattern, trees grew steadily almost from germination, while the other pattern showed prolonged suppression (100–160 years) followed by a rapid growth phase, which could eventually exceed the growth of the "early starters."

The post-1860 cohort also demonstrated two patterns: one showing rapid growth and the other suppressed growth (Figure 14). It is likely that, given the opportunity, trees in this second group could release in a similar way to the suppressed trees shown in Figure 13.

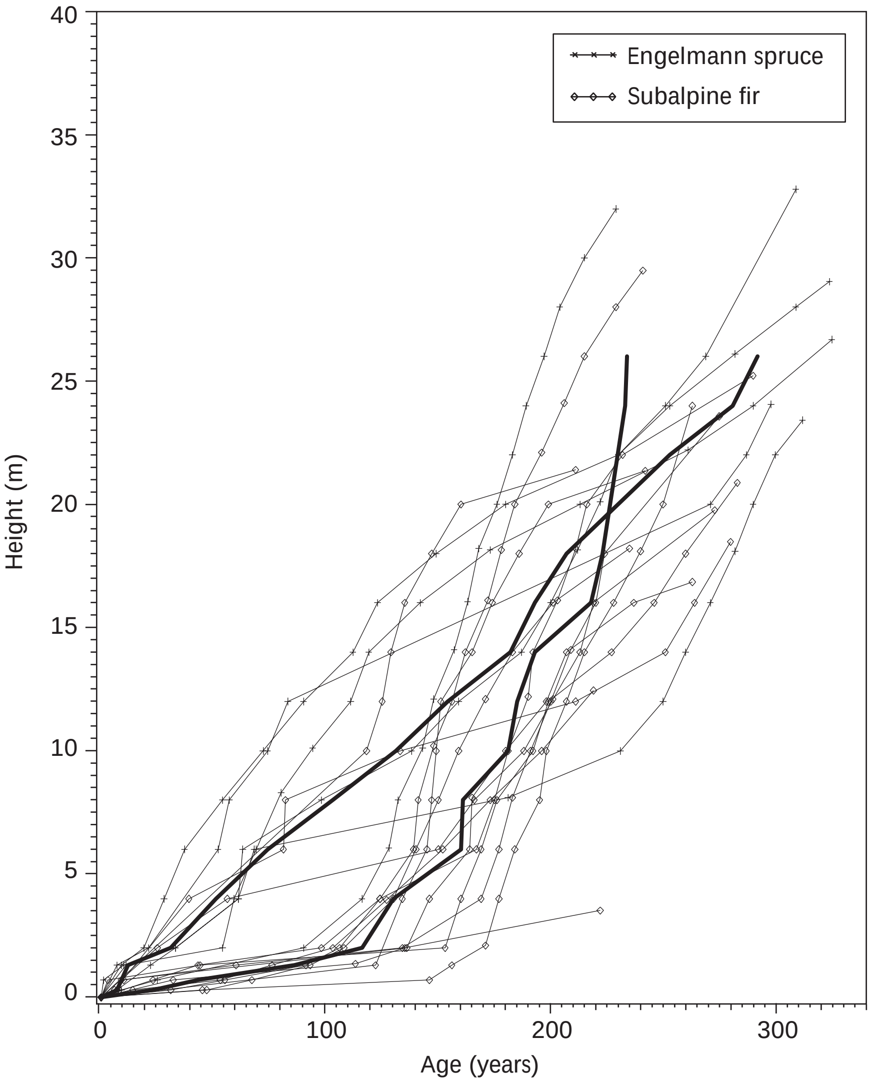

FIGURE 13 *Height growth of trees over 200 years old (heavy line is the midpoint of each growth pattern).*

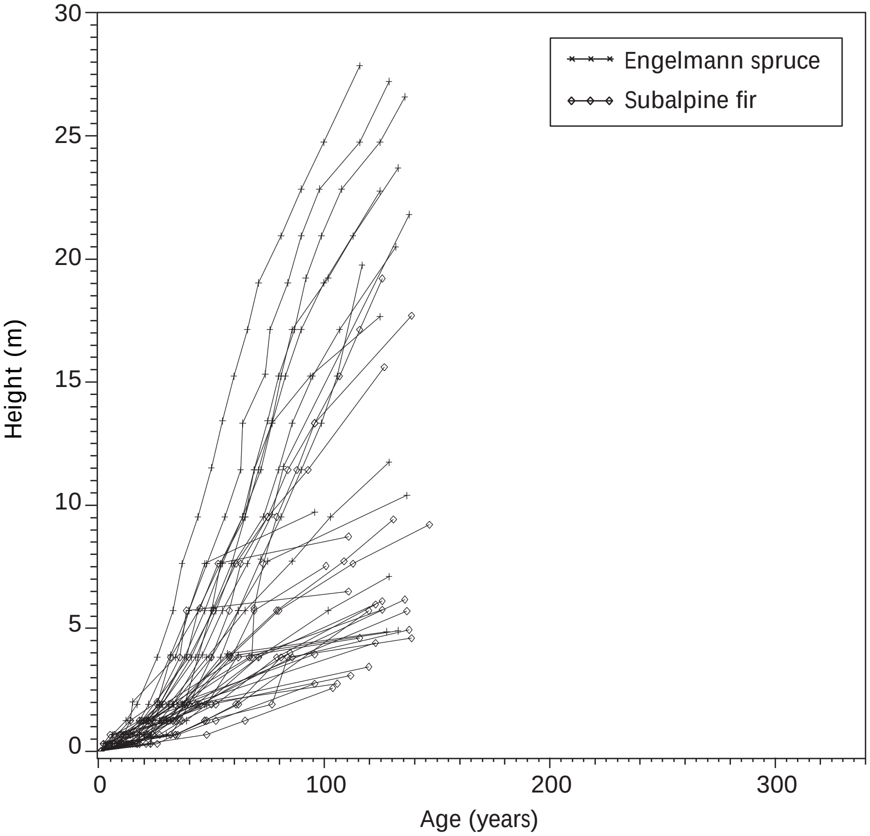

FIGURE 14 *Height growth of trees less than 150 years old.*

STAND HISTORY

Based on the information collected, a hypothesis of stand history at Sicamous Creek can be proposed. The oldest trees were 337 years old, far short of the potential 500–600 year life span of Engelmann spruce (Oosting and Reed 1952; Alexander and Shepperd 1990). This suggests that a widespread disturbance, probably fire, removed all previously standing trees and that the stand probably originated in the 1650s.

Trees dating from the late 1600s and early 1700s grew more slowly than trees from the late 1800s. Two factors could account for this. The climate from 1650 to around 1860 could have been less amenable to growth. A wealth of corroborative evidence suggests temperatures were cooler in this period; for example, Crowley (1996) reports that climate during the "Little Ice Age" (approximately 1450–1890) was 0.5–1.0°c lower than today. At subalpine elevations in Washington, Graumlich and Brubaker (1986) found mean annual temperatures between 1590 and 1900 were about 1.0°c lower that those in the 20th century. A short period from 1650 to 1690 exhibited temperatures comparable to the present.

Another reason for the slow growth is suppression from competing trees. If trees established densely after the initiating disturbance, many would be suppressed, especially those originating 50 or more years after the disturbance. Because there are areas in the forest without stumps or other traces of old trees, this regeneration must have occurred in dense patches rather than uniformly throughout the forest.

A forest of dense patches with moderately growing dominant trees and a suppressed subcanopy probably existed until the mid-1800s when a large number of trees were removed from the canopy. What caused the death of these trees? The timing of release over two decades, plus a lack of evidence of scarring on the surviving trees, suggests that fire was not a cause. Blowdown from wind would most likely have destroyed a single large tract or eliminated dominant trees in wet patches, but this is not likely to have recurred over two decades.

The pattern of release suggests small-scale biotic disturbances by insects and diseases which typically kill individuals or groups of trees. Subalpine fir is especially susceptible to western balsam bark beetle, *Armillaria* root disease, and Indian paint fungus (*E. tinctorium*) which are all prevalent throughout the present-day stand (H. Merler, B.C. Forest Service, pers. comm., 1996). An outbreak of balsam bark beetle is the most likely candidate for the death of canopy subalpine fir through 1864 to 1874, and possibly into the 1880s. Balsam bark beetle or one of the stem or root diseases could be responsible for deaths leading to the low background level of release in the sample.

By the mid- to late 1800s, climate was warming in the northwest. Wahl and Lawson (1970) indicated that summer temperatures during the 1850s and 1860s were warmer than those in 1931–1960, while winter temperatures were cooler. Growth of Engelmann spruce and subalpine fir is positively correlated with summer temperatures, especially June and July (Colenutt and Luckman 1991). The combination of warm summers and newly opened canopy could have led to enhanced growth in newly germinated and released trees.

Growth continued at this enhanced rate for several decades with some reduction after 1916. The warmer summer temperatures in the 1930s (cf., Fritts et al. 1979) could also enhance growth and this was probably coupled with an increase in canopy openness from the death of subalpine fir.

Because of repeated small-scale disturbances, the trees in this forest are constantly renewing. The forest may be about 350 years old, but the vast majority of the trees are under 150 years. The ability of both Engelmann spruce and subalpine fir to release after disturbance has implications for silviculture. Trees suppressed for over 150 years are capable of rapid recovery and growth. Silviculture regimes that preserve advanced regeneration would have an advantage in reforesting this site.

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Containment and Concentration of Western Balsam Bark Beetle in Patch- and Partial-Cut Silvicultural Systems

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ABSTRACT

Attack by western balsam bark beetle (*Dryocoetes confusus* [Swaine]) was concentrated using pheromone baiting in a manner compatible with partial- and patch-cutting silvicultural systems. In one 10-ha block (trial 1), trees located every 20 m along lines 50 m apart were baited with exobrevicomin, with a release rate of 0.4 mg/24 hr. Interbait centres were located every 20 m on lines between and parallel to the bait lines. In the second 10-ha block (trial 2), two trees were baited in the centre of what would be 0.1-ha patch cuts. In this trial, interbait centres were located in the centre of 0.1-ha leave areas. No baiting was done in the control block. The number of attacked trees within 5 m of the bait centres and interbait centres in trials 1 and 2 were analyzed as well as randomly chosen non-bait centres in the control block. Trial 2 bait centres had the highest numbers of attacked trees (median = 3) followed by trial 1 bait centres (median = 1). Interbait centres in trials 1 and 2 and the randomly chosen centres in the control block, had attack numbers approaching zero. Attack densities were also calculated as a function of distance from bait and interbait centres. For bait centres, average attack density was highest close to bait centres, dropping to close to zero as the distance from a bait increased. Interbait centres had attack densities close to zero, regardless of distance from the centre.

INTRODUCTION

As lower-elevation sites are approaching maximum cut levels, subalpine fir (*Abies lasiocarpa*) is gaining importance as a commercial species. Before the effects of human manipulation are assessed, a baseline of data must be established to document the effects of natural processes in unmanaged systems. The western balsam bark beetle (*Dryocoetes confusus* [Swaine]) is an obvious mortality agent in subalpine stands. To help develop reasonable management strategies and expectations for high-elevation stands, the spatial and temporal pattern of mortality produced by this insect should be investigated. The Sicamous Creek research area provided a unique opportunity to obtain bark beetle incidence information over time in various managed and unmanaged systems.

Life History

The life cycle of the western balsam bark beetle was first described by Mathers (1931). The bark beetles have a flight period some time during June to August. Male beetles initiate attack and colonization on suitable subalpine fir hosts, creating nuptial chambers in the phloem tissue. They mate with three to four females. The females burrow tunnels away from the nuptial chamber under the bark, which creates a characteristic gallery pattern. Eggs are laid in niches along these galleries. These adults may overwinter in the tree and produce a second brood there in the following year. Sometimes after the second brood is produced, these adults leave their host trees to form a second minor late summer flight and a third brood is attempted. Eggs hatch the same summer that they are laid. Young beetles spend their first winter as larvae, the second as callow adults, emerging for the flight period in the following spring.

Trees colonized by western balsam bark beetles are killed by the fungus C*eratocystis dryocoetidis*, which is introduced with burrowing activity (Kendrick and Malnar 1965). The summer after the tree is initially girdled and successfully colonized, the needles of dying trees turn a distinctive bright red colour. Over the next two or more years, these red needles fade and fall off leaving only the fine branches. The trees at this time are called "greys." Eventually the fine branches are lost, the bark cracks and starts to slough away, and the tree becomes a snag. Subalpine fir killed by these bark beetles presumably spend a few years in both the red and grey phases. Snags may remain standing for many years.

Colonization Dynamics

Subalpine fir are not passive participants in the colonization process. They actively resist bark beetles by producing large amounts of resin which drowns the beetles and their brood. Mechanisms of resistance to the *Ceratocystis* fungus are less well known. Many living, apparently healthy trees have dried resin streams on their trunks as evidence of successful resistance. To overcome the resistance of chosen host trees and to moderate competition on already heavily colonized trees, bark beetles use co-operative, chemical-mediated behaviour.

Mortality Estimates

Past estimates show that western balsam bark beetles are responsible for substantial mortality. In the Bulkley Timber Supply Area (TSA) where subalpine fir was a leading species in stands, standing dead trees made up 38% of the total stand volume (Stock 1991). Forty percent of the dead volume consisted of trees killed within the last six years by these bark beetles. This amounts to an average annual loss of 4.2 m^3/ha per year. Similar losses are documented by Garbutt and Stewart (1991) and by Unger and Stewart (1986).

Pathocon Consulting (1987) estimated that these beetles were associated with 55.8% of declining or recently dead subalpine fir in the Bulkley and Morice TSAS. It is likely that the beetle acts as the final agent of mortality in many dying trees. However, it is unknown whether the beetles' relationship with other mortality agents is suppressive, synergistic, or additive.

Preliminary data from Sicamous Creek show large basal area losses in three, 10-ha blocks. Recent beetle-induced mortality (green attack, red, and grey trees) accounts for 7.8–13.7% of the losses, while snags represented another 17.7–23.8% of the stand's basal area. It is unlikely that annual volume losses are counteracted by additional tree growth. Beetle mortality must vary over time.

High subalpine fir mortality is encountered when losses are estimated over large areas. This either indicates bias in the methodology (e.g., living biomass is in a steady state, averaged across all old-growth subalpine fir stands), or synchrony in the mortality (e.g., if most subalpine fir stands experience high mortality within similar time frames).

Pest Management

The purpose of baiting stands for bark beetle is to either shift attack from high- to low-value trees, or to concentrate and remove beetles from the area, which reduces losses in the remaining stands after harvest. This approach can be integrated with partial and patch-cut silvicultural systems like those used in the Sicamous Research Project. Before logging, bark beetles are drawn into the areas to be harvested. Then, beetles that attack both baited and surrounding trees are removed from the stand (along with the brood they produce) when it is logged. This approach to pest management is more cost effective and requires less expertise than finding and removing bark beetles as they naturally colonize stands. It also reduces the risk of beetle attack on remaining trees in the stand.

Using bioassays, Stock and Borden (1983) found aggregation and antiaggregation pheromones in western balsam bark beetles. Subsequent work by Borden et al. (1987) identified the aggregation pheromones myrtenol (produced by males) and exo-brevicomin (produced by females). Stock et al. (1993) attached exo-brevicomin releasing devices to subalpine fir trees and found that they could concentrate new attacks around these bait centres.

Research Objectives

Our research objectives at Sicamous Creek were:

- To quantify temporal and spatial patterns of recent mortality (i.e., red trees) from 1993 to 1995–6 at the Sicamous research site using 1:5000 aerial photographs. Similar analysis on grey and snag trees will occur if these trees are readily resolved on the photographs. Spatial patterns detected will help to generate hypotheses about underlying mechanisms. Hypotheses may also be generated by looking for associations between recent beetle-caused mortality and mapped soil and vegetation types.
- To estimate basal area or volume losses for subalpine fir caused by western balsam bark beetles in the Sicamous Creek research area.
- To demonstrate the feasibility of managing *D. confusus* with pheromone-based manipulation of new beetle colonies in ways compatible with partial- and patch-cutting harvesting regimes.

METHODS

Three 10-ha sites were used. In June 1995, baits were placed in a grid pattern in two of the three areas. In trial 1, bait lines were placed 50 m from each other. Every 20 m along the bait line, bait containing the aggregation pheromone exo-brevicomin was attached as high as possible on the north side of the nearest large subalpine fir. In this trial, bait was systematically distributed throughout the stand to provide even, complete coverage. Only one bait per centre was used to create many small openings in a stand compatible with a partial-cutting regime. In trial 2, bait lines were 33 m apart with a two-tree bait centre located every 33 m along these bait lines. At each bait centre in trial 2, one bait was placed on each of two large subalpine fir that were growing in close proximity to one another. This created fewer larger openings compatible with a 0.1 ha patch-cut system, where beetles are drawn into areas to be harvested. Two baits were used to encourage larger aggregations of beetles. In the control area no bait was used.

After beetle flight was finished in September, a complete survey of the three areas was conducted. A stem map was prepared of trees that were baited, as well as those that were classified as attacked, mass-attacked, or "red" and "grey" (see classification regime in Table 1). Baited, attacked, mass-attacked, and red trees were also measured for diameter at breast height.

TABLE 1 *Description of subalpine fir tree classes attacked by western balsam bark beetles (Stock 1991)*

Tree class	**Description**
Attacked trees	Streams of resin on trunk; indicates unsuccessful attack
Mass-attacked	Frass on trunk and possibly resin on trunk; indicates intense successful colonization of tree
Red	Red foliage present; represents old attack; new mature beetles emerge from these trees
Grey	Needles mostly gone, but fine twigs present; bark generally intact
Snag	A long-dead tree with bark loose or absent and fine twigs gone; trees must be greater than 2 m tall and 120 mm dbh

To compare baited and non-baited areas within a trial, midpoints between bait centres were mapped and called "interbait centres." For trial 1, lines running midway between, and parallel to, bait lines were mapped. Every 20 m along these lines, an interbait centre was plotted. For trial 2, interbait centres were located on the bait lines in the centre of areas that would not be cut. In the control plots, non-baited centres were randomly distributed.

Mensuration data was taken in ten 15 m radius plots randomly located in each of the treatment areas. In each plot, the dbh, species, and tree class was noted for each tree over 90 mm in diameter.

Analysis

A computer program was written to calculate the distance between bark beetle attacked trees and bait centres. The number of attacked and mass-attacked trees within 5 m of each treatment was counted for each bait and interbait centre. A non-parametric test (Kruskal and Wallis – Minitab) was used to test for differences between the following populations, while a non-parametric test for unequal sample sizes was used to compare the means (Zar 1984):

- Trial 1 – bait centres
- Trial 1 – interbait centres
- Trial 2 – bait centres
- Trial 2 – interbait centres
- Control – randomly chosen centres

A non-parametric test was chosen because the variance in the number of attacked trees was very high close to the bait trees, decreasing to near zero as distance from the baited trees increased, and for counts around interbait centres and randomly chosen centres in the control.

The number of trees in different distance classes was counted, and the density calculated using the following formula:

$$\text{Density} = \text{area}/\#\text{ of trees in a distance class } d_1,$$

$$\text{where:}\quad \text{area} = \sum_{1}^{t} \pi d_1 d_1 - \sum_{1}^{t} \pi d_2 d_2,$$

$$d_2 = d_1 - 1\text{ m, and}$$
$$t = \text{number of bait centres.}$$

For trials 1 and 2, the distance classes ran from 1 to 10 m (at 1-m increments) and from 1 to 16 m (at 1-m increments), respectively. Attack density was calculated for both treatments for both bait and interbait centres.

RESULTS

The number of trees in each attack class in each treatment area is shown in Table 2. A significant difference existed in the number of attacked trees within 5 m of the different centres (Table 3). No significant difference existed in the number of trees between control centres and the interbait centres in both trials 1 and 2. The bait centres for both trials 1 and 2 were significantly different from interbait centres for trials 1 and 2 and the centres in the control area. They were also significantly different from each other (Table 4).

Figure 1 shows the average attacked-tree (unsuccessful and successful) density around bait and interbait centres for trials 1 and 2 as a function of distance. In trial 1 bait centres, attack density was very high close to the bait, but decreased rapidly as distance from the bait centre increased. This illustrates how most bait trees attract beetles with some residual spillover into neighbouring trees. Trial 2 shows a similar pattern, although attack density was not as high close to the bait centres when compared to trial 1. However bait centres in this trial are located between the two bait trees.

TABLE 2 *Number of trees per hectare in the three treatment plots*

Attack category	Number of trees per hectare		
	Trial 1	Trial 2	Control
Attack	2.0	7.5	2.2
Mass attack	2.5	8.7	4.3
Red trees	8.8	27.5	15.8
Grey trees	21.7	19.9	10.6
Snags	114.2	122.4	99.2

TABLE 3 *Kruskal-Wallis single factor analysis of variance by ranks for the number of attacked and mass-attacked trees within 5 m of various centres*

Trial and bait centre	*N*	Median	Average rank	*Z* value
Trial 1: Bait centres	86	1.0	261.1	7.99
Trial 1: Interbait centres	93	0.0	127.0	–5.86
Trial 2: Bait centres	41	3.0	325.7	9.31
Trial 2: Interbait centres	41	0.0	128.6	–3.46
Control	102	0.0	129.3	–5.98
Overall	363		182.0	
$H = 187.43$ $df = 4$	$p = 0.000$			
$H = 277.91$ $df = 4$	$p = 0.000$ (adjusted for ties)			

TABLE 4 *Ranking (in ascending order) of attack within 5 m of bait (BC), interbait (IBC), and control centres in trials 1, 2 and the control (underlined mean ranks indicates no significant difference [$p \leq 0.05$])*

Ranking	Trial 1 (IBC)	Trial 2 (IBC)	Control	Trial 1 (BC)	Trial 2 (BC)
Rank sums	11 811.0	5272.6	13 188.6	22 446.0	13 353.7
N	93	41	102	41	86
Mean ranks	127.0	128.6	129.3	261.0	325.7

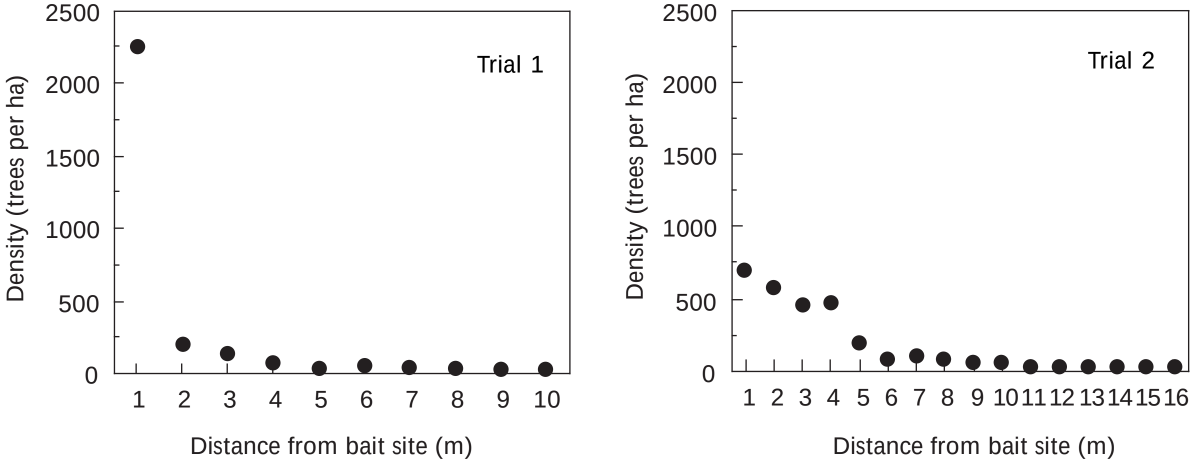

FIGURE 1 *Average density of attacked (successful and unsuccessful) trees as a function of distance from bait centres.*

The attack density did not drop off as rapidly as in trial 1, but was diffused over a larger distance.

To summarize, more attacked trees occurred within 5 m of bait centres than interbait and randomly chosen centres in the control plot. More attacked trees also occurred within 5 m of the two-tree bait centres than the one-tree bait centres. However, in trial 2 more red trees per hectare occurred than in trial 1, which could partially explain this difference. More bark beetle colonies may have existed in the trial 2 area than in the trial 1 area, as red trees are their source.

The number of attacked trees around centres in the control area did not differ significantly from interbait centres in trials 1 and 2. However, the number of attacked trees per hectare was higher in the control plot than in trial 1 (Table 2). Since most of the attack in trial 1 was concentrated around bait centres, the lack of difference between the interbait centres in trial 1 and the control centres is likely attributed to the method of analysis used.

While baiting may not influence the overall numbers of trees killed by western balsam bark beetles, it definitely helped to concentrate the attack in a predictable fashion.

DISCUSSION

This baiting experiment shows that it is possible to concentrate western balsam bark beetles in subalpine fir stands in ways compatible with partial- and patch-cut silvicultural systems. However, one of the goals of pheromone baiting is also to reduce future mortality by removing bark beetles from the system. The effectiveness of this strategy depends mainly on the dispersion patterns of bark beetles across the landscape and the type of interactions beetles have with other mortality factors within the stand.

Removing enough bark beetles from a stand will create a small area with depressed beetle numbers that is embedded in a landscape of uncontrolled patches. Reducing future mortality will depend on how quickly the surrounding uncontrolled bark beetles disperse into the previously baited area in large enough numbers to successfully colonize trees. The ratio of areas where bark beetles are actively controlled to areas of no control will affect how quickly this dispersal occurs. The spatial relationship of controlled to uncontrolled areas is also likely to be important. If the beetles disperse readily over large distances (i.e., over 1 km), baiting over larger areas may be necessary to control mortality over a meaningful time period. It is also possible that in the year following successful baiting, stands will be bypassed by beetles dispersing from other areas, if pheromone sources are largely eliminated from the stand.

The western balsam bark beetle is often associated with mortality agents such as root rots (James and Goheen 1981; Pathocon Consulting 1987). It is not known how the removal of bark beetles from the system will affect mortality patterns with time. If interactions are additive or synergistic, baiting could be a success.

This trial demonstrates that within-stand management of western balsam bark beetles is possible in partial- and patch-cut baiting systems. In a system simulating partial cutting, single-tree baits produced many small concentrations of attack that are easily found. This system is flexible and allows multiple entries into a stand over time, providing a choice of future management options. Controlling bark beetles in leave areas is also a useful option. Using the single-tree baiting regime, some expertise is needed to recognize and remove mass-attacked trees around bait centres.

Two-tree bait centres created fewer but larger areas of attack in a manner compatible with patch-cut silvicultural systems. This system requires little expertise as the whole area around the bait centres will be cut. However, because larger volumes are extracted, future baiting options are constrained to a greater extent than in partial-cutting systems.

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Forest Pathology at Sicamous Creek

HADRIAN MERLER

ABSTRACT

The silvicultural systems research installation at Sicamous Creek offers an opportunity to study the effects of forest tree disease with various stand harvesting treatments in high-elevation forests.

Permanent survey plots were established to track forest tree pathogens on regeneration and residual mature trees. Tree stem and root dissections were done to investigate the phenomenon of small copse tree mortality. Observed pathogens include *Armillaria ostoyae* root disease and *Phellinus pini* heart rot. A large proportion of root damage to subalpine fir was caused by rodent feeding.

INTRODUCTION

The pathology of higher-elevation forested stands, particularly in the Engelmann Spruce – Subalpine Fir (ESSF) biogeoclimatic zone, is not complete. A study of pathogenic organisms that exist in these stands is needed to write proper prescriptions for this ecosystem. This will be done by an inventory of pathogens and examination of groups of dead trees.

Forest tree diseases may pose impediments to successful site reforestation and stand management goals. Depending on the diseases present, silvicultural systems other than clearcutting may increase the diseases' effect on residual woody forest species.

The intent of this project is to inventory pathogens and correlate their occurrence with the life and death of forest trees. The study will assess the effect that opening size might have on pathogen dynamics. Destructive sampling of individual trees will help determine the organisms which cause tree mortality. This will help to define individual stem health and small group, forest community health. Tracking the development of revegetation within treatment blocks will also determine pathology in this ecosystem.

METHODS

Rates of mortality of existing stems were determined by locating permanent plots within the intact forest found in leave strips between harvested treatments. Sampling consisted of three 50 × 2 m strip plots per treatment unit. This resulted in a total of 45 plots (three replicates of five treatments times three plots).

Within each plot, the location of all stems (distance from origin) as well as the species, diameter, health (grey, red, or green), and pathology (symptoms or signs) were recorded. These measurements will be repeated in 2-, 5-, and 10-year intervals.

A sample of mature trees that represented group mortality was dissected for an identical forest type, 300 m outside of the Sicamous Creek research study area. Ten groups of dead trees consisting of at least three dead trees per group were randomly selected in a 10-ha area. Each group had their root systems exposed and examined for pathogens, wounds, or decay. Samples were taken for fungal isolation to determine decay-causing organisms. Within the Sicamous Creek research study area, 10 randomly selected groups of dead trees were chosen to compare the tree dissection observations. The groups of dead trees within the study area were not destructively sampled. Observations of obvious pathogens, wounds, or decay were made by temporarily exposing root systems of such groups of trees.

RESULTS

Abiotic damage resulting from frost and wounds caused by falling neighbouring trees was responsible for most of the aboveground symptoms. Proportionately greater damage occurred on subalpine fir than on spruce. Frost damage was normally distributed by stem diameter. Blowdown, resulting from wind or snow, occurred exclusively on trees of less than 15 cm diameter. Of all treatments, wounds on residual trees were greatest in the single-tree selection treatment.

Four groups of organisms are implicated in the pathology of the forest vegetation (Table 1). These agents were found consistently throughout both the dissection plots outside the study area and within the study area. The status of subalpine fir and spruce in the survey plots is presented in Tables 2 and 3.

TABLE 1 *Four groups of organisms implicated in forest pathology*

Agent	Occurrence on hosts
Rodents	89% roots of subalpine fir only
Pathogenic root fungi	30% of both host species' roots had *Armillaria ostoyae*
Decay fungi	60% of dead stems had *Phellinus pini* present
Insects	Several species of bark-inhabiting insects were noted; not identified nor quantified

TABLE 2 *Subalpine fir status by percentage in treatment plots*

Treatment	Healthy	Red	Grey	New stumps
10 ha	68	3	25	4
1 ha	69	3	23	5
0.1 ha	64	1	24	11
Single-tree selection	60	2	13	17
Control	78	1	17	4

TABLE 3 *Spruce status by percentage in treatment plots*

Treatment	Healthy	Red	Grey	New stumps
10 ha	97	0	3	0
1 ha	87	0	8	5
0.1 ha	89	2	2	7
Single-tree selection	69	0	0	31
Control	85	0	3	12

The presence of damaging agents may be a direct result of the way in which trees grow in these forests. Tree distribution is strongly influenced by mounds or hummocks left by previous generations of dominant vegetation. This appears to contribute to the habitat for rodents and concentrations of various tree-inhabiting fungi.

Rodent damage was dominant on subalpine fir. Lesions from rodent feeding occurred as oblong patches of striped root bark from 10 to 80 cm long. Some roots less than 8 cm diameter had been girdled. Larger roots were rarely girdled. The lesions extended into the primary xylem tissues. Most of this damage had occurred during the 1995 growing season, as seen by the lack of tree wound response. Some older rodent-caused lesions were observed. Older lesions near the centre of the tree were associated with wood decay. The red-backed vole (*Clethrionomys gapperi*) may be the rodent responsible, but this has not been observed. These observations suggest this damage is periodic in nature. Very few lesions occurred on spruce and were not as large in size as on the subalpine fir.

Armillaria ostoyae rhizomorphs were quite abundant on roots of both subalpine fir and spruce. However, *A. ostoyae*-associated mortality within the plots was restricted to spruce greater than 25 cm in diameter and immature subalpine fir. Only one large subalpine fir (35-cm dbh) was killed by this pathogen. Correlations of this disease with the various treatments will be made during future measurements.

Frost cracks and stem damage caused by frost were most prevalent on stems that also exhibited white rot decay from organisms such as *Phellinus pini*. Uncertainty exists as to whether this decay results from freezing damage to the outer bark, or if the decay causes stems to expand differentially during freezing and thawing, which would lead to stem splitting.

Wind and Temperature Regimes in the B-5 Clearing at the Sicamous Creek Silvicultural Systems Research Area: Preliminary Results from 1995

MICHAEL NOVAK, ALBERTO ORCHANSKY, RALPH ADAMS, WENJUN CHEN,
AND RICHARD KETLER

ABSTRACT

One of the main objectives of our research at Sicamous Creek was to validate an extensive set of wind and turbulence measurements that we made recently in the Mechanical Engineering wind tunnel at the University of British Columbia using various forest and clearing configurations. A further objective was to measure the effects of wind regimes on microclimate and tree sway. These measurements can not be done in the wind tunnel, but are crucial if we are to achieve our ultimate goals of determining effects of clearing size and shape on tree seedling growth and windthrow. During July to November 1995, preliminary measurements of wind speed and direction and air temperature were made along a west–east transect in the B-5 10-ha clearing and at the top of the forest adjacent to the west (upwind) edge.

The results show that wind was generally from the west during daytime in July and August, but this shifted to the south in the fall. At night, wind was generally from the south (down the prevailing slope). When the wind was from the west, strong variations in average wind speed occurred across the clearing. These results agreed to within 15% with our wind tunnel measurements made with a 1:200 scale model of the B-5 clearing. We believe that the discrepancies are mainly related to factors that were not modelled in the wind tunnel—surface roughness and topographical changes along the transect at Sicamous Creek. Midday air temperature also varied across the clearing, being higher near both forest edges, but especially in the quiet zone at the upwind edge. According to our model, daytime sensible heat flux density is considerably less in the upwind quiet zone (despite the higher average air temperature) than in the middle of the clearing. This is a symptom of weaker turbulent transfer in this zone.

Current forest harvesting strategies or those considered for use in British Columbia range from clearcutting to single-tree and group-selection methods. Clearcutting of very large blocks was previously the method of choice, but the province's recently implemented Forest Practices Code puts upper limits on cutblock area. Recognition of the negative environmental effects of traditional methods of forest removal is forcing the industry to consider less damaging options, including single-tree and group selections.

Two major aspects that must be considered in choosing among management scenarios are the microclimates experienced by planted tree seedlings and the blowdown of timber. Successful forest regeneration in large clearcuts is difficult, especially at the higher elevations that are now routinely harvested in British Columbia. In large part, this is because of the harsh microclimates created in cleared areas. Regeneration should improve in smaller clearings and partially cut areas which more closely mimic natural forest conditions. Major factors governing microclimate are wind speed and turbulence, which depend strongly on clearing size, shape, orientation, and topography, and vary spatially in clearings and partially cut areas in a complex way.

Blowdown of trees along cutblock edges is a chronic problem for the province's forest industry, costing millions of dollars annually. Immediately following harvesting, trees along cutblock edges are not yet adapted to the higher wind speed and turbulence levels that typically occur in the cleared areas. This problem may be more acute with smaller cutblocks and partial cutting because the total length of edges will actually increase, although lower wind speed and turbulence in the smaller blocks should lessen the effect. Wind forces on edge trees for different clearing size, shape, orientation, and topography are still unknown. Steady forces that arise from mean wind speed and dynamic forces that are associated with turbulence are implicated in the blowdown process.

The objective of our research is to systematically investigate the effects of clearing size, shape, orientation, and topography on microclimate and windthrow potential. Most of the experiments are being done in the Department of Mechanical Engineering blow-through wind tunnel with forest and clearing configurations made from model trees 15 cm in height (Chen et al. 1995; Liu et al. 1996). However, an essential component consists of field measurements at the Sicamous Creek Silvicultural Systems research area. These are required to validate the wind tunnel model results and to gather information not measurable in the wind tunnel (e.g., microclimate and tree sway data). This paper reports some preliminary results from our first field experiments conducted at Sicamous Creek in July to November, 1995.

The B-5 10-ha clearing is square with sides 326 m in length that are oriented about 10° off true north–south and east–west directions (see Vyse, this proceedings, Figure 2, page 9). It is situated on about a 12% north-facing slope with contours running roughly east–west. The surrounding forest is estimated to be 30 m tall, so that the clearing is 10.9 tree heights on each side. A west–east transect of six 8-m towers was established along the midline of the clearing during July. Locations were at 25, 75, 125, 200, 250, and 300 m from the west edge, designated as towers 1–6, respectively. RM-Young 05103 anemometers were mounted at an 8 m height on towers 2–5, and an RM-Young 05305 anemometer (more sensitive with lower stall speed) was mounted at an 8 m height on tower 1. These anemometers measure horizontal wind speed and direction. A 75-μm chromel/constantan thermocouple was mounted at a 2 m height on each tower to measure air temperature. A li-lor 200-s silicon pyranometer mounted level and facing south on tower 5 measured solar irradiance. Unless indicated otherwise, output from these instruments was sampled at 0.5 Hz and averages and wind statistics were recorded half-hourly and daily from July 13 to November 16 using Campbell Scientific cr21-x programmable micro-loggers. Only 15 days of data (August 26–29 and September 26–October 6) were lost.

In September, a 5-m tower was installed at the top of a completely limbed tree (27 m high) located 100 m into the forest on the west edge along the transect. An RM-Young 05103 anemometer and chromel/constantan thermocouple were installed on the tower at heights of 5 and 0.3 m, respectively, above the top of the limbed tree. We have not yet measured the true height of the forest at the site, but we are confident that the anemometer was located very near the top of the trees. Sampling at 0.5 Hz and recording half-hourly and daily from this tree occurred from September 26 to November 16, although data monitoring actually began on September 13 with the intensive experiment (described below). One role of this tower, designated as tower 0, is as an overall reference for all experiments in the B-5 clearing.

An intensive experiment was carried out September 13–25. A Swissteco s-1 net radiometer was mounted facing south at a height of 1.5 m near tower 2. The air temperature thermocouples were monitored differently during this experiment. The temperatures were sampled at $1 / \Delta t = 2$ Hz, where Δt is the sampling time interval, and the cubic structure function was accumulated on-line and averaged half-hourly. The cubic structure function is defined as, $\overline{\Delta T^3} = \overline{[T(t) - T(t - \Delta t)]^3}$, where T is air temperature, t is time, and the overbar is an average for a half-hour period. During the intensive experiment wind direction was not available from any of the towers.

According to Chen et al. (1996), the cubic structure function can be used to calculate the sensible heat flux density in the atmosphere, *H*, which for the west–east transect is a measure of the effectiveness of vertical mixing by turbulence at the different locations. The final formula is given by:

$$H = -0.4\rho c_p \left[\frac{\overline{(\Delta T)^3}}{\Delta t}\right]^{1/3} u_*^{2/3} \frac{z}{h^{2/3}}, \qquad \text{for } 0.2h < z \le h + 2(h - d), \tag{1}$$

$$H = -0.4\rho c_p \left[\frac{\overline{(\Delta T)^3}}{\Delta t}\right]^{1/3} u_*^{2/3} \frac{z}{(z - d)^{2/3}}, \quad \text{for } z > h + 2(h - d) \text{ or } z \le 0.2h,$$

where: ρ and c_p = the density and specific heat of air, respectively,
such that $\rho c_p \approx 1200$ J/m^3 per °c,
u_* = the friction velocity,
z = the height above the surface,
d = the displacement height, and
h = the height of the underlying roughness elements.

The time delay, Δt (which is also the sampling interval), should be such that $-\overline{(\Delta T)^3}/\Delta t$ is at its maximum value. To calculate u_* we used the logarithmic wind law for neutral conditions:

$$u_* = \frac{k u_z}{\ln[(z - d)/z_0]} \tag{2}$$

where: $k \approx 0.4$ = von Karman's constant,
u_z = the mean horizontal wind speed at height z (8 m for the clearing and 30 m for the top of the forest), and
z_0 = the roughness length.

We made the standard approximations that $d = ⅔\, h$ and $z_0 = h/10$. For the clearing it was assumed that $h = 0.5$ m, which seemed reasonable considering the amount of existing slash, stumps, mounds (from site preparation), and vegetation, while for the top of the forest, $h = 30$ m. The logarithmic law is strictly valid in the inertial layer for horizontally homogeneous situations under neutral atmospheric conditions. The situation in the clearing is anything but that, especially near the west edge, and the top of the forest is in the roughness sublayer which is known to differ from the overlying inertial sublayer. However, we felt that possible errors were acceptable, given the preliminary nature of the results.

RESULTS AND DISCUSSION

Solar irradiance for selected weeks throughout the study is shown in Figure 1. Energy loading in the clearing is considerable in July, reaching a maximum of about 900 W/m^2 at midday on clear days. The long daylength in summer is evident. By early in November, the maximum solar irradiance had declined to about 400 W/m^2 at midday. Also shown is net radiation, R_n, for one week during the intensive experiment.

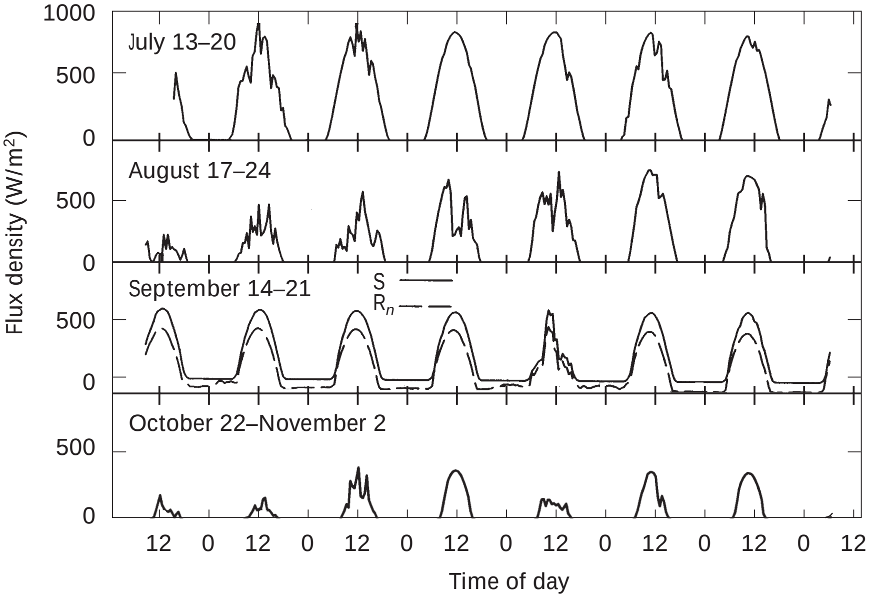

FIGURE 1 *Solar irradiance during selected weeks at Sicamous Creek in 1995. Also shown is net radiation for one week during the intensive experiment in September.*

During this period the weather was excellent with a number of consecutive clear days.

Average wind speed measured along the west–east transect throughout the experiment is shown in Figure 2. Only those wind speeds that were within 30° north or south of west at towers 2 to 5 are shown. The percentage of time the wind blew in this direction is shown in brackets for each month. A strong decline in westerly wind frequency occurred between July and November with an associated decrease in wind speed (from about 2–2.5 to 1.5–2 m/s near the east edge of the clearing). Generally, winds were from the west during the daytime. At night, winds usually blew downslope from the south—a typical anabatic flow. We designed the experiment based on the previous two years of meteorological data collected by one of us (R.S. Adams) at the Sicamous Creek base camp, 2 km to the west of the B-5 clearing. These data suggest that westerly winds would prevail, especially during the period of highest wind speed in fall. In 1995 at least, this turned out to be true more in summer than in fall.

The monthly average patterns of wind speed across the clearing shown in Figure 2 are remarkably consistent during the study period (wind speed at tower 1 was not available in October and November because of sensor failure). This is also seen during midday on four days in July and August (Figure 3). The low wind speed near the (upwind) west edge is consistent with the existence of a "quiet" zone extending about 3–6 tree heights into the clearing at $z = 8$ m (0.27 of tree height) and a "wake" zone extending to very near the downwind edge (Figure 4). Such a pattern is a recognized feature of flow between windbreaks (McNaughton 1988). The decrease of wind speed near the downwind edge suggests that a weaker quiet zone exists there.

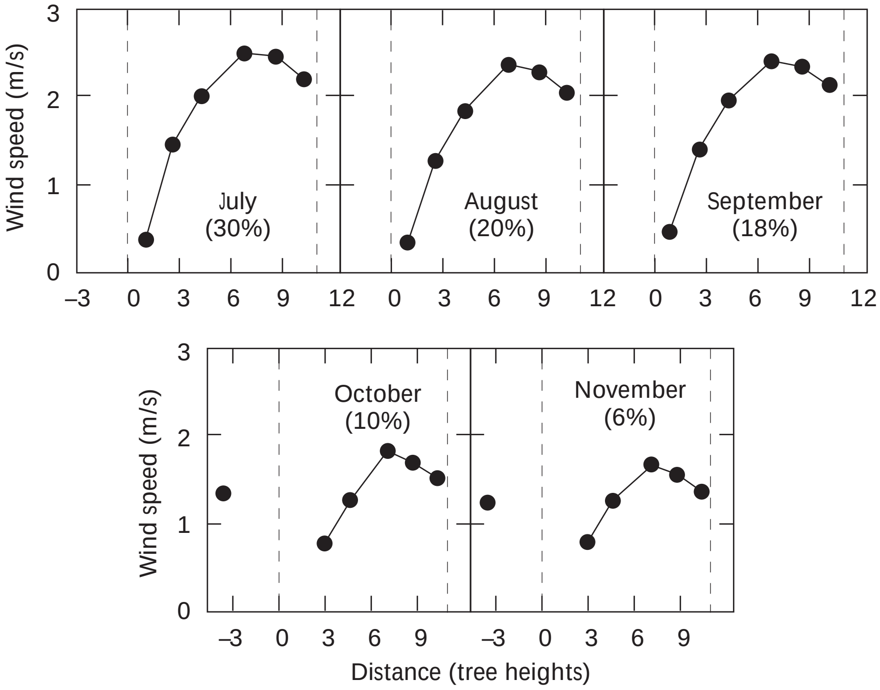

FIGURE 2 *Mean wind speed variation across the B-5 clearing at Sicamous Creek for winds from 270° ± 30° (according to towers 2 to 5) for July to November 1995 (also shown is the percentage of time that the wind blew from this direction).*

Midday air temperature profiles across the clearing show that these zones have a measurable effect on air temperature (Figure 3). Air temperature is about 2°c higher in the upwind quiet zone compared to the wake zone (tower 4). A smaller increase of temperature occurs near the downwind edge. Note that because the transect is west–east, all towers have similar exposure to solar irradiance at midday.

Figure 5 compares the July, August, and September average wind speed profiles across the B-5 clearing with that measured in the wind tunnel at the University of British Columbia. For the wind tunnel, we set up a 1:200 scale model of the B-5 clearing. The surface of the clearing in the model was smooth and flat (plywood). Measurements were made at locations which correspond to the six towers in the clearing, plus the additional locations indicated (z = 4 cm in the wind tunnel corresponds to z = 8 m in the field). The measurements were normalized by the value at tower 6, since the reference tower was not yet installed in the field.

Agreement between the field and wind tunnel measurements is quite good (within 15%). The maximum disagreement is for tower 4. We suspect that this is because the surface of the wind tunnel is both smoother and flatter than conditions which exist in the field. In our most recent wind tunnel work, the model clearing was roughened by adding 1 cm high uniformly spaced trees, which increases the wind speed measured at z = 4 cm by about 5–10%. Topographic maps (1:10 000) of the Sicamous Creek area and a recent field elevation survey show that the soil

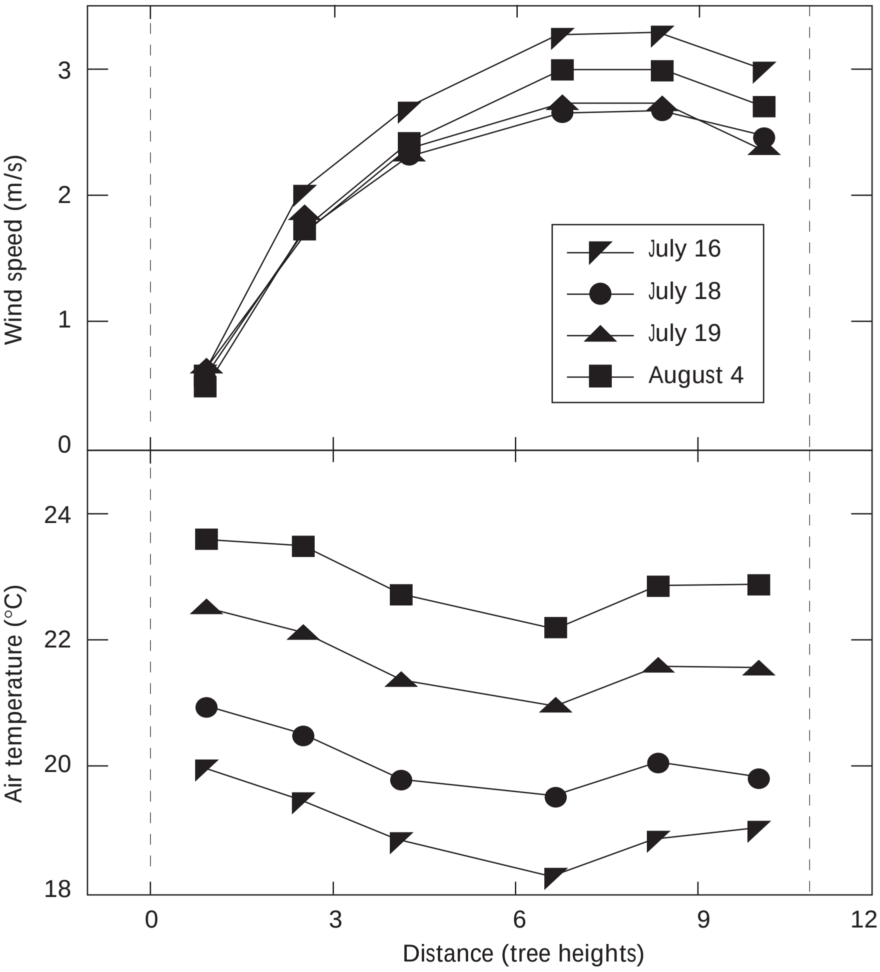

FIGURE 3 *Midday average wind speed and air temperature variation across the B-5 clearing at Sicamous Creek on four selected days in July and August 1995. Only winds from 270° ± 30° (according to towers 2 to 5) are included.*

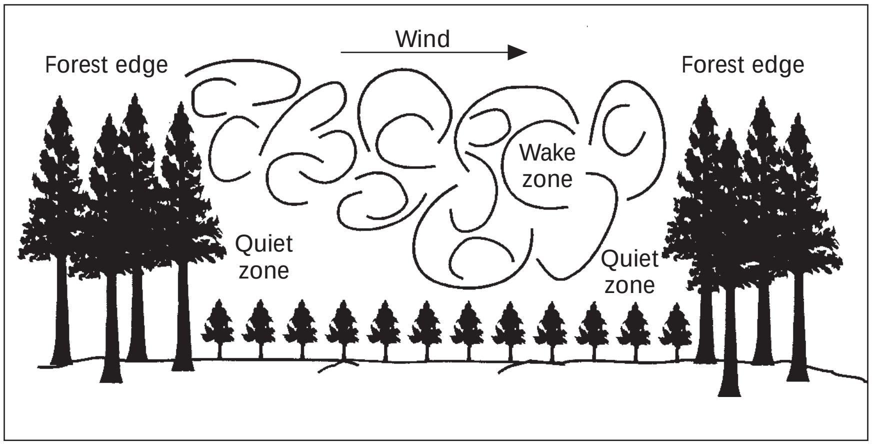

FIGURE 4 *Schematic of the wind and turbulence in a forest clearing, showing upwind and downwind edge quiet zones and the wake zone.*

surface at tower 4 is about 5 m above both west and east edges of the clearing. Therefore, wind speed near the middle of the clearing should be higher than if this surface was flat because wind speed increases with height above the surface at all locations in the clearing. We estimated this effect by substituting the $z = 7$ cm values for the locations corresponding to towers 3 and 4 in the wind tunnel results, as indicated in Figure 5. This improves the agreement for tower 4, but the wind speed is then overestimated for tower 3. More recent work suggests that this point is anomalous, however. The wind tunnel measurements shown are based on a single profile at each location and because of the intrinsic variability of turbulent flow, are subject to random fluctuations. With more profiles, the average at this location for $z = 7$ cm is lower. Using our measurements of the topography of the B-5 clearcut, we plan to construct a scale model that will allow a more precise comparison between the field and wind tunnel.

Figures 6 and 7 show diurnal variations of measured net radiation (R_n) and calculated sensible heat flux density (H) (from equations [1]and [2]),

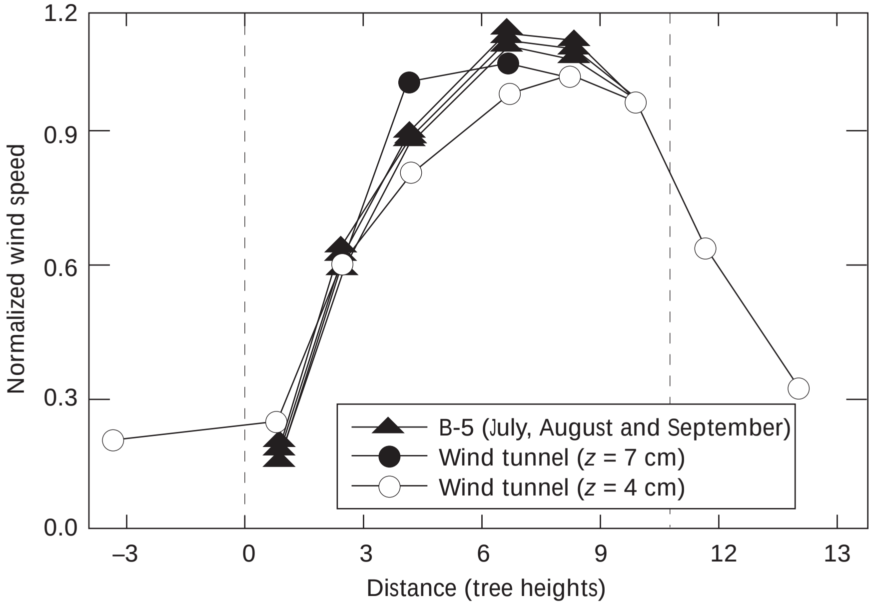

FIGURE 5 *Mean wind speed variation across the B-5 clearing at Sicamous Creek (for winds from 270° ± 30°, according to towers 2 to 5) for July, August, and September compared to wind speed at z = 4 cm across a 1:200 scale model of the clearing in the wind tunnel. Also shown is the variation in the model clearing with z = 7 cm wind speeds substituted for towers 3 and 4.*

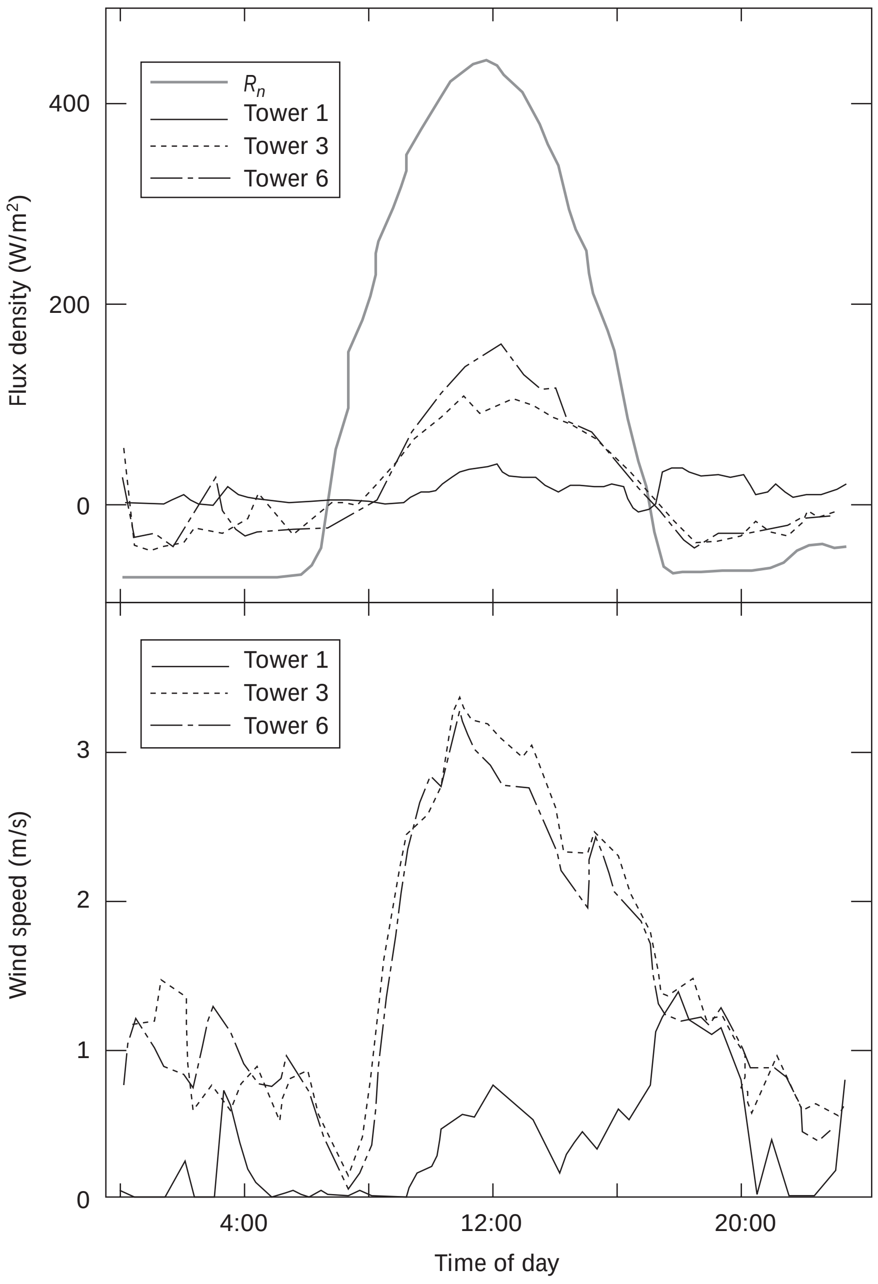

FIGURE 6 *Diurnal variations of sensible heat flux density calculated with the renewal model, net radiation (tower 2 only), and wind speed for towers 1, 3, and 6 in the B-5 clearing at Sicamous Creek on September 17, 1995 (daytime winds were mostly from the west).*

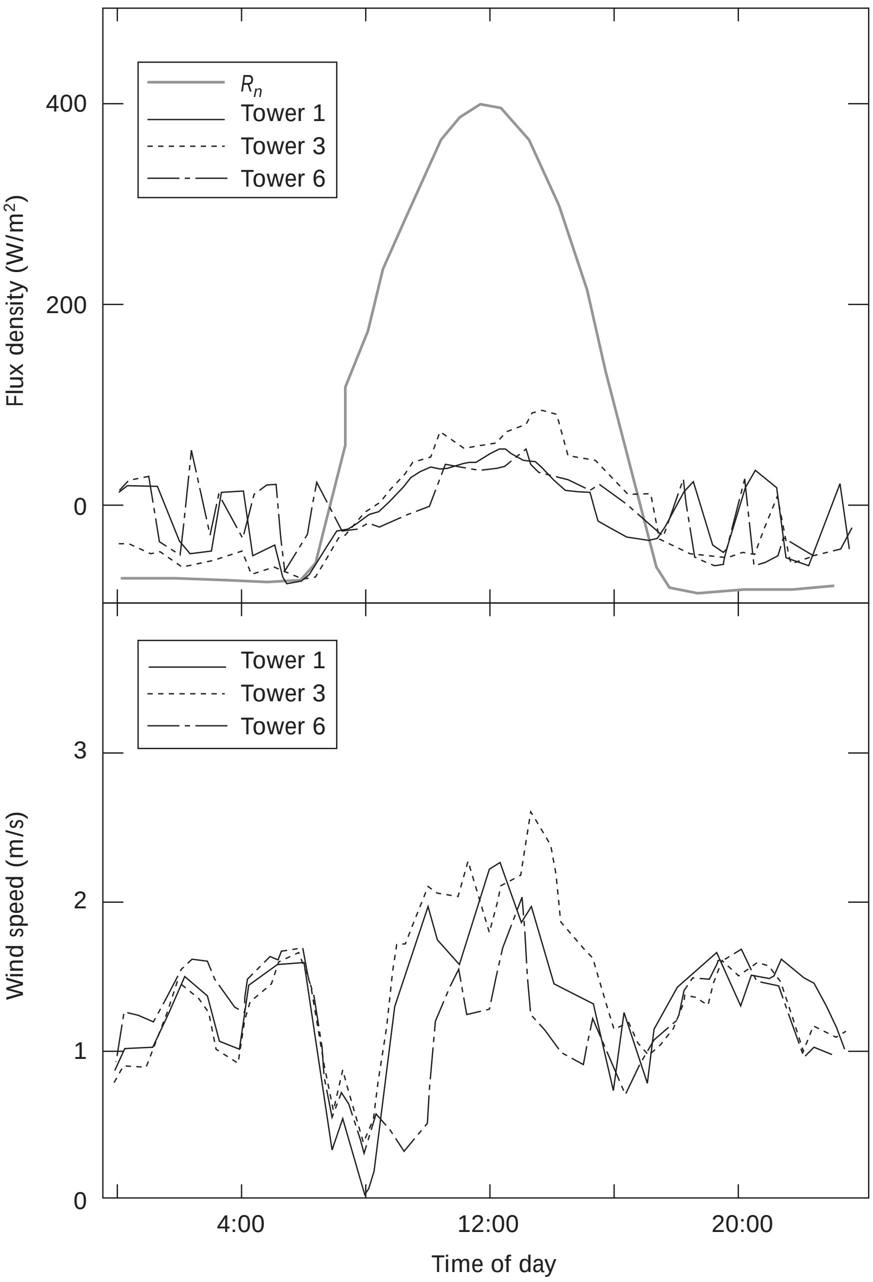

FIGURE 7 *Diurnal variations of sensible heat flux density calculated with the renewal model, net radiation (tower 2 only), and wind speed for towers 1, 3, and 6 in the B-5 clearing at Sicamous Creek on September 23, 1995 (daytime winds were mostly from the north or south).*

and measured friction velocity ($u_{8\mathrm{m}}$) for towers 1, 3, and 6 on September 17 and 23, respectively. On September 17, daytime wind blew mostly from the west, while on September 23 it was either from the north or south since wind speed was very similar for all towers in the clearing. For September 17, daytime sensible heat flux density increased progressively from towers 1 to 6, the largest increase being between towers 1 and 3. In accordance with Figures 2, 3, and 5, friction velocity increased across the clearing as well. The increase of sensible heat flux density across the clearing is attributed to a combined increase of u_* (calculated from equation [2]) and of $-\overline{(\Delta T)^3}$. Results for towers 2, 4, and 5 are intermediate between those shown. In contrast, for September 23 both $u_{8\mathrm{m}}$ and H were very similar across the clearing.

For both days, daytime H was a small fraction of R_n (at most about 30%), which seems reasonable because Sicamous Creek is a wet site. The difference between R_n and H goes into latent heat flux and soil heat flux. Presumably both of these were greater during daytime in the upwind quiet zone compared to the wake zone on September 17. Considering the large amount of slash on the site, soil heat flux was probably quite low, so most of the decrease of daytime H in this quiet zone would have been accompanied by an increase of latent heat flux. This is supported by the higher measured air temperatures—saturated vapour pressure increases strongly with temperature (McNaughton 1988). An increase of the vertical vapour pressure gradient would be required to overcome the lower turbulent transfer capability of this quiet zone.

SUMMARY AND CONCLUDING REMARKS

Micrometeorological measurements were made during July to November 1995 in the B-5 10–ha clearing at the Sicamous Creek Silvicultural Systems research area. During summer, daytime wind blew mostly from the west, but this predominance declined in the fall; at night, wind blew from the south, or down the main slope. When the wind was from the west, the variation of wind speed across the clearing was remarkably consistent during the five months. This reflected the existence of a strong quiet zone near the upwind edge and a weak zone at the downwind edge, with a large wake zone between. These zones were first described in the context of wind flow through and over windbreaks.

The variation of wind speed across the clearing agrees within 15% with that measured in the wind tunnel using a 1:200 scale model. Disagreement is mostly attributed to differences in surface roughness and topography between field and wind tunnel. These are aspects that we shall investigate further.

Sensible heat fluxes were calculated from measured wind speed and air temperature fluctuations with a renewal model. These heat fluxes are considerably lower in the upwind quiet zone compared to the wake zone. We speculate that these higher temperatures are a symptom of poorer turbulent mixing in the quiet zones and, at least in the upwind zone, are associated with lower (not higher as one might expect) sensible heat fluxes.

The 1995 measurements and application of the renewal model remain preliminary. We are testing the renewal model in the summer of 1996 with direct measurements of sensible heat flux density by eddy correlation using one-dimensional sonic anemometers. We also intend to measure all components of the surface energy balance in the quiet and wake zones in the 10-, 1-, and 0.1-ha clearings. Recent wind tunnel measurements show that the quiet zone dominates the two smaller clearings. Ultimately, we plan to use three-dimensional sonic anemometers to measure turbulence levels in the field for comparison with the wind tunnel measurements.

ACKNOWLEDGEMENTS

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Spatial and Temporal Response of Vegetation to Silvicultural Treatments in ESSF Forests at Sicamous Creek

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ABSTRACT

Several complementary studies have been initiated to evaluate the effects of forest harvesting on forest succession, floral diversity, plant phenology, and weed invasion. Stand-level effects of opening size, proximity to cutblock edges, and different site preparation practices are being studied. The measured variables include species composition, abundance, vigour, height, phenology, and site factors. The response of all vascular and non-vascular plants is being monitored.

Preliminary results for the first year after harvesting is summarized. The data set provides a good baseline against which future changes in composition, cover, and vigour can be compared. A sharp gradient in plant community structure and species height occurs within 7 m of the cutblock edge. The diversity, abundance, and height of moss, lichen, herb, and shrub species decrease at a short distance from the cutblock edge. Species diversity and vigour is similarly reduced as the level of logging disturbance and opening size increases from the uncut controls to 10-ha openings. The initial response of moss and lichen species is more dramatic than for shrubs or herbs.

INTRODUCTION

Professional foresters have debated the merits and methods of harvesting British Columbia's interior high-elevation forests for decades. Increased public awareness, poor regeneration success, and recognition of other forest values has prompted growing concern over the suitability of clear-cutting at high elevations. The harsh climatic conditions encountered in the Engelmann Spruce – Subalpine Fir (ESSF) zone, including a short growing season, heavy snow packs, mid-summer frosts, cold soils, and low temperatures, places ESSF forests at the upper-elevation limits of tree establishment and growth (Lloyd et al. 1990).

Large areas of the ESSF zone are harvested each year, initiating the process of secondary succession. Trees, shrubs, herbs, mosses, and lichens are key species in all terrestrial ecosystems and are often the most visible indicators of environmental changes induced by logging and silvicultural practices. A good understanding of the consequences of forest harvesting on forest succession is critical to sound forest management (Coates 1987; David 1987; Caza 1991). However, the rates of vegetation recovery, successional pathways, and changes in species diversity, vigour, phenology, and biomass production following various silvicultural practices in the ESSF zone are not well documented. The interactions between vegetation and wildlife response, nutrient cycling, regeneration success, and microclimate are also poorly understood. In general, vegetation serves as hiding cover, shelter, and a source of food for many small mammals, birds, insects, and microfauna. Understorey vegetation may represent an impediment to tree regeneration through its influence on light availability, competition for moisture and nutrients, or by physically damaging seedlings through whipping or vegetation press (Coates 1987; Lloyd and Stathers 1994). Following logging disturbances, the root systems of many understorey species serve as refugia for microfauna and flora, including mycorrhiza which is often important for long-term forest productivity and health. Roots and surface vegetation represent an important nutrient sink and a source of organic matter. Plants also recycle nutrients from deep in the soil, which enhances soil fertility. Vegetation also protects the soil from raindrop and surface erosion.

OBJECTIVES

Four complementary studies were established to explore various specific vegetation response questions (see Figure 1). All evaluate the effects of silvicultural practices in the ESSF zone on natural tree regeneration and the spatial and temporal changes in the species diversity, composition, abundance, height, and the phenology of vascular and non-vascular plants.

Two studies focus on the effect of five harvesting treatments on plant community development. The five treatments (control, single-tree removal, and 0.1-, 1-, and 10-ha clearcuts) represent a range of canopy opening sizes which creates a gradient in climatic conditions for light, snow accumulation and melt, soil temperature, and air temperature. The resulting climatic variation should have a profound influence on vegetation response. The first of these studies looks intensively at the mesic ESSFwc2-01 site series (Lloyd et al. 1990) within the mid-elevation replicate at the study site. The second uses large permanent sample plots, samples all three elevation replicates, covers five ecologically distinct soil moisture conditions, ranging from subxeric to subhydric, and includes the 01, 04, 06, 08, and 10 site series. The third is an edge effect study, which examines tree seedling and vegetation response at increasing distances from the edge of cutblocks, both into the opening and into the adjacent forest. This study also compares north with south edges and undisturbed surface substrates with bare mineral soil, at distances of 0, 7, 14, 28, and 56 m from the cutblock edge. These distances correspond to 0, 0.25, 0.5, 1, and 2 times the average tree height of the adjacent stands.

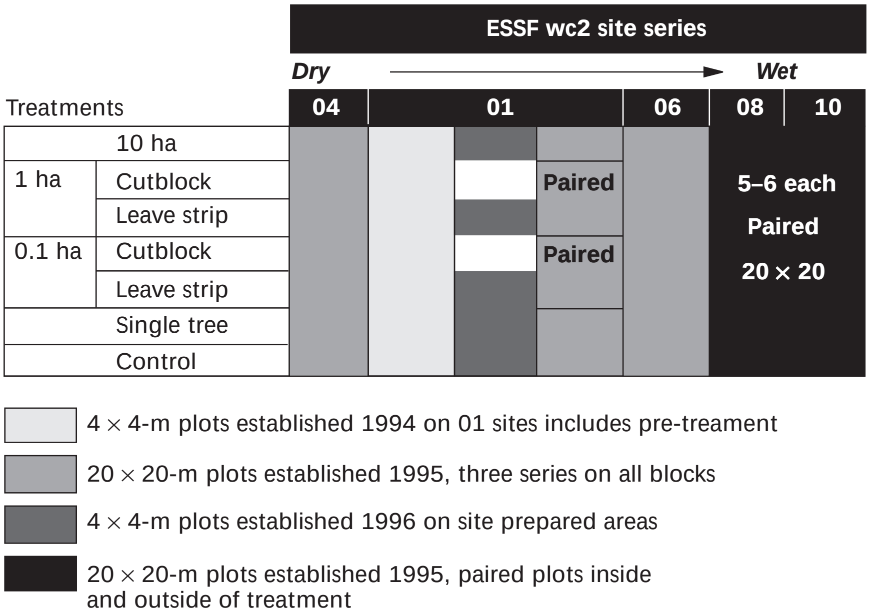

FIGURE 1 *Sicamous vegetation response studies.*

A fourth study examines vegetation response to four site preparation treatments: burning, mounding, scalping to remove the surface organic mat, and an untreated control. Each is replicated three times on each of the five stand opening treatments.

METHODS

Extensive vegetation sampling at the Sicamous research site started in 1995 following logging. For a variety of personnel- and funding-related reasons, pre-treatment sampling was very limited. As a consequence, a heavy reliance will be placed on the controls to determine treatment responses. Sample plots were generally selected in a stratified random fashion. Sample points were rejected if they fell on locations disturbed by other studies or if the location contained a heterogeneous mixture of ESSFwc2 site series.

Two plot sizes have been used: 16 and 400 m^2. All are permanently marked with rebar and their locations mapped relative to permanent, easily recognized tie-points. The 16 m^2 plots are used for three of the four studies. This size provides greater consistency in cover estimates and is suitable for the species diversity and plant community structure at Sicamous. Smaller plots tend to be more expensive to establish, maintain, and monitor because greater replication is required to cover the microsite variation. Therefore, large 20 × 20-m plots have been used to characterize the site series mapped at Sicamous and to document major trends in vegetation response among the five harvesting treatments.

The same standardized methods (Magurran 1988; Luttmerding et al. 1990) have been used to establish and monitor plots for all of the four studies outlined. Plots are sampled at the peak of the growing season, which generally occurs from mid-July to mid-August. Both vascular and non-vascular plants are assessed to determine changes in species composition and percent cover. Percent cover is an estimate of the ground area covered when the species' crowns (leaves and stems) are projected vertically on to the ground. Mean height and phenology are assessed for the vascular plants, and the distribution and abundance of the non-vascular moss, lichen, and liverwort species on different substrates has been noted. Mean height is calculated by averaging the heights proportional to the cover of individual plants. Phenology refers to stages a plant goes through in its life cycle from germination to death in annuals, and from bud break to fruit development in perennials.

To ensure that plant species are accurately and consistently identified now and in the future, voucher specimens are collected for all species. Unknown plants are identified and sent to provincial experts for verification. Simple identification keys for the moss and lichen species found at the research site are being developed to ensure consistent future identification.

Site factors including site series, slope, aspect, and elevation are summarized. Estimates are made of the proportion of the plot occupied by decomposed and intact coarse woody debris, rocks, mineral soil, humus, and water surface substrates. The surrounding stand structure and position relative to standing trees or plots within the uncut leave strips is also noted for each plot.

Rates of plant community development, rates of change, and differences in successional pathways will be determined by sampling vascular plants and dominant non-vascular plants at planned intervals of 1, 2, 3, 5, 10, 20, 30, and 50 years following logging. The complete assemblage of non-vascular plants will be sampled less frequently because of the high monitoring costs.

To assess soil temperature responses, thermocouple wires with recording attachments have been installed at depths of 1 and 15 cm in a subset of the site preparation and edge effect study plots. Point sampling is done four to six times between July 1st and October 1st. Sampling occurs during both the hot sunny and the cool wet seasons. Soil temperature measurements will be used to help interpret seedling, soil, insect, small mammal, and vegetation responses.

Study 1

This study was established in 1994 to examine the effects of five stand-level opening sizes on plant community development. This study is unique at Sicamous because it contains the only permanent vegetation plots with pre-treatment measurements. There are 150 stratified random 4 × 4 m vegetation plots with 30 replicates per harvesting treatment. Sampling is restricted to mesic conditions (the 01 site series) within the mid-elevation replicate block or elevation band used in the broader Sicamous Creek research design. Unfortunately, we were unable to sample all three replicates of each treatment in 1994 because of insufficient funds.

Understorey vegetation in the unlogged leave strips between the 1- and 0.1-ha treatments were also sampled. These plots provide a comparison among leave strips of different widths (30 m, 100 m, and control). They also represent a component of the broader treatments containing different opening sizes. In the 0.1- and 1.0-ha treatments, leave strip plot locations mirrored (in both direction and distance) those inside the adjacent opening. Differences in light, humidity, snow, wind, and temperature regimes should vary among the different uncut strips, as should the vegetation patterns. The results will likely contribute to leave strip width guidelines. These plots have not been subjected to site preparation or planting.

Study 2

In 1995, 125 permanent 20 × 20-m plots were established to compliment the work in study 1. The coverage to all three elevational block replicates was expanded permitting sampling of the five major ecological site series found at the research site. Five sample replicates covering each 01, 04, and 06 site series were located in each of the five opening size treatments. Where possible, at least one replicate for each site series was placed in each of the three elevation bands (blocks) used in the research design. All sample plots were located using a stratified random sampling technique. Candidate plot locations were rejected if they contained a heterogeneous mixture of site series, or if other researchers had planned heavy disturbances such as manipulating coarse woody debris or soil sampling. For the 0.1- and 1-ha treatment units, a pair of plots were established, one inside the cutblock and one in the adjacent stand. The same paired-plot approach was used for the 08 and 10 site series, but sampling occurred in an opportunistic way. Since the occurrence of these wet sites is very limited, a comparison will be made between clearcuts (regardless of opening size) and the adjacent uncut stand. Because of limited budgets and time, the species composition of all vascular and non-vascular species was only sampled on the plots disturbed by logging in 1995. Sampling of all vascular and non-vascular plants is planned for 1996, 1998, 1999, 2004, and 2014.

Twelve additional 20 × 20-m plots were also established in 1996 to support efforts to characterize ecological map units identified for the area, and to provide a more complete species diversity inventory for the study site.

Study 3

This study was initiated during the 1995 field season to evaluate the effect of distance from cutblock edges on vegetation development. At the outset, the objective was to evaluate vegetation response along edges of the 10-ha and 0.1-ha openings. However, because of funding limitations only the north and south edges of the three 10-ha openings were studied. As shown in Figure 2, parallel lines of four permanent 2 × 8-m vegetation plots were established at distances of 0, 0.25, 0.5, 1, and 2 tree heights from the stand edge (0, 7, 14, 28, and 56 m, respectively). Sampling was done at these distances both within the cutblock and the adjacent leave strip. These distances were selected because they correspond to trends in shading and snow accumulation and melt patterns which occur along the northern and southern stand edges. Half of the 216 plots were treated in the fall of 1995 with a small backhoe to remove the organic mat and surface debris and expose the mineral soil for half of the plots in each strip of plots parallel to the edge.

Soil temperature and light will be measured along transects which are perpendicular to the six stand edges where vegetation is being monitored.

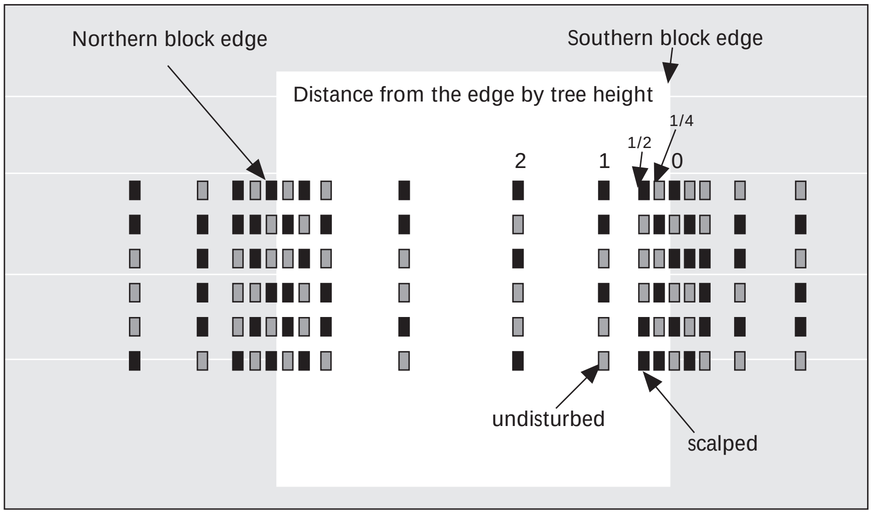

FIGURE 2 *Schematic drawing showing the sampling regime used to study the effects of forest edges on vegetation response.*

Four transects will be established per cutblock edge, with point samples located within the vegetation plots. Half of the sample points will be located on plots with exposed mineral soil.

Study 4 In 1995, four site preparation treatments (scalp, burn, mound, and untreated control) were established in small 30 × 30-m patches within each of the 15 overstorey removal treatment blocks. Within each overstorey treatment, the four site-prepared plots are located close to each other and are restricted to relatively uniform sites representing the 01 site series. All four are replicated once in each of the 15 treatment blocks at the Sicamous research site. Five 4 × 4-m permanent plots have been established within each treatment replicate. The establishment, survival, and growth of natural and planted regeneration will also be assessed in these treatments.

Soil temperature is measured using thermocouple point sensors. Eight probes are installed in each of the site-prepared treatments, four each at 1 and 15 cm below the soil surface. This sampling takes place in the middle of established vegetation plots. To reduce some of the natural variation, probes are not installed close to large organic accumulations or large rocks. Monitoring takes place four to five times between July 1st and October 1st each year.

RESULTS AND OBSERVATIONS

The following results are of a very preliminary nature. These long-term studies were only initiated in 1994 and the analysis of the first-year post-logging data has just started. However, the vegetation data collected to date provides a good baseline against which future changes in composition, cover, and vigour can be compared.

General

Moss and lichen species appear to be most sensitive to environmental changes induced by forest harvesting. Ironically, lichens and bryophytes are a significant component of most forest ecosystems, however very little is known about the effects of forest harvesting on their composition and abundance.

Two species new to science were identified: one lichen and a liverwort (tentatively called *Jungermannia sicamousensis*). It is unlikely that either species is rare, but their discovery reflects the lack of detailed inventories in the ESSF forests of the southern Interior of British Columbia.

Edge Effects Studies

A sharp edge effect on plant community structure is visually apparent within 7 m of the stand edge. This trend is illustrated in Figure 3.

The most profound effect has obviously been the elimination of trees, however, a 67–86% reduction in the abundance of mosses and lichens also occurs within a short distance of the cutblock edge. Moss and lichen cover will be reduced further in 1996 because many species observed in 1995 either had poor vigour or appeared to be dying.

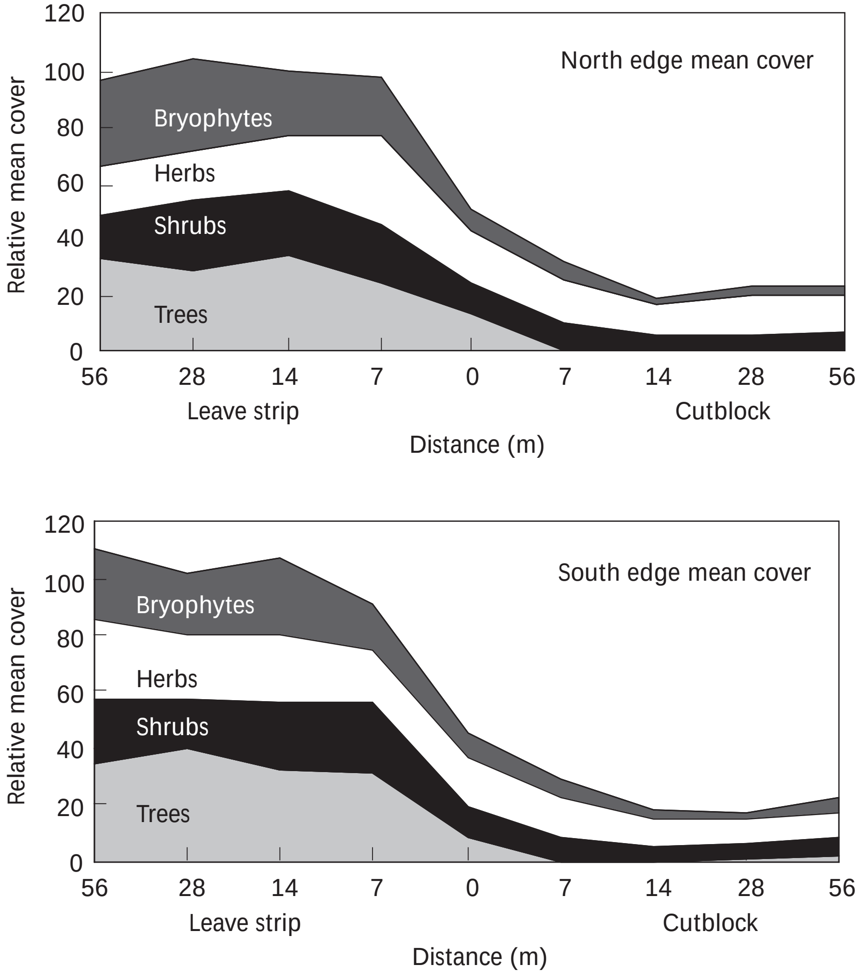

FIGURE 3 *Edge effects on plant community structure.*

First-year post-harvesting results (see Figure 3) show little difference between north and south edges of the block. The post-harvest period has been too short, so this probably reflects the effects of forest harvesting and is not the result of changes in the environmental conditions. Differences along the edge of older cutblocks in the ESSF indicate that subtle differences exist in vegetation development between the north and south edges within 0–15 m of the edge. Continued sampling should permit us to determine both the species, community structure, and biomass production changes, and the rates at which those changes occur.

The abundance of *Rhododendron albiflorum*, *Rubus pedatus*, and *Arnica latifolia* has decreased the most. However, other species including *Valeriana sitchensis* appear less sensitive to forest harvesting.

The mean height of shrubs has also been reduced. This is probably because of logging damage within the cutblock as illustrated in Figure 4.

Response to Five Harvesting Treatments

As shown in Figure 5, the overall cover of shrubs, including *Rhododendron albiflorum*, *Vaccinium membranaceum*, and *Arnica latifolia*, is also reduced in response to the forest harvesting treatments. These results mirror those of the edge effect studies, with the 10- and 1-ha clearcuts exhibiting the most significant reduction in species abundance following treatment. This reflects the increased level of disturbance and accumulation of logging debris which has either killed or buried numerous plants.

Dicranum mosses appear most resistant to the changes. This moss continues to persist on even the most exposed rocks and bedrock in all cutting treatments.

Figure 6 illustrates the effects of different site series on plant community

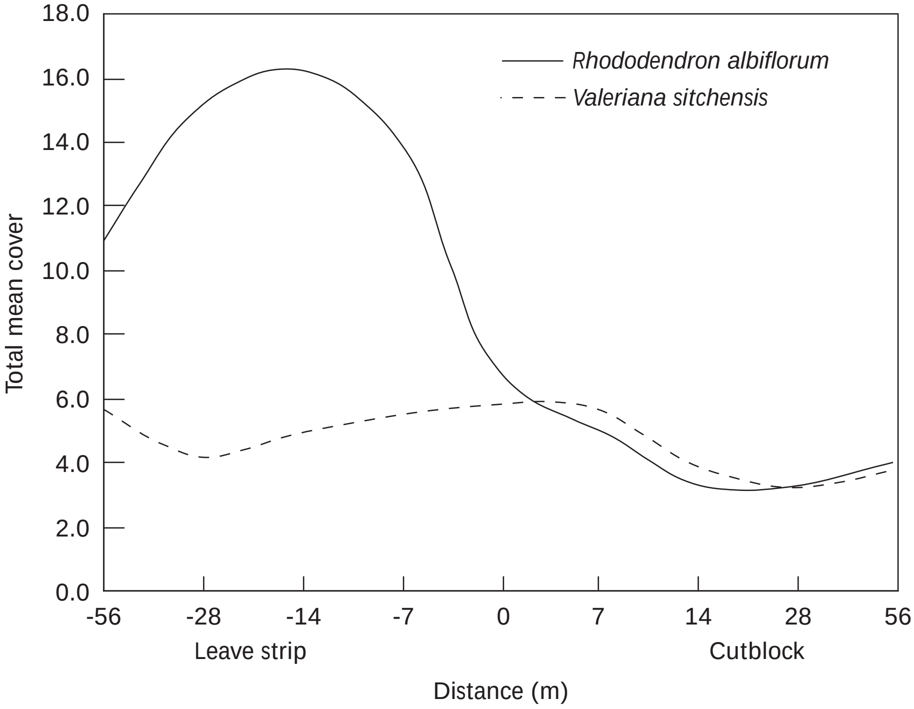

FIGURE 4 *Edge effects on species cover response.*

response. While this response is not surprising, it should demonstrate to research colleagues at Sicamous the importance of stratifying samples and interpreting their results on the basis of site series.

Site Preparation Studies

The burning and scalping treatments have eliminated almost all bryophytes and lichens from treatment plots except on small inadvertently undisturbed patches. Shrubs have been killed back to ground level. Many rhizomotous species such as *Valeriana sitchensis*, *Arnica latifolia*, and *Athyrium filix-femina* have survived these treatments. Their abundance and vigour has been reduced in the first year after disturbance.

Pioneer species including *Epilobium angustifolium*, *Luzula parviflora*, and *Valhodea atropurpurea* have seeded into the burning and scalping treatments in the first growing season after disturbance.

It is anticipated that the burning and scalping treatments will result in the development of a herb-dominated community which differs from the mixed shrub/herb plant community which dominates undisturbed areas.

The numerous microsites created by mounding treatments should result in an increase in plant diversity.

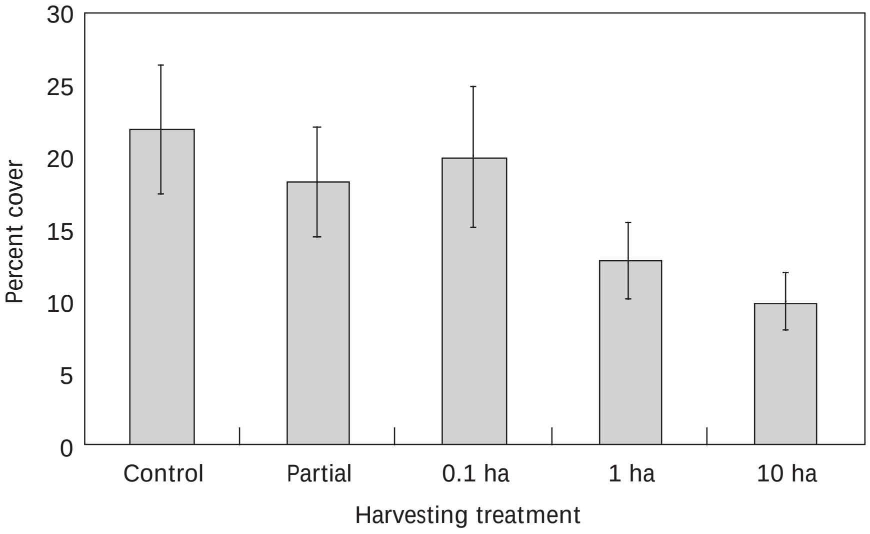

FIGURE 5 *First-year shrub cover by treatment type.*

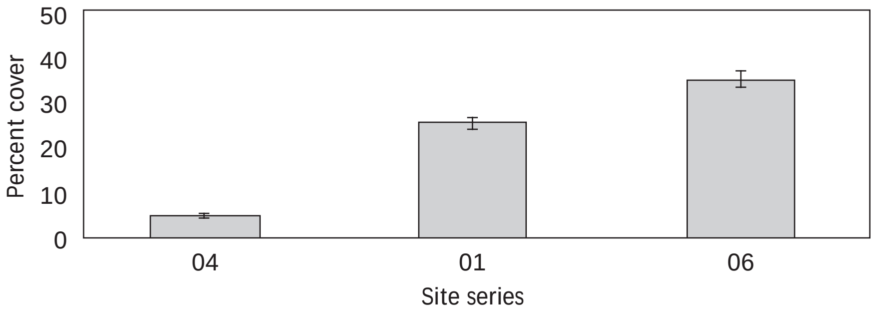

FIGURE 6 *Herb cover by site series for controls.*

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Notes on the Assessment of Lichen Diversity in Old-growth Engelmann Spruce – Subalpine Fir Forests

TREVOR GOWARD AND ANDRE ARSENAULT

ABSTRACT

Qualitative and quantitative methods are used to determine an optimal sampling strategy for assessing and monitoring lichen abundance and distribution in different silvicultural treatments in high-elevation Engelmann spruce – subalpine fir forests near Sicamous, British Columbia. The resulting sampling methods are described in detail, and a list of 99 species reliably identifiable in the field is provided. Based on principal components and cluster analysis, we propose that lichen diversity in the study area may adequately be monitored on the basis of nine substrate units.

INTRODUCTION

Central to the maintenance of lichen diversity in British Columbia is the question whether lichens occur randomly in forests of different ages, or whether some species depend on old-growth forests (Goward 1996). Old-growth dependency among lichens is already well documented in western Europe (Rose 1976; Esseen et al. 1981; Tibell 1991) and eastern North America (Maass 1980; Selva 1994). Indeed, lichens have long been used as indicators of environmental continuity within forest ecosystems. There is now growing evidence that they may serve a similar function in western North America (e.g., Neitlich 1993; Goward 1993, 1994).

Forest ecosystems of the Engelmann Spruce – Subalpine Fir (ESSF) zone of British Columbia have received little attention from lichenologists. Some information is contained incidentally in the floristic studies of McCune (1982), Goward and Ahti (1992), and Debolt and McCune (1993). However, no comprehensive floristic studies have yet been undertaken on the lichens of the ESSF zone, nor is much information available regarding the ecological behaviour of even the more common species within this zone.

Most existing data on the lichens of the ESSF zone has derived from the work of wildlife biologists (e.g., Edwards and Ritcey 1960; Edwards et al. 1960; Stevenson 1979; Palmer 1982; Antifeau 1987; Rominger et al. 1994) studying various epiphytic "forage lichens" (in the genera *Alectoria* and *Bryoria*) that are a primary winter food of the mountain caribou. These and other authors consistently stress the existence of a strong positive correlation between forage lichen biomass and forest age. Field observations suggest a similar correlation may exist between forest age and species diversity, but no published data are available on this subject.

In August 1993, the senior author initiated a detailed study on the lichens of the ESSF zone at Sicamous Creek. This study is intended to reveal the extent to which lichens depend on old-growth ecosystems. It consists of three phases. Phase 1 was initiated during the 1993 and 1994 field seasons. The primary objectives were:

- to document all non-saxicolous woodland lichen species present in the study area; and
- to summarize the local status and distributional ecology of these species.

This work continued through 1995, with the further phase 2 objective of describing lichen floristics and abundance in plots laid out to reflect different silvicultural practices. Phase 3 will begin in 1997 or 1998. Its objective is to monitor the plots at intervals for evidence of disturbance-related changes in lichen floristics and community structure. Simultaneously, other studies will specifically examine substrates strongly associated with old-growth ecosystems, especially large snags, tip-up mounds, and large logs. Over the duration of this project, comprehensive identification keys and detailed species accounts will be prepared; these, together with findings not included in the present paper, will be published at a later date.

The objectives of this paper are primarily methodological:

1. to give a brief account of sampling methods used for phases 2 and 3 of this study;
2. to identify the largest possible subset of the lichen flora that can be reliably sampled in the field; and
3. to determine the minimum number of substrates required for a comprehensive sampling of the lichen flora (i.e., without loss of ecologically significant information).

These points should be of interest to others who wish to investigate lichen floristics and ecology in high-elevation conifer forests.

Study Area

This study is part of the Sicamous Creek Silvicultural Systems Project and is located in the Sicamous Creek research area, approximately 12 km southeast of Sicamous (50°49'N 118°50'W) at an elevation of between 1450 and 1770 m. The forests here belong to the Wet Cold subzone of the Engelmann Spruce –Subalpine Fir zone (ESSFwc2) (Meidinger and Pojar 1991), and are dominated by *Abies lasiocarpa* and *Picea engelmannii* in the overstorey. (See Lloyd and Inselberg [this proceedings, page 79] for a more complete description of the study area.)

Sampling

Owing to problems of scheduling, we were unable to sample the study area prior to silvicultural treatment. Though initially this was of concern to us, we now believe our post-logging assessments have provided an appropriate starting point from which to monitor future changes in lichen community structure: plots sampled prior to logging would have been subject to varied and unpredictable disturbance as a result of cutting, depending on harvesting methods, ground saturation, snow depth, etc.

Sampling was thus initiated four to five months after logging had terminated. During this interval, a few foliose and fruticose lichens had apparently died as a result of disturbance, whereas others were showing signs (e.g., discoloration) of physiological stress. We found no evidence, however, that any species had yet decayed beyond recognition, making it possible to perform a fairly complete inventory of lichen community structure as it would have existed at the time of logging.

Our sampling plots measured 20 × 20 m and were clustered in four silvicultural treatments:

- partial cuts (12 plots);
- 0.1-ha cuts (13 plots);
- 1-ha cuts (15 plots); and
- 10-ha cuts (13 plots).

A similar number of plots will be established in control stands in 1996. In total, 53 plots were assessed between July 14–25 and August 14–27, 1995. Field work was performed by Trevor Goward, with the assistance of David Miège.

Numerous sampling methods have been developed for the quantification of lichen abundance. Recently, McCune and Lesica (1992) evaluated three of these methods: the whole-plot ocular method, the belt transect method, and the micro-plot method. Each method was found to represent a trade-off between species capture and quantitative accuracy. The whole-plot method, for example, yields the most accurate estimate of species richness, but also provides the least accurate estimate of species cover. Notwithstanding this, McCune and Lesica (1992) judged this method to be adequate for detection of at least the most important changes in community structure over time, and recommended its use in studies such as the present one, in which emphasis is given to rare or infrequent species. A coarse sampling approach would seem appropriate given the potential disturbance to which many plots in the Sicamous Creek research area are subject (e.g., by repeated sampling by various researchers), with resulting impacts on some substrates.

Our use of the whole-plot method was intended to allow maximum species capture with a minimum of sampling effort; for most substrates, this method is considerably more efficient than other available methods (McCune and Lesica 1992). In performing the whole-plot ocular method, a thorough reconnaissance is made of a plot of fixed size. Each lichen species within the plot is assigned an abundance class estimate for each substrate on which it occurs. As required, the estimates are gradually revised to reflect improving knowledge of the plot.

In their study, McCune and Lesica (1992) assessed lichen abundance through use of percent cover classes. It may be argued, however, that percent cover per se may not always provide a sensitive measure of ecological adaptedness. In the first place, different lichen species differ in size by at least two orders of magnitude, depending on growth form. It follows from this that percent covers assigned to species having large thalli (e.g., some foliose and fruticose lichens) will far exceed those accorded to species with minute thalli (e.g., some crust lichens), even when these occur in equal abundance. A similar observation can also be made for large fruticose lichens (e.g., *Alectoria sarmentosa)* attached to their substrate at a single point.

And in the second place, many lichen species routinely occur in low abundance, and thus tend to occupy only a minute percentage of any given substrate. As McCune and Lesica (1992) themselves point out, low abundance values are especially difficult to reliably quantify using percent cover, and are generally greatly overestimated. More problematic still is the assessment of percent cover for arboreal habitats, in which the species occupy three-dimensional space (McCune 1990; Stevenson and Enns 1993).

For all these reasons, we prefer in the following frequency and abundance scale to reserve percent cover classes for the assessment of species having moderate to high abundance; species present in low abundance are assigned by us to frequency classes based on actual numbers of thalli present. Embedded in this approach is the assumption that those species present in low abundance in a given site are more likely than other species to experience substantial shifts in abundance as a result of environmental disturbance; their baseline frequency status therefore warrants careful assessment. A similar argument might be made for species having very high abundance. Hence our decision, in the following scale, to reserve four of five frequency units (i.e., units 1, 2, 4, and 5) for species present in notably low or notably high abundance; by far the majority of lichen species would thus generally be accorded a rating of 3.

Frequency unit	**Description**
1	2 or fewer colonies per trunk (and associated branches) for epiphytic species, or per 16m^2 for terricolous species
2	3–5 colonies per tree or per 16 m^2
3	6 colonies as above, or up to 20% cover (under optimum conditions)
4	from 21 to 50% cover (under optimum conditions)
5	51% cover or greater (under optimum conditions)

In interpreting our use of the above scale, four points must be borne in mind:

1. For epiphytic species, ratings are intended to reflect abundance within 2.5 m of the ground; lichens of the middle and upper canopies were not assessed.
2. Our scale is intended to represent lichen occurrence within an area approximately 4 × 4 m, which in our experience is the largest area reliably assessable at mesoscale. Our procedure has thus involved mentally subdividing our 20 × 20-m plots into subplots of appropriate size—a procedure recommended by McCune and Lesica (1992).
3. Ratings were assigned based on lichen performance in those portions of the plot to which a given species appeared to be ecologically most suited. The ratings are thus intended to reflect optimum growing conditions experienced by each species within each plot, as opposed to merely the "average" of the entire plot.
4. Abundance levels for a given substrate in one 4 × 4 m portion of a plot were occasionally found to be more than one suitability unit higher than that expressed elsewhere in the same plot. In such cases, we assigned a whole-plot suitability rating one unit lower than the optimum.

All species of unknown identity were assigned a field name and collected. The specimens were later sorted, curated, and examined in the laboratory using dissecting and compound microscopes, as well as chemical tests. Several crustose specimens were forwarded to various specialists for verification. Voucher specimens will be deposited in the herbaria of the Kamloops Forest Region and the University of British Columbia Department of Botany upon completion of the project.

Definition of Substrates

Lichens are capable of colonizing a wide variety of substrates. To reflect this, our original substrate classification was designed to capture as much ecological information as possible, without, however, overwhelming our sampling methodology. In total, we recognized 21 substrate units (Table 1), each of which was routinely evaluated for lichen abundance. Rock surfaces were excluded from consideration, in order to standardize our assessments for substrates present in all plots.

A snag is defined as a dead standing tree more than 1 m tall. During the summer of 1995, most of the snags present in our plots were felled according to British Columbia Workers' Compensation Board guidelines. A few snags did, however, escape cutting and were assessed as "Bl dead."

A stump, as defined here is a dead standing tree less than 1 m tall, which had died prior to logging. By contrast, the term "cut live" is used for the basal remnant of a live tree felled during logging.

Only a few shrub species were encountered in the study area. The most common was *Rhododendron albiflorum*. Because *Menziesia ferruginea* was rare at Sicamous Creek (it was encountered in only one plot), this species was included with *Rhododendron albiflorum* as "Rhododendron." Likewise, *Vaccinium membranaceum* and *V. ovalifolium* were grouped as "Vaccinium." *Ribes lacustre* and *Lonicera utahensis*—both poor substrates for epiphytes—were excluded from consideration.

TABLE 1 *Definition of 21 substrate units recognized in the Sicamous Creek research area*

Code	Definition
Bl branch	*Abies lasiocarpa* branch
Bl trunk	*Abies* trunk
Bl cut live	*Abies* stump cut as a live tree
Bl dead	*Abies* snag, standing
Bl cut dead	*Abies* stump cut as a snag
Se branch	*Picea engelmannii* branch
Se trunk	*Picea* trunk
Se cut live	*Picea* stump cut as a live tree
Se dead	*Picea* snag, standing
Se cut dead	*Picea* stump cut as a snag
Snag	Dead conifer of uncertain identity
Vacc	*Vaccinium* spp.
Rhodo	*Rhododendron albiflorum* (and *Menziesia ferruginea*)
M logs	Mossy logs present before logging
Logs	Logs present before logging
Up root	Upturned roots, creating tip-up mound
Dec stump	Decayed stump lacking more than half its bark
Moss	Moss on ground
M rock	Mossy rock
Duff	Organic matter on forest floor
Soil	Mineral soil

Our assessments were often complicated by woody debris introduced into the plots as a result of logging. Some plots were further disrupted by the placement of skid roads, as well as by the felling of snags. To maintain sampling consistency among the plots, we assessed only those substrates that would have been present prior to a disturbance. We also excluded all lichen species introduced from the middle and upper canopies by logging activities; such species are unlikely to persist in their new habitats.

Finally, we examined only those trees and shrubs that were actually rooted within the plot perimeter. Portions of trees and shrubs that extended outside the plot boundaries were not considered.

Analysis of Lichen-Substrate Relationships

Lichen community structure and its relationship to substrate was described using ordination methods. Our analysis was restricted to the partial-cut treatments because these supported a more complete assemblage of substrate units than did the clearcut treatments. Principal Components Analysis (pca) was used on an unstandardized co-variance matrix that contained the abundance and frequency indices for 64 species in 178 plot-substrates. We arrived at the latter figure by combining the total number of substrates in all 12 plots examined. Three of the substrate units, however, lacked an appreciable lichen cover: soil, duff, and dead

Engelmann spruce stumps. The first two PCA axes were used to display variation in species composition and to elucidate relationships with substrate units. In addition, the plot-substrate units were classified numerically using the Ward minimum variance algorithm. This procedure allowed us to determine degrees of similarity between the lichen floras of different substrates.

RESULTS AND DISCUSSION

Species Inclusion

One hundred and seventy-six lichens were recorded at the Sicamous Creek research area during this study (Goward et al., in prep.). This flora encompasses 20 species not previously reported from British Columbia, including eight species new to North America, and at least two species new to science.

Not all of these species were found to be reliably identifiable in the field. Most troublesome are certain crustose lichens, especially species in the Caliciales, as well as *Lecanora*, *Lecidea* s. lat., and *Micarea*. Based on comparisons of consistencies and inconsistencies in our use of field names, only 99 species are judged by us to be sufficiently large or otherwise distinctive to permit reliable recognition under a wide range of field lighting conditions. These species are listed in Table 2, which is drawn from as wide an assortment of substrate types as possible. Only species denoted by an asterisk, however, were actually recorded in the plots under discussion; the remaining species are included primarily on the basis of field work conducted in 1993 and 1994.

For routine field assessments of a few taxonomically or morphologically difficult species, we have found it advisable to broaden our concepts to include closely related species having similar ecologies. Thus we list *Bryoria fuscescens* s. lat. (= *B. fuscescens*, *B. glabra*, and *B. lanestris*), *Cladonia ochrochlora* s. lat. (= *C. norvegica* and *C. ochrochlora*), *Cladonia sulphurina* s. lat. (= *C. pleurota* and *C. sulphurina*), *Cladonia symphycarpia* s. lat. (= *C. cariosa* and *C. symphycarpia*), *Mycoblastus sanguinarius* s. lat. (= *M. affinis* and *M. sanguinarius*), *Ochrolechia oregonensis* s. lat. (= *O. oregonensis* and *O. szatalensis*), and *Pertusaria ophthalmiza* s. lat.(= *P.* cf. *multipuncta* and *P. ophthalmiza*). Some sterile specimens recorded as "*Pyrrhospora cinnabarina*" doubtless include the morphologically very similar *Ochrolechia gowardii*.

Lichen Community Structure

The ordination of 178 plot-substrate units is shown in Figure 1. The first pca axis explained 35% of the total variation and clearly separated two distinct lichen communities. The group of substrates located on the right side of the ordination corresponds to standing live and recently dead trees and includes branches and trunks of live *Picea engelmannii* and *Abies lasiocarpa*, as well as snags of *Abies lasiocarpa*. This group is associated with many species that occur primarily over the bark of trees (epiphytes), including species in the genera *Bryoria* and *Hypogymnia*. The left portion of the ordination contains the remaining 14 substrate types, and is dominated by lichens that grow on the forest floor, including species of *Cladonia* and *Peltigera*. Although divisions between lichen associations are

TABLE 2 *List of lichen species reliably identified at the Sicamous Creek research area (asterisks denote species which were found in the 53 vegetation monitoring plots)*

* *Agyrium rufum* (Pers.) Fr.
* *Ahtiana pallidula* (Tuck. ex Riddle) Goward & Thell
* *Alectoria sarmentosa* (Ach.) Ach.
Arthrorhaphis citrinella (Ach.) Poelt
* *Baeomyces rufus* (Hudson) Rebent
Biatora flavopunctata (Tønsberg) Hinteregger & Printzen
* *Bryoria fremontii* (Tuck.) Brodo & D. Hawksw.
* *Bryoria fuscescens* (Gyelnik) Brodo & D. Hawksw. s. lat.
* *Bryoria pseudofuscescens* (Gyelnik) Brodo & D. Hawksw.
* *Calicium glaucellum* Ach.
* *Cetraria chlorophylla* (Willd. *in* Humb.) Vainio
Cetraria ericetorum Opiz subsp. *reticulata* (Räsänen) Kärnefelt
* *Cetraria orbata* (Nyl.) Fink
Cetraria platyphylla Tuck.
* *Cetraria subalpina* Imsh.
* *Chaenotheca furfuracea* (L.) Tibell
* *Chrysothrix candelaris* (L.) J.R. Laundon
* *Cladonia bellidiflora* (Ach.) Schaerer
Cladonia botrytes (K. Hagen) Willd.
* *Cladonia carneola* (Fr.) Fr.
* *Cladonia cenotea* (Ach.) Schaerer
* *Cladonia chlorophaea* (Flörke *ex* Sommerf.) Sprengel
Cladonia cornuta (L.) Hoffm. ssp. *cornuta*
Cladonia crispata (Ach.) Flotow var. *crispata*
* *Cladonia ecmocyna* Leighton ssp. *intermedia* (Robbins) Ahti
* *Cladonia fimbriata* (L.) Fr.
* *Cladonia gracilis* (L.) Willd. ssp. *turbinata* (Ach.) Ahti
* *Cladonia macilenta* Hoffm.
Cladonia macrophyllodes Nyl.
* *Cladonia merochlorophaea* Asah.
Cladonia multiformis G. Merr.
* *Cladonia ochrochlora* Flörke s. lat.
Cladonia pyxidata (L.) Hoffm.
* *Cladonia sulphurina* (Michaux) Fr. s. lat.
Cladonia symphycarpia (Flörke) Fr. s. lat.
* *Cliostomum* sp. nov.
Esslingeriana idahoensis (Essl.) Hale & M.J. Lai
Fuscopannaria mediterranea (Tav.) P.M. Jørg.
* *Hypogymnia austerodes* (Nyl.) Räsänen
* *Hypogymnia imshaugii* Krog
* *Hypogymnia metaphysodes* (Asah.) Rass.
* *Hypogymnia occidentalis* L. Pike
* *Hypogymnia physodes* (L.) Nyl.
* *Hypogymnia rugosa* (G. Merr.) L. Pike
* *Hypogymnia tubulosa* (Schaerer) Hav.
* *Icmadophila ericetorum* (L.) Zahlbr.
Kaernefeltia merrillii (Du Rietz) Thell & Goward
* *Lecanora circumborealis* Brodo & Vitik.
Lepraria cacuminum (Massal.) Lothander
* *Lepraria jackii* Tønsberg
* *Letharia vulpina* (L.) Hue
Lopadium disciforme (Flotow) Kullhem
Massalongia carnosa (Dickson) Körber
* *Melanelia exasperatula* (Nyl.) Essl.
Melanelia subelegantula (Essl.) Essl.
* *Mycoblastus sanguinarius* (L.) Norman s. lat.
* *Nephroma arcticum* (L.) Ach.
* *Nephroma bellum* (Sprengel) Tuck.
Nephroma parile (Ach.) Ach.
Nephroma resupinatum (L.) Ach.
Nodobryoria abbreviata (Müll. Arg.) Common & Brodo
Nodobryoria oregana (Tuck.) Common & Brodo
* *Ochrolechia oregonensis* H. Magn. s. lat.
* *Pannaria pezizoides* (G.H. Weber) Trevisan
* *Parmelia hygrophila* Goward & Ahti
* *Parmelia sulcata* Taylor
* *Parmeliopsis ambigua* (Wulfen *in* Jacq.) Nyl.
* *Parmeliopsis hyperopta* (Ach.) Arnold
* *Peltigera aphthosa* (L.) Willd.
Peltigera britannica (Gyelnik) Holt.-Hartw. & Tønsberg
Peltigera canina (L.) Willd.
* *Peltigera chionophila* Goward, ined.
Peltigera cinnamomea Goward
Peltigera degenii Gyelnik
Peltigera didactyla (With.) J.R. Laundon
Peltigera kristinssonii Vitik.
Peltigera leucophlebia (Nyl.) Gyelnik
Peltigera malacea (Ach.) Funck
* *Peltigera membranacea* (Ach.) Nyl.
* *Peltigera neopolydactyla* (Gyelnik) Gyelnik
Peltigera occidentalis (E. Dahl) Kristinsson
* *Peltigera polydactylon* (Necker) Hoffm.
Peltigera ponojensis Gyelnik
Peltigera praetextata (Flörke *ex* Sommerf.) Zopf
* *Peltigera scabrosa* Th. Fr.
Peltigera venosa (L.) Hoffm.
* *Pertusaria ophthalmiza* (Nyl.) Nyl. s. lat.
* *Platismatia glauca* (L.) Culb. & C. Culb.
* *Psoroma hypnorum* (Vahl) S. Gray
* *Pyrrhospora cinnabarina* (Sommerf.) Choisy
* *Solorina crocea* (L.) Ach.
* *Stereocaulon alpinum* Laurer *ex* Funck
Stereocaulon tomentosum Fr.
Thrombium epigaeum (Pers.) Wallr.
Trapeliopsis granulosa (Hoffm.) Lumbsch
* *Varicellaria rhodocarpa* (Körber) Th. Fr.
Vulpicida pinastri (Scop.) J.-E. Mattsson & M.J. Lai
* *Xylographa vitiligo* (Ach.) J.R. Laundon

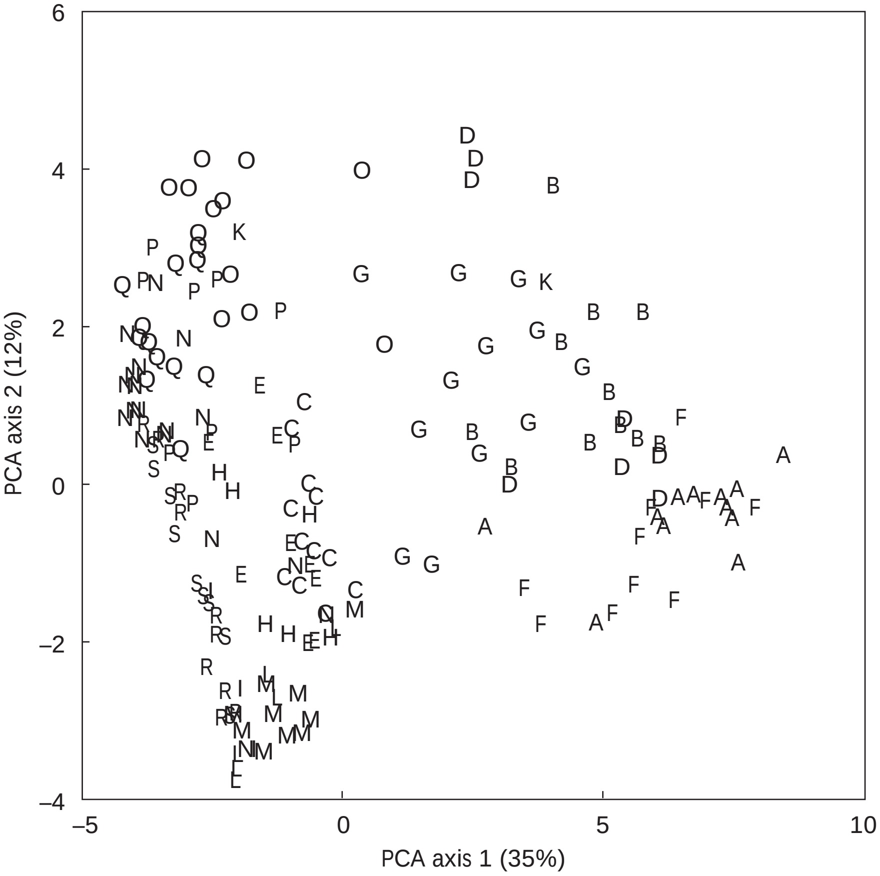

FIGURE 1 *Principal components analysis ordination of 178 plot-substrate units. Substrates are denoted by letter according to the following: A = Bl branch, B = Bl trunk, C = Bl cut live, D = Bl snag, E = Bl cut dead, F = Se branch, G = se trunk, H = Se cut live, I = Se snag, J = Se cut dead, K = Snag, L = Vacc, M = Rhodo, N = M logs, O = Logs, P = Up root, Q = Dec stump, R = Moss, S = M rock.*

not as clear along the second pca axis, several main groups emerge. Similar assemblages of lichens are found on live *Picea* and *Abies* as shown by the overlap of substrate units associated with these two tree species. However branch and trunk substrates occupy distinct positions along the second axis and represent different lichen communities. Species on subalpine fir snags spread across both groups, apparently because they included samples both on trunks and on branches. It is also possible to divide the main group on the left side into five species subgroups along the second axis which includes:

1. species growing on logs,
2. species growing on decaying stumps,
3. species growing on mossy substrate,
4. species growing on freshly cut stumps, and
5. epiphytes growing on shrubs.

The classification of plot-substrate units accords well with the ordination, and suggests that several substrates may be combined without losing important ecological information (Table 3). Ninety percent of *Picea engelmannii* and *Abies lasiocarpa* branches, for example, were classified as belonging in the same group. The trunks of these two species were also largely classified into one group. Similarly, *Vaccinium* and *Rhododendron* may be combined as a shrub substrate, while mossy logs, mossy rocks, and decaying stumps may likewise be merged. Finally lichen communities occurring on stumps that arise from freshly cut live or dead trees are sufficiently similar to justify their placement in a single substrate unit. The remaining substrates do not classify well and should therefore be sampled separately. This scheme allows the original 21 substrates to be reduced to nine: conifer branch, conifer trunk, snag, hard stump, hard log, tip-up mounds, elevated mossy substrates, soil, and duff. Given that each plot sampled in 1995 required between 2.5 and 4 hours of field time, such a reduction in substrate units, and therefore in sampling effort, seems desirable. This revised methodology will be adopted for the 1996 field season.

TABLE 3 *Relationship between groups classified using Ward minimum variance cluster algorithm and substrate types. Percentages of plot-substrate types classified in the ten clusters are shown.*

	Cluster no.										
Substrates	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	***N*[a]**
Bl branch	92.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.0	12.0
Se branch	90.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.0
Bl trunk	0.0	92.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.0	12.0
Se trunk	0.0	42.0	16.5	0.0	0.0	0.0	0.0	0.0	16.5	25.0	12.0
Rhodo	0.0	0.0	17.0	67.0	0.0	0.0	8.0	8.0	0.0	0.0	12.0
Vacc	0.0	0.0	9.0	73.0	0.0	0.0	9.0	9.0	0.0	0.0	11.0
Bl cut live	0.0	0.0	67.0	0.0	0.0	0.0	8.0	0.0	25.0	0.0	12.0
Se cut live	0.0	0.0	33.0	0.0	0.0	0.0	67.0	0.0	0.0	0.0	6.0
Bl cut dead	0.0	0.0	67.0	0.0	0.0	0.0	22.0	0.0	11.0	0.0	9.0
Bl snag	13.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	37.0	8.0
Se snag	0.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0	0.0	0.0	2.0
Snag	0.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0	50.0	0.0	2.0
Dec stump	0.0	0.0	0.0	0.0	58.3	33.3	8.3	0.0	0.0	0.0	12.0
Logs	0.0	0.0	0.0	0.0	0.0	75.0	0.0	0.0	25.0	0.0	12.0
Up root	0.0	0.0	0.0	0.0	22.0	33.0	11.0	0.0	33.0	0.0	9.0
M log	0.0	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0	0.0	12.0
M rock	0.0	0.0	0.0	0.0	67.0	0.0	0.0	33.0	0.0	0.0	9.0
Moss	0.0	0.0	0.0	0.0	27.0	0.0	0.0	73.0	0.0	0.0	11.0

[a] *N* is the number of plot-substrate units associated with each substrate.

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Ecosystem Mapping for the Sicamous Creek Silvicultural Systems Research Site

DENNIS LLOYD AND ALEX INSELBERG

ABSTRACT

The purpose of ecosystem mapping is to provide a visual summary of the distribution and extent of ecosystems. The procedure used to complete the mapping at Sicamous is described briefly. A very preliminary map representing about 50% of the study area is presented. Mapping of the entire study area should be completed during the summer of 1996. The completed map and an accompanying report, which outlines the methods and detailed descriptions of the mapped polygons, will be available in 1997. A preliminary summary is presented to provide the reader with a generalized description of the ecological characteristics of the Sicamous research site. This summary also provides an ecological context within which the results of studies by other Sicamous research co-operators can be considered in other parts of British Columbia.

INTRODUCTION

Ecosystem mapping stratifies the landscape into relatively uniform segments which are characterized by a distinctive climate, physiography, soils, and vegetation (Mitchell et al. 1989). Reconnaissance surveys of the Sicamous research site completed by G. Hope and F. Russell in 1994 resulted in a very broad-scale map that was used to stratify the research area into three elevational blocks for statistical purposes. Their data also confirmed that the Northern Monashee wet cold Engelmann Spruce – Subalpine fir biogeoclimatic variant (ESSFwc2) site series described by Lloyd et al. (1990) was a suitable basis for classifying and mapping the area. Biogeoclimatic classification is complete for the entire province and is widely used as a framework for developing and presenting resource management guidelines (Lloyd et al. 1990; Braumandl and Curran 1992; Banner et al. 1993; Green and Klinka 1994). Linkage of research results to specific ecological classification units provides the basis for extrapolating these research results beyond the study site.

The Sicamous research site is located in the ESSFwc2 biogeoclimatic variant. The ESSFwc subzone occupies 1 876 000 ha or 14% of the area covered by provincial ESSF. It is the most widespread ESSF subzone in the southern interior. It occurs extensively in the Nelson, Kamloops, and

Cariboo forest regions. Based on generalizations about the ESSFwc2 reported in Lloyd et al. (1990), the Sicamous area is estimated to receive 1000 mm of precipitation annually. The snowpack accumulates to about 2.5 m, depending on aspect and elevation, and generally melts by mid- to late June. The mean annual temperature is 1°C, and the continuous frost-free period amounts to less than 40 days between mid-July and mid-August.

The research site is located on the Shuswap Highlands Ecosection (Demarchi 1996). It occurs in the interior wetbelt where high levels of precipitation result in a reduced natural fire frequency and lower-elevation stands of *Thuja plicata* and *Tsuga heterophylla*. The site occupies a north aspect with an average slope gradient of 20–40%. The soils are derived mostly from morainal deposits laid down during the last glacial period. Soils are primarily Humo-Ferric Podzols with a discontinuous Ae layer and a Hemihumimor humus form. The soil texture varies, but is predominantly a sandy loam with 25–40% coarse fragment content. The underlying bedrock is primarily granitic gneiss.

The forest cover type is predominantly an old-growth subalpine fir – Engelmann spruce stand. The understorey is dominated by a shrub and herb plant community. The main shrubs are *Rhododendron albiflorum*, *Vaccinium membranaceum*, and *Vaccinium ovalifolium* and the herb layer is dominated by *Valeriana sitchensis*, *Arnica latifolia*, *Gymnocarpium dryopteris*, *Tiarella unifoliata*, and *Streptopus roseus*.

The mapping completed to date has been used by researchers to stratify samples and ensure that the same ecosystem type is compared in various treatment effect studies. The ecological classification provides a basis for interpreting research data and should help us understand the interactions among the ecosystem components that are studied.

OBJECTIVES

The purpose of ecological mapping at the Sicamous research site is to:

1. provide a detailed visual summary of the distribution and extent of easily recognized ecological site conditions, referred to as the "site series"; and
2. provide data to briefly describe each site series which represents the range of biophysical conditions in the study area.

METHODS

The landscape at the Sicamous research site is covered by a complex arrangement of six ESSFwc2 site series. This complexity is attributed to variations in slope, aspect, slope position, soil depth, soil drainage, and snowmelt, and precipitation runoff patterns. Large-scale mapping, which recognizes the presence and distribution of small patches or strings of individual site series, was required to produce an ecosystem map that would be useful to researchers.

The area was mapped through a detailed ground survey that covered the entire research site. This approach was a contrast to the more typical site series mapping which relies heavily on the interpretation of air photos to identify map boundaries and polygon contents. A 50 × 50-m grid was established over the area using a compass and tape. Then, each grid cell received a systematic on-site examination to determine the location of site series boundaries. Where possible, map boundaries were also tied to landscape features such as wetlands, rock outcrops, natural openings, and disturbances such as roads and cutblocks which readily appear on 1:5000 air photos. The smallest map unit represents about 0.125–0.063 ha, which is equivalent to one-quarter to one-half of the 50 × 50-m ground-based cell. In the future, global positioning system (GPS) technology should be available to transfer the site series boundaries observed on the ground and therefore eliminate the need for gridding an area before mapping. However, the current level of accuracy and cost makes this prohibitive.

One-quarter of the research site was mapped in 1994 before logging. Mapping focused on the 0.1-, 1-, and 10-ha cutblocks to capture plant communities in an undisturbed state. An additional 25% of the area was mapped in 1995 and mapping of the remaining 50% is planned for 1996.

Field sampling of 124 20 × 20-m plots was completed to characterize the vegetation, site, and soils of mapped site series. The six site series recognized at Sicamous are distinguished on the following basis.

- 02 sites are distinguished by their shallow soils, an understorey dominated by *Vaccinium membranaceum* with a high cover of mosses and lichens and a general lack of herbs (usually less than 5% cover).
- 04 sites are dominated by a shrub and moss plant community composed of *Rhododendron albiflorum*, *Vaccinium membranaceum*, and an understorey dominated by mosses and 5–25% cover of herbs. These sites lack *Gymnocarpium dryopteris* and have little or no *Valeriana sitchensis*.
- 01 sites have moderately well-drained soils and a mixed shrub- and herb-dominated understorey. These sites are distinguished from the 04 site series by the presence of *Gymnocarpium dryopteris* and more than a 2% cover of *Valeriana sitchensis*.
- 06 sites have a herb- and shrub-dominated understorey with imperfectly drained soils—a fluctuating water table lies within 75 cm of the soil surface. The surface mineral horizon is often an organically enriched Ah. The combined presence and abundance of *Athyrium filix-femina*, *Lonicera involucrata*, a high cover of *Ribes lacustre*, and an overall dominance of herbs in the plant community distinguishes the 06 from the 01 site series.
- 08 sites are distinguished by a plant community dominated by *Equisetum arvense*, *Equisetum sylvaticum*, *Calamagrostis canadensis*, and *Senecio triangularis*. *Platanthera hyperborea*, *Parnassia fimbriata*, *Trollius laxus*, and *Leptarrhena pyrolifolia* are also common. The water table is within 50 cm of the soil surface; the soils are gleysols and generally contain a well-developed, organically enriched surface horizon.
- 10 sites have very poorly drained organic soils with a water table at or near the surface for the entire growing season. The vegetation is dominated by graminoids; including many *Carex* and *Juncus* species.

Willows, bog orchids, cotton-grass, and horsetails are also common. This site series could be split into two or more new units, one of which is easily recognized by the presence of *Potentilla fruticosa* and a high cover of willows.

RESULTS

A map showing the distribution and extent of the six site series on the Sicamous site is presented in Figure 1. The 01, 04, and 06 site series are equally widespread and most dominant across the area. The 04 unit is more widespread in the upper-elevation block at the research site, while the 06 occurs more commonly on toe slopes, gullies, and lower slopes which are most common in the lower-elevation block. The 08 and 10 are restricted to small areas where soil moisture tends to accumulate. Figure 2 provides a tabular summary of the ecological characteristics of the six mapped site series.

DISCUSSION

The 1996 field sampling and mapping will provide a more localized description of the ESSFwc2 conditions at the Sicamous research site. A complete vegetation list for all species found on the site will also be prepared. Two new site series, both with a very limited distribution, will likely be recognized. Future data collection will concentrate on characterizing the soil moisture regime, nutrient status, coarse woody debris content, soil temperature regime, and light conditions within each site series. This information should help researchers develop and refine resource management interpretations. Mapped units will be placed on geographic information systems (GIS) so that area summaries may be calculated.

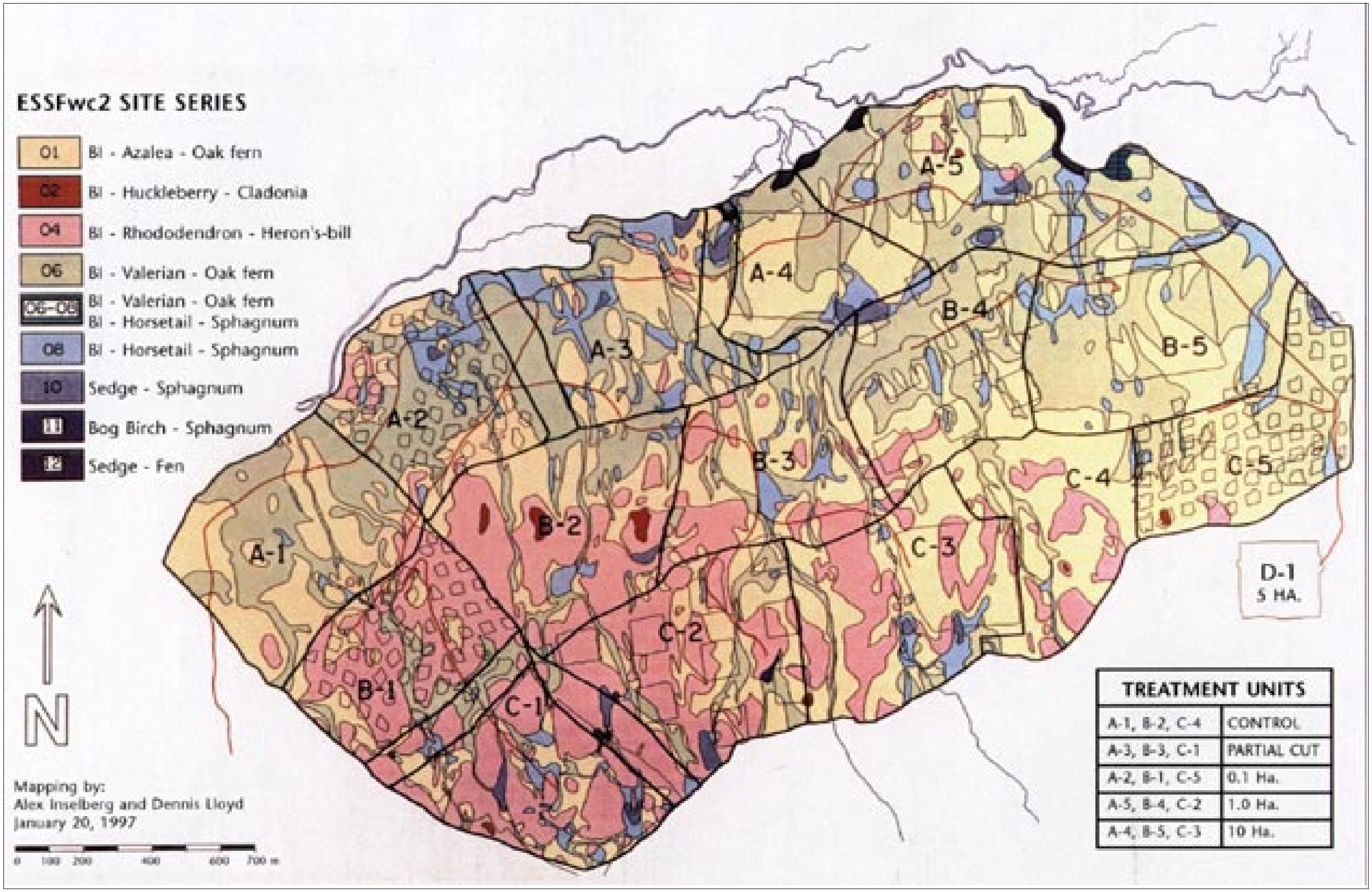

FIGURE 1 *Biogeoclimatic site series map for the Sicamous Creek research area.*

		Site series					
		02	04	01	06	08	10
Site features	Soil moisture regime	Very xeric to subxeric	Subxeric and submesic	Mesic and submesic	Subhygric and (mesic)	Hygric	Hygric to subhygric
	Soil drainage	Very rapid	Well to rapid	Well and moderately well	Imperfect	Poor	Very poor
	Slope	5–60%			5–20%	0–5%	Flat
	Aspect	Primarily north				N/A	N/A
	Slope position	Upper and crest	Mid- to upper	Mid- and level	Level, lower, and toe	Level and depressions	Level and depressions
	Parent material	Morainal veneers	Morainal blankets	Morainal blankets	Morainal blankets and fluvial blankets	Fluvial blanket and fluvial veneers over morainal blankets	Organic blankets
	Soil development	Humo-Feric Podzols			Gleyed Podzols	Gleysols	Organics
	Humus form	Hemimors				Hydromulls	Histomors
Vegetation	**Trees**						
	Abies lasiocarpa	***	***	***	**	*	
	Picea engelmannii	**	**	**	*	*	
	Shrubs						
	Rhododendron albiflorum	+	***	***	*	*	
	Vaccinium membranaceum	***	***	**	*	*	
	Vaccinium ovalifolium		*	**	**		
	Ribes lacustre			+	**		
	Lonicera involucrata				*	+	
	Salix spp.						*
	Herbs						
	Lycopodium annotinum	+	+	+			
	Rubus pedatus	*	**	*	*	*	
	Arnica latifolia		*	**	*		
	Mitella breweri		+	+	*		
	Streptopus roseus			*	*	*	
	Tiarella unifoliata		+	*	*	*	
	Gymnocarpium dryopteris			*	**		
	Valeriana sitchensis			**	***	**	+
	Veratrum viride				*	*	+
	Calamagrostis canadensis				+	*	*
	Athyrium filix-femina				*	+	
	Equisetum arvense				+	**	+
	Carex spp.					*	**
	Senecio triangularis					**	*
	Equisetum sylvaticum					*	+
	Mosses						
	Dicranum spp.	***	***	*	*		
	Pleurozium schreberi	**	*	*	+		
	Rhytidiopsis robusta	*	**	*			
	Brachythecium spp.		+	*	*		
	Mnium spp.			*	**	+	
	Sphagnum spp.					+	**

Presence and abundance codes: + may be present; * often present in low abundance; ** usually present in moderate abundance; *** generally present in high abundance.

FIGURE 2 *A summary of the biophsyical characteristics of the site series located at the Sicamous research site.*

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Effects of Silvicultural Systems on Soil Productivity

GRAEME HOPE

ABSTRACT

Soil projects at the Sicamous Creek Silvicultural Systems study area were established with two major aims: first, to gain an understanding of the processes controlling long-term soil productivity in the ESSF zone, with a focus on soil biological properties and soil organic matter; and second, to understand the effects of canopy disturbance on those processes. A simple conceptual model illustrates how the studies fit within an overall research strategy. Preliminary sampling of soil chemical properties at Sicamous Creek indicates that the soils have typical characteristics of ESSF forest areas in the Kamloops Forest Region. The soils have a medium nutrient status and are not strongly leached.

INTRODUCTION

In the early stages of a forest rotation, the most successful plantations in the Engelmann Spruce – Subalpine Fir (ESSF) biogeoclimatic zone of the Kamloops Forest Region appear in areas with the heaviest soil disturbance or most severe prescribed burns (Mather 1987). Over the longer term, demands by trees on the soil increase, and soil properties that exist after the disturbances of harvesting and site preparation may not be sufficient to maintain productivity. The focus of soils research at Sicamous Creek is twofold: (1) to understand the effects of disturbance on long-term soil productivity, and (2) to use this knowledge to sustain soil productivity.

The key soil properties in relation to long-term productivity that may be altered by any silvicultural system are soil organic matter and soil porosity (Powers et al. 1990). Soil organic matter, not porosity, is primarily affected in the ESSF zone because much of the harvesting here is done on a considerable snowpack.

Many of the soil processes involving soil organic matter are mediated by soil biological activity. Nitrogen release from organic matter, asymbiotic nitrogen fixation, and soil food web structure and function, from bacteria and fungi to nematodes and micro-arthropods, are all affected by organic matter changes. Evidence from other high-elevation *Abies* forests shows that decomposition of the forest floor, soil wood[1], and fine roots is relatively more important in maintaining forest productivity than in lower-elevation forests (Vogt et al. 1989).

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[1] Soil wood is primarily old decaying logs within the forest floor.

Sollins et al. (1983) suggest that the first step in predicting the long-term consequences of management practices is to develop a conceptual model of soil processes. Powers et al. (1990) proposed a general conceptual model of soil productivity. This may need some modification for the ESSF zone. Frost frequencies are high and soil temperatures are low throughout most of the zone, whereas lack of soil moisture appears to control soil productivity only on the driest sites. Figure 1 is a modification of the model of Powers et al. It incorporates more specific soil biological and soil heat features than the original model.

Several soil-related projects that will investigate specific parts of the soil process model in Figure 1 are being established at Sicamous Creek. These projects are summarized in Table 1. In addition, all studies will rely on microclimate data and interpretations provided by the microclimate studies at Sicamous Creek (see Novak et al., this proceedings, page 45). The combined results of all studies should allow the formulation of more comprehensive organic matter and nutrient models than would be possible from a single soil study. Such a process has given valuable information on soil and ecosystem function in other studies (Gosz et al. 1976).

Many of the soils projects will investigate the effects of canopy disturbance. Any harvesting system involves some loss of organic matter and site nutrients, regardless of opening size and level of forest floor removal. The patterns of change in organic matter and nutrient dynamics following disturbance in large openings are well known in semi-quantitative terms, but with opening sizes of less than 1 ha, little is known (Vitousek 1985). Rather than examine average soil properties in the opening, the quantification of "edge effects" will be the focus of many of the soil studies.

Initial results from some of the soil studies at Sicamous are presented elsewhere in this proceedings. The rest of this paper presents a summary of soil and site conditions at the Sicamous Creek Silvicultural Systems Project research area.

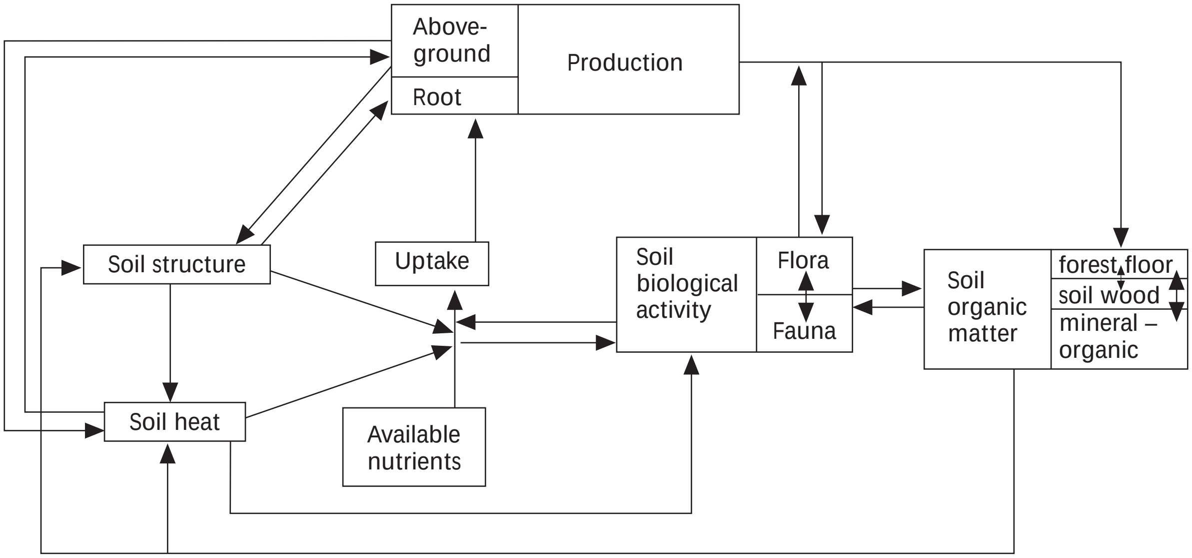

FIGURE 1 *Conceptual model of soil processes controlling productivity in a wet ESSF forest.*

The Sicamous Creek site consists predominantly of moderate (20–40%) north-facing slopes, with more gentle slopes in the valley bottom and at the upper, south end of the site. Surficial materials are of morainal origin on most of the slopes. The depth of material varies from less than 50 cm on bedrock-controlled areas to greater than 1 m on the majority of the slopes. These morainal materials are derived from coarse-grained

TABLE 1 *Summary of soil and nutrient studies at Sicamous Creek*

Study leader(s)	Study focus	Sampling locations	Parameters measured	End products
07 Hope and Prescott	Organic matter and nitrogen dynamics	All treatments: mineralization and decomposition transect in one 10-ha block	Soil C, N and S; N mineralization; litter decomposition; N uptake by seedlings	Change in soil nutrient content; mineralization and decomposition rates; long-term C and N predictions
08 Nadel	Soil micro-arthropods	Control, 0.1 1.0 and 10 ha, plus transect in one 10-ha block	Mites and collembola: numbers and diversity	Abundance of mites and collembola in treatments; diversity to species level
09 Jones and Durall	Ectomycorrhizal diversity and hypogeous fungal sporocarps	Control, 0.1, 1.0, and 10 ha, plus transects in three 10-ha blocks	Type of live ectomycorrhizal root tips; soil bioassays; sporocarp biomass and identification	Number and diversity of ectomycorrhizal types; biomass and diversity of sporocarps
11 Feller	Nutrient budgets; nitrogen fixation	One control and one 10-ha block	Nutrients in precipitation and soil leachate; nitrogen fixation and denitrification	Nutrient budget, gaseous N fluxes
12 Hunt	Fine roots	Control, 0.1, 1.0, and 10 ha, plus transect on one 10-ha block	Fine-root biomass and nutrient content; fine-root decomposition	Nutrient capital and fluxes from fine roots
25 Prescott	Litter and coarse woody debris inputs	Control	Annual litter and dead wood inputs	Rates of input
33 Hope and Johnson	Soil food webs and nitrogen dynamics	Control, 0.1 ha and 10 ha	Total soil bacteria, fungi, and nematodes; N mineralization and denitrification	Food web biomass, nitrogen mineralization, and denitrification in the root zone

metamorphic bedrock. A veneer of silty fluvial materials covers many of these deposits on the gentle lower slopes. Organic deposits occur in small wetland areas.

Soils on the site are predominantly sandy loam textured Orthic Humo–Ferric Podzols (Agriculture Canada 1987). These soils occur on the ESSFwc2/01 and 04 site series (Lloyd, this proceedings, page 83). On wetter site series, soils are Gleyed Sombric Brunisols and Orthic Humic Gleysols.

Forest floors are thin throughout the site (Table 2). Humus forms are predominantly Hemimors. Forest floors contain approximately 10–30% decayed wood, and also contain an intermittent, thin charcoal layer at the mineral soil–forest floor interface.

TABLE 2 *Site and soil characteristics at Sicamous Creek*

	Site series unit		
	ESSFwc2/01	**ESSFwc2/04**	**ESSFwc2/06**
Slope position	midslope	upper, rocky slopes	lower and seepage areas
Soil moisture regime	4 (mesic–subhygric)	3–2 (submesic)	5 (subhygric)
Soil nutrient regime	poor	poor	medium
Landform	glacial till	glacial veneer over rock	fluvial veneer over till
Soil classification	Orthic Humo-Ferric Podzols	Orthic Humo-Ferric Podzols	Gleyed Sombric Brunisols
Soil texture	sandy loam	sandy loam–loamy sand	silt loam over sandy loam
Humus form Thickness (range in cm)	Hemimor 4.0 (1.5–9)	Hemimor 3.6 (1–12)	Mor and Moder 5.0 (1–14)
Soil drainage	well	well to rapid	imperfect (to poor)
Sensitivity to:			
Compaction	moderate	low to moderate	high to very high
Surface erosion	moderate to high	moderate	high
Forest floor displacement	high	high	low
Soil displacement	moderate	high	low
Mass wasting	moderate	moderate	moderate

METHODS

Soil samples were collected from representative locations of the major site series at Sicamous Creek in August 1993. At two locations of each site series, seven samples were collected from the forest floor and a 0–20 cm depth in the mineral soil. Samples were air-dried, sieved through a 2-mm sieve, and analyzed at the B.C. Ministry of Forests Laboratory, Victoria. Standard methods of analysis were used: pH, in water (forest floor), or $CaCl_2$ (mineral soil); carbon, nitrogen, and sulphur with a Leco analyzer, cation exchange capacity (CEC) in 1 N ammonium acetate; and anaerobic incubation for mineralizable nitrogen.

RESULTS AND DISCUSSION

Soil chemical property means from the preliminary soil sampling are summarized in Table 3. No detailed statistical analysis was performed on these samples. More detailed sampling will be carried out for studies of carbon and nitrogen dynamics.

TABLE 3 *Soil chemical properties at Sicamous Creek*

		Soil chemical property					
Sample	**Site series**	**Carbon (%)**	**Sulphur (%)**	**Nitrogen (%)**	**Mineralizable N (ppm)**	**pH**	**CEC (cmol/kg)**
Mineral soil	01	3.8	0.02	0.19	46	4.3	19.9
(0–20 cm)	04	5.4	0.03	0.25	30	3.8	26.1
	06	6.2	0.05	0.37	65	4.2	29.3
Forest floor	01	42.8	0.14	1.39	532	4.9	79
	04	49.4	0.16	1.66	655	4.2	78
	06	41.7	0.17	1.58	452	4.7	84

The preliminary results support what is known about ESSF soils. The soils are not extremely acid, and exchangeable cation status is reasonable. Carbon:nitrogen ratios, nitrogen contents, and mineralizable nitrogen are moderate. Sulphur contents are typical of soils in the interior of British Columbia. The forest floor is more nutrient-rich than the mineral soil, and differences in nutrient concentrations on different site series are small. When these results are compared with those from similar soils in the Interior Cedar Hemlock (ICH) zone below Sicamous Creek (G. Hope, unpublished data), measures of nutrient status are similar for ESSF and ICH soils, even though soil productivity appears to be higher in the ICH. Therefore, the importance of soil organic matter and rates of nutrient cycling in the ESSF zone should be clarified as should the relationship between measures of nutrient availability and actual nutrient uptake. Future work will involve more detailed sampling, allowing more definite conclusions about soil properties.

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Sicamous Creek Silvicultural Systems Trial: Litter Input, Decomposition, and Nitrogen Mineralization

CINDY PRESCOTT AND GRAEME HOPE

ABSTRACT

Annual aboveground litter input was measured in 0.25-ha plots in the three old-growth forests between September 1995 and 1996. Fine litter, collected in wooden trays totalled 149 g/m^2, 87% of which was needles. No trees fell in the old-growth plots during this time. Samples of spruce-fir needle litter enclosed in fibreglass mesh bags were installed in plots of each opening size and at nine stations across the 10-ha clearcut. Decomposition of needle litter, as well as pine needles, aspen leaves, spruce and fir roots, and forest floor material will be measured at each location for five years.

Rates of nitrogen mineralization in forest floor and mineral soil were measured at nine locations along a north–south transect in the 10-ha clearcut from June 26–August 17, 1996. Potential rates of mineralization were also measured during simultaneous laboratory incubations of the same materials. In the forest floor samples, most of the nitrogen was in the form of ammonium, and rates of mineralization were highest in the northern part of the clearcut. The same trend occurred in the laboratory incubation. In the mineral soil, nitrogen concentrations and rates of mineralization were smaller, but tended to be higher in the clearcut than in the forest.

INTRODUCTION

Nitrogen is the most commonly limiting nutrient in British Columbia forests. Nitrogen availability depends on release from organic matter through the interrelated processes of decomposition and mineralization. Decomposition refers to the actual breaking down of material and is measured as rates of weight loss over time. Mineralization refers to the release of nutrients from organic material as it is broken down. Some portion of the mineralized nutrients are taken up and temporarily immobilized by micro-organisms, then remineralized when the microbial biomass dies and decays. Rates of mineralization are therefore net rates; that is, the difference between rates of mineralization and immobilization. Because decomposition and mineralization are carried out by micro-organisms in the forest floor, rates of these processes are affected by factors which influence microbial activity. The most important factors influencing microbial activity are moisture, temperature, and the chemical composition of the organic material. Clearcutting and other silvicultural systems can affect each of these factors, and so have the potential to substantially alter rates of decomposition and nitrogen mineralization in forest floors.

Rates of decomposition often increase after clearcutting, but may also be slower or unchanged (Binkley 1984; Yin et al. 1989). Thinning of forest stands may also increase (Piene and Van Cleve 1978), decrease (Weetman 1965), or have no effect (Will et al. 1983) on decomposition rates. Very little is known about the effects of opening size or partial removals on decomposition rates.

Rates of net nitrogen mineralization in the forest floor and soil usually increase after cutting, and this is usually attributed to greater microbial activity resulting from the warmer, moister conditions in clearcuts (Edmonds and McColl 1989; Frazer et al. 1990; Smethurst and Nambiar 1990). Little is known about the effect of opening size or partial removals on N mineralization. In lodgepole pine forests, higher rates of net nitrogen mineralization were reported in a 0.25-ha patch cut (Prescott et al. 1992), and after removal of at least 15 trees (Parsons et al. 1994).

Coarse woody debris (cwd) has several ecological roles in ecosystems, including as habitat, as carbon storage, and as an organic matter reservoir. One of the functional roles that has been suggested for coarse woody debris is storage and cycling of carbon and nitrogen in forests, but little data exist to demonstrate its importance compared to other litter types. The recent surge in interest in coarse woody debris in the last two decades has prompted many studies, most of which quantify the mass of cwd in forests; less is known about the rates of input compared to other litter types.

Investigations of litter dynamics were established in the Sicamous Creek Silvicultural Systems trial to determine:

- rates of decomposition of several litter types in openings of different sizes and across one 10-ha clearcut;
- rates of net nitrogen mineralization in forest floor and mineral soil across a 10-ha clearcut; and
- rates of cwd input and other aboveground litter in old-growth ESSF forests.

METHODS

Study Site and Treatments

The study site is at Sicamous Creek, about 15 km east of the town of Sicamous, in the Salmon Arm Forest District. The site is in the Northern Monashee Wet Cold Engelmann Spruce – Subalpine Fir biogeoclimatic zone variant (ESSFwc2) at 1550–1800 m. The area is a mosaic of mesic, submesic, and subhygric sites (ESSFwc2/01, 04, and 06) and is occupied by an old-growth forest of Engelmann spruce and subalpine fir. The area is divided into three blocks, each containing one treatment unit of each silvicultural system: old-growth, 10-ha, 1-ha, and 0.1-ha openings and a partial cut. All cutting was done in the winter of 1994–1995.

Litter Decomposition

Rates of mass loss are being measured in five substrates:

- spruce and fir needles,
- forest floor material from the old-growth plots,
- fine roots of spruce and fir,
- lodgepole pine needles from a province-wide experiment, and
- aspen leaves from a province-wide experiment.

Two grams of each substrate were placed in fiberglass mesh bags with pore size of 2 mm. In October 1995, litterbags were installed in openings of each size and at seven stations along a north–south transect in the 10-ha clearcut (in forest at north and south edges, 12.5 and 25 m from each edge, and in the centre of the clearcut). One bag of each type was placed at each station: needles and leaves were pinned on the surface, forest floor and root bags were buried in the organic layer. All bags at seven stations will be collected from each location at yearly intervals for five years.

Nitrogen Mineralization

Rates of net nitrogen mineralization were measured in eight samples of forest floor and mineral soil (0–10 cm) collected at each of nine stations along a north–south transect in the 10-ha clearcut. These stations were designated by uppercase letters from A to I and were located in the following positions:

- A: in the forest at the south edge of the cut;
- B: 12.5 m from the south edge of cut;
- C: 25 m from the south edge of cut;
- D: 50 m from the south edge of cut;
- E: in the centre of the cut;
- F: 50 m from the north edge;
- G: 25 m from the north edge;
- H: 12.5 m from the north edge; and
- I: in the forest at the north edge.

One-half of each forest floor or mineral soil sample was placed in a polyethylene bag, sealed, and inserted back in its place of origin for 52 days (June 26–August 17). The other half of each sample was sent to the University of British Columbia, where it was sorted to remove roots and woody material. A portion of each sample was weighed, dried at 70°c, and reweighed to determine moisture content. One 5-g dry weight equivalent portion of each sample was extracted in potassium chloride (KCl) and initial concentrations of NH_4-N (ammonium) and NO_3-N (nitrate) were measured with an Alpkem autoanalyzer (MacMillan Bloedel Lab, Nanaimo). Another portion of each moist sample was incubated in polyethylene bags in the laboratory at about 20°c for 31 days (July 2–August 2). Following incubation either in the field or in the laboratory, samples were extracted in KCl and final concentrations of NH_4-N and NO_3-N were measured with the autoanalyzer.

Coarse Woody Debris and Litter Input

In each of the three old-growth (uncut) blocks, one 50 × 50-m permanent plot was established in August 1994. All live trees were tagged and the species and diameter (dbh) were recorded. All standing dead trees were also tagged and the species, diameter, and decay state were recorded. All fallen trees were marked with orange spray paint. One year later, each plot was surveyed to locate any trees that had fallen since the plots were established. Forty wooden 0.25 m^2 litter trays were installed in each of the three old-growth plots in September 1994. The litter that had fallen into each tray was collected three times in the next year (October 1994, June 1995, and September 1995) and sorted as needle, woody, or leaf litter material. After drying at 70°c, the weight of each material in each tray was measured. The needle litter was then used in the decomposition experiment.

TABLE 1 *Average annual litter input (g/m^2 per year) in old-growth plots (standard deviation appears in parentheses)*

Block	***N***	**Needle**	**Woody**	**Leaf**	**Total**
A	38	136.5 ab (95.0)[a]	16.6 (24.0)	3.1 (3.6)	155.9 (101.6)
B	40	97.8 b (91.6)	18.3 (15.5)	5.0 (10.4)	121.1 (95.0)
C	40	153.3 a (111.1)	13.0 (8.1)	2.3 (2.8)	168.6 (117.4)

[a] Within a column, mean values followed by different letters are significantly different ($p < 0.05$), based on one-way ANOVA and Bonferroni's multiple range test.

RESULTS

Litter Decomposition

The first year's samples were collected in October 1996.

Nitrogen Mineralization

Most of the extractable nitrogen in the initial (pre-incubation) samples of the forest floor was in the form of NH_4-N (Figure 1). Concentrations of extractable NH_4-N were higher at station H than at stations A, B, C, E, or I. Net nitrogen mineralization during the field incubation generally increased from station A to F, then decreased (Figure 2). This trend was also apparent in the samples incubated in the lab. This suggests that the differences were related more to the nature of the material than to differences in microclimate among the stations. In the lab incubation, net nitrogen mineralization was greater at stations F to I than at stations A to E. In both incubations, nitrate concentrations were low and net nitrification ranged from –2 to 2 μg/g. Moisture content of forest floor samples ranged from 71.3 to 78.4%.

In the mineral soil, initial concentrations of NH_4-N and NO_3-N were both lower than in forest floor, and more comparable to one another (Figure 3). Concentrations of NH_4-N at station D were greater than at stations A, B, C, F, G, and I. No significant differences in rates of net nitrogen mineralization or mineralized nitrogen occurred during incubations of mineral soil in the lab or the field (Figure 4). Most of the mineralized

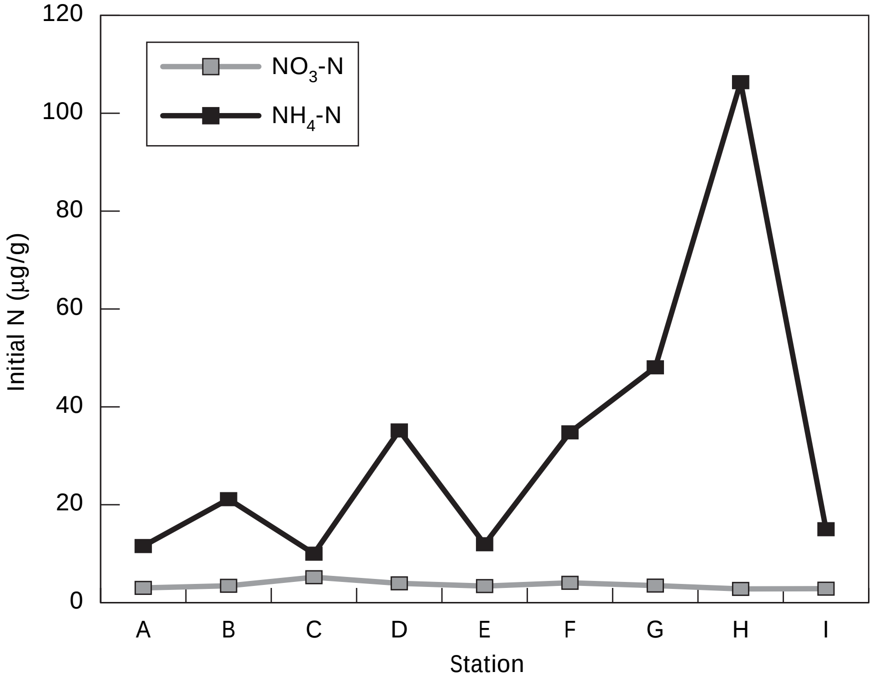

FIGURE 1 *Initial concentrations of extractable nitrogen in forest floor samples from nine points along a north–south transect in a 10-ha clearcut. Each value is the mean of eight samples.*

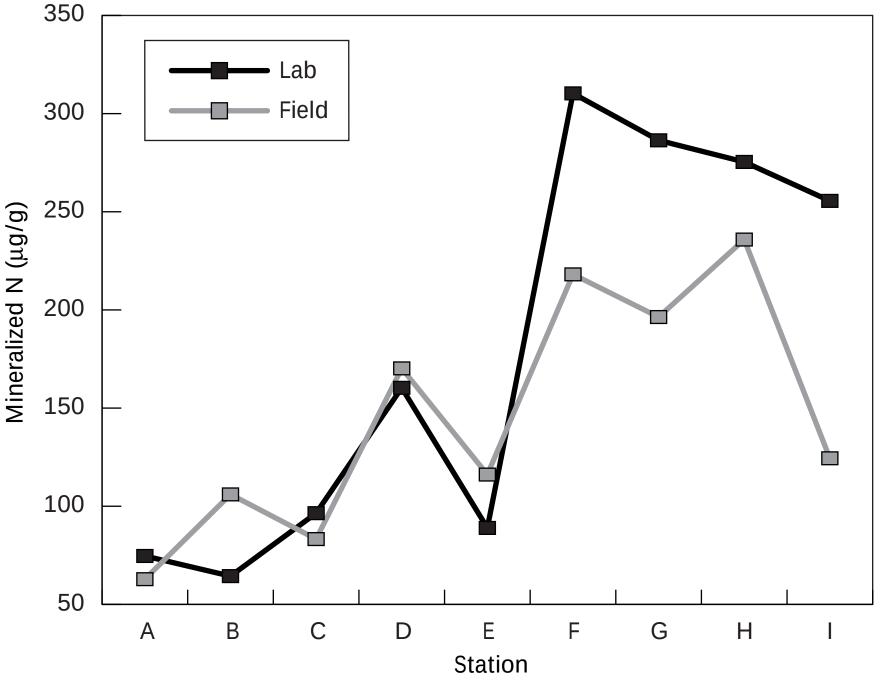

FIGURE 2 *Mineralized nitrogen (final concentrations) in forest floor samples from nine points along a north–south transect in a 10-ha clearcut after a 52-day field incubation and a 31-day lab incubation. Each value is the mean of eight samples.*

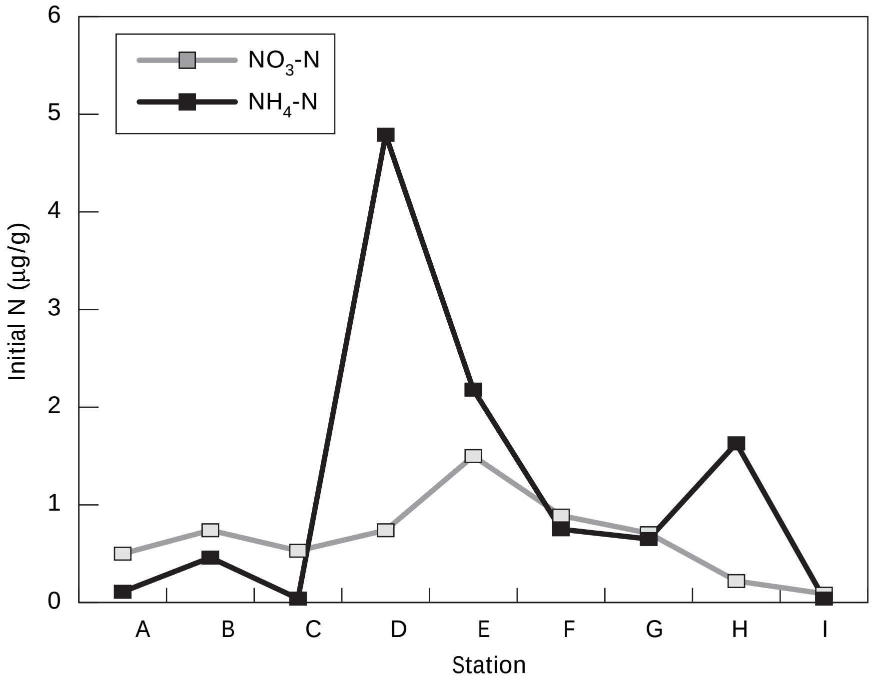

FIGURE 3 *Initial concentrations of extractable nitrogen in mineral soil samples from nine points along a north–south transect in a 10-ha clearcut. Each value is the mean of eight samples.*

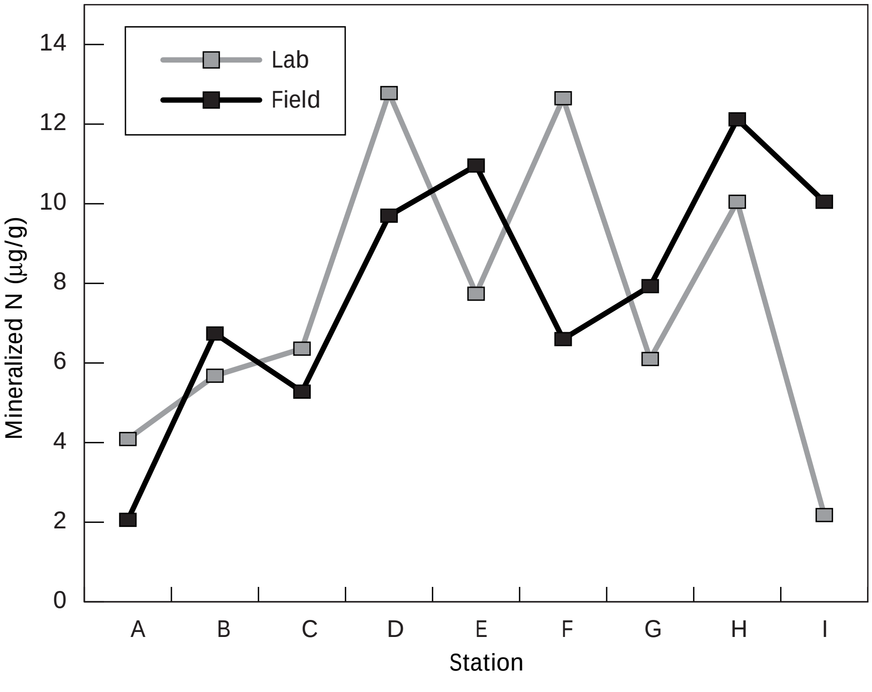

FIGURE 4 *Mineralized nitrogen (final concentrations) in mineral soil samples from nine points along a north–south transect in a 10-ha clearcut after a 52-day field incubation and a 31-day lab incubation. Each value is the mean of eight samples.*

nitrogen occurred as NH_4-N; rates of net nitrification ranged from –1.25 to 1.10 μg/g. Rates of net nitrogen mineralization and mineralized nitrogen were similar in mineral soil incubated in the lab and the field. Moisture content of mineral soil samples ranged from 34.4 to 48.9%.

Coarse Woody Debris and Litter Input

An average of 148.5 g/m^2 of aboveground litter fell in the three old-growth plots between September 1995 and 1996. Of this total, 87% was needles (129.2 g/m^2), 11% was fine woody debris (16 g/m^2), and 2% was leaves (3.4 g/m^2). More litter fell in block C than in block B; block A was intermediate. No trees fell in the three control plots during this period.

DISCUSSION

Nitrogen mineralization in the forest floor generally peaked in the northern half of the clearcut. This may be attributable to greater insolation, and thus higher temperatures in this part of the clearcut. However, the same trend was apparent during the lab incubation. This indicates that the differences observed in the field incubation resulted from differences in the nature of the forest floor material, since the climate was constant for all samples during the lab incubation. Similar results were found in another silivicultural systems trial in a montane forest in coastal British Columbia (Prescott 1995). Greater rates of nitrogen mineralization in forest floor material from clearcuts during the lab incubation was attributed to reduced availability of fresh carbon substrates resulting from reduced litter input. This would limit the size of the microbial biomass and its ability to immobilize nitrogen as it was mineralized from the residual forest floor. In the Sicamous Creek study, the higher nitrogen mineralization in the northern part of the transect may be indirectly related to microclimate because this material was more decomposed and hence had less carbon available to immobilize the nitrogen as it was mineralized. It is surprising that these changes were apparent within a few months of clearcutting. We can not rule out the possibility that the differences within this clearcut are the result of microsite variation. This will be tested by measuring rates of mineralization in other 10-ha cuts and in each opening size in 1996.

Concentrations of nitrate and rates of net nitrification during incubation were very low, suggesting that little leaching of nitrate will occur from the 10-ha clearcut. This can be tested using nitrogen leaching information collected by M. Feller (see this proceedings, page 121).

Each of the measurements taken at various distances from the edge will be compared with microclimate data across the same clearcut, to see if rates of decomposition and mineralization can be related to temperature or moisture conditions across the clearcut. These rates should also be measured in openings of various sizes, to determine if the rates in the openings can be predicted from rates at different distances from edges.

Annual aboveground litter input at the Sicamous Creek ESSF site (149 g/m^2) was higher than the average of 124 g/m^2 measured in a dry

ESSF site in the Kananaskis Valley of Alberta (Prescott et al. 1989). Treefall added another 47 g/m² at the Kananaskis site, but was variable between years. Treefall should be monitored at the Sicamous Creek site for several more years to get an accurate estimate of input rates of coarse woody debris. Concentrations of carbon and nitrogen in both types of litter should also be measured to ascertain the importance of this debris in cycling and storage of carbon and nitrogen in this ecosystem.

CONCLUSIONS

Higher rates of nitrogen mineralization were measured in forest floor samples from the northern half of the 10-ha clearcut than in the southern half or in the centre of the clearcut. This resulted from differences in the nature of the material rather than microclimate, since the same trend was apparent in the laboratory incubation. No differences in nitrogen mineralization occurred in mineral soil samples from across the clearcut. Annual aboveground litter input averaged 149 g/m² in the old-growth plots. No treefall occurred during the first year of measurement in the old-growth plots.

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Effects of Several Silvicultural Systems on Ectomycorrhizal Diversity and Hypogeous Sporocarp Biomass in the Engelmann Spruce – Subalpine Fir Zone: Preliminary Results

MELANIE JONES AND DANIEL DURALL

ABSTRACT

Mycorrhization is essential for the proper growth of conifers in natural environments and can result in increased growth rates of outplanted tree seedlings. Fungi that produce hypogeous (belowground) sporocarps are important in temperate forest ecosystems because most enter into ectomycorrhizal associations, and because they are an important food source for small mammals. The objective of this part of the Sicamous Creek Silvicultural Systems Project is to determine whether opening size affects the production of hypogeous sporocarps and the diversity of ectomycorrhizal inoculum retained on an Engelmann Spruce – Subalpine Fir biogeoclimatic zone site in the southern interior of British Columbia. Soil cores were sampled in early August to determine how ectomycorrhizal diversity was affected by the three different cutblock sizes. The roots examined were primarily those of trees removed the previous winter. Thus, the major mycorrhizal types observed were the same as those sampled before logging. Ectomycorrhizal diversity was lower in the centre of the 10-ha clearcut than at points up to 25 m from the edge of the clearcuts or in the adjacent undisturbed forest. However, linear regression of diversity against distance was not significant. Previous studies show that ectomycorrhizae survive for a year following logging. We therefore expect differences in the ectomycorrhizal community across opening size to become greater over the next two years. Hypogeous sporocarps were collected from circular 4 m^2 plots after stripping the plots of all litter to the depth of the A horizon. Hypogeous sporocarps were absent from all sizes of cutblocks and their biomass on the undisturbed control was about one-third of the amount found in July 1994 (pre-harvest).

INTRODUCTION

Ectomycorrhizal Diversity

Ectomycorrhizae are symbiotic associations between specific fungi and the fine roots of many temperate woody plants. They comprise the nutrient and water-absorbing structures of these plants. Mycorrhizal formation is essential for the proper growth of conifers in natural environments, and can result in increased growth rates of outplanted tree seedlings (e.g., Villeneuve et al. 1991). Major differences exist, however, between the 5000 fungal species that form ectomycorrhizae. For example, some are better at breaking down complex soil nutrients; some form hyphal strands which are effective in transporting water; some colonize roots growing in decaying wood, while others are more common in mineral soil (Bruns 1995). Given the importance of ectomycorrhizal fungi to nutrient uptake, maintaining a high ectomycorrhizal diversity is arguably an important aspect of forest health and sustainability (Amaranthus and Perry 1994). On a site with a high diversity of ectomycorrhizal fungi, roots can associate with the fungus best adapted to the soil microsite through which they are growing. This allows the tree to efficiently exploit the heterogeneous soil environment. The decrease in fungal inoculum and species richness, and the changes in fungal community composition which occur following clearcutting (Parke et al. 1984; Pilz and Perry 1984; Parsons et al. 1994) may constitute a reduction in forest sustainability on a site. Some changes in the ectomycorrhizal fungal community with logging will be inevitable; however, some site preparation treatments (Jones et al. 1996) and replanting regimes (Jones et al., unpublished data) retain more diversity than others. The objective of this part of the Sicamous Creek Silvicultural Systems Project is to determine whether opening size affects the diversity of ectomycorrhizal inoculum retained on an Englemann spruce – subalpine fir site in the southern interior of British Columbia.

Hypogeous Fungi

Fungi that produce hypogeous (belowground) sporocarps are important in temperate forest ecosystems because most enter into ectomycorrhizal associations, and because they are an important food source for small mammals (Maser et al. 1978). As much as 90% of the diet of some rodents is composed of hypogeous sporocarps. In return, the fungi depend on the small mammals to disperse their spores (Trappe and Maser 1976). The interdependence of fungi which produce hypogeous sporocarps and other organisms in the forest ecosystem emphasizes the importance of studying these fungi and their interactions with other organisms, both before and after disturbance.

Amaranthus et al. (1994) found a reduced frequency of occurrence of hypogeous sporocarps in regenerating clearcuts when compared to a 180-year-old Douglas-fir forest. Nothing is known about the effect of different silvicultural systems on the production of hypogeous sporocarps. The objective of this part of the study is to determine whether opening size affects the production of hypogeous sporocarps on an Englemann spruce-subalpine fir site in the southern interior of British Columbia.

Site Description

The Sicamous Creek study site is located in the Engelmann Spruce – Subalpine Fir biogeoclimatic zone (ESSFwc2 grading into ESSFvv) in the Salmon Arm Forest District (see Vyse, this proceedings, page 4). The site is located south of the north fork of Sicamous Creek and north of Mount Mara (see Vyse, this proceedings, page 4). Five treatments with three replications were laid out in a randomized complete block design. Each block extends along the slope at a different elevation. The treatments, which were imposed in the winter of 1994–95, removed 30% of the tree cover from each of the 30-ha plots (see Vyse, this proceedings, page 4).

The treatments included opening sizes of 0.1, 1.0, and 10 ha, a selection cut, and an uncut control.

METHODS

Ectomycorrhizal Diversity

During late August 1993 (before harvesting), 20 samples were collected with a spade from each of the nine plots on the sites of the future 10-ha, 0.1-ha, and control treatments. Of these, 10 consisted of duff and 10 of mineral soil. Samples were collected in a 4 × 3-m grid located approximately in the centre of the plot. Soil and duff samples were removed from alternate sampling points, 1 m apart on the grid. Samples were placed in plastic bags, returned to Okanagan University College in Kelowna and kept at 4°c until examined. In early August 1995 (first summer after harvest), 10 soil cores (5 cm diameter by 30 cm long) were removed from points at least 1 m away from each other in the centre of three replicate cutblocks of each of the three sizes: 0.1, 1.0, and 10 ha. Sampling was performed at six additional points in the 10-ha treatment unit: in the rooting zone inside (2–3 m) the north and south ends of the clearcut, 25 m in from the north and south ends of the clearcut, and 40 m into the adjacent forest at the north and south ends. In total, 300 cores (10 sampling locations × 10 cores per location × 3 replicate treatment units) were collected. After transportation to the laboratory, the soil was stored at 4°c until examined.

For both the pre- and post-harvest samples, roots were washed free of soil and cut into 2-cm fragments. On randomly selected fragments, every mycorrhiza judged to be alive was described, until a total of 100 (pre-harvest) or 200 (post-harvest) tips per sample had been described. The number of dead or non-mycorrhizal root tips was also recorded. Detailed morphological examination of the mycorrhizae, according to the protocol of Goodman et al. (1996), was used to categorize the tips into different morphological types. In brief, this involved describing the colour and general appearance of the mycorrhiza and any mycelial strands under a dissecting microscope. Then, under higher magnification, the pattern and size of hyphae on the surface of the root and the appearance of emanating hyphae and cystidia were described. Each morphological type is presumed to be formed by a different ectomycorrhizal fungus, although some may represent different developmental stages of the same mycorrhiza and some morphological groupings may comprise mycorrhizae formed by closely related species. The morphological groupings are now being confirmed using dna analysis. Examples of each type were photographed at 50× and 400× magnification.

For the 1995 samples, root tips were categorized as “live” or “dead” using the criteria of Harvey et al. (1976). Ectomycorrhizae were considered alive if they were turgid and the stele was intact.

Richness, evenness, and Simpson’s diversity index, which is based on the morphological groupings, were calculated using the AID1 programs (Overton 1987). Separate regressions of these indices against distance into the opening were performed using the data from the centre of each gap, and using data from only the samples in the 10-ha openings. In cases where variances were heterogeneous, non-parametric tests were performed (Wilcoxon/Kruskal-Wallis). All analyses were performed using JMP (2.0, SAS Inc.).

Sampling and Identification of Hypogeous Sporocarps

Hypogeous sporocarps were collected from circular 4-m^2 plots (Luoma et al. 1991) after stripping the plots of all litter to the depth of the A horizon. In 1994, hypogeous sporocarps were collected before the harvest treatments during three sampling periods: June 14–29, July 20–29, and September 11–18. Each time, sporocarps were collected from the three replicate control, 0.1-ha, and 10-ha treatment units. Sporocarps were collected from thirteen 4-m^2 plots per treatment unit, located on mesic to subhygric ecosystems on the site. Post-harvest sampling occurred in July 1995 because sampling the previous year had shown maximum hypogeous sporocarp production at this time.

Sampling circles were positioned in the following manner: one on each of fifteen 0.1-ha cutblocks per treatment unit, three on each of five 1.0-ha cutblocks per treatment unit, and fifteen on each 10-ha cutblock. The circles were located approximately in the centre of the cutblocks, at least 50 m from the forest edge, at least 2 m from any naturally regenerating seedlings, and with 10 m between circles. For the control treatments, sampling was conducted in the 01 or 04 site series. Sporocarps were dried and weighed, and identified using keys and descriptions from Castellano et al. (1989) and Smith et al. (1981).

RESULTS AND DISCUSSION

Ectomycorrhizae

In the first summer following harvest, the percentage of roots which were judged to be dead ranged from 20 to 30% of the total observed. No differences existed between the proportion of roots classified as dead in the centre of the cutblocks (22–29%) and in the control samples (20–25%). This implies that little obvious mortality of fine roots had occurred during the first summer following harvesting. Other studies showed that ectomycorrhizae appear to remain active on a site for one or two years following harvesting (Harvey et al. 1986; Perry et al. 1987). We would therefore expect the proportion of dead conifer roots to increase in subsequent sampling years.

Following harvest, mean ectomycorrhizal diversity was lower in the centre of the 10-ha openings than in the sampling locations closer to the forest edge, but regression analyses did not detect significant effects of distance (Figure 1a: $p = 0.17$). The richness component of diversity was affected less than the evenness component. This implies that the relative proportion of the different mycorrhizal types changed more than the number of types present. No differences in diversity occurred among samples taken at the centres of the different opening sizes (Figure 1b). Because these diversity values are based on samples collected the first summer following logging, we expect differences across the 10-ha clearcuts to increase over the next several years.

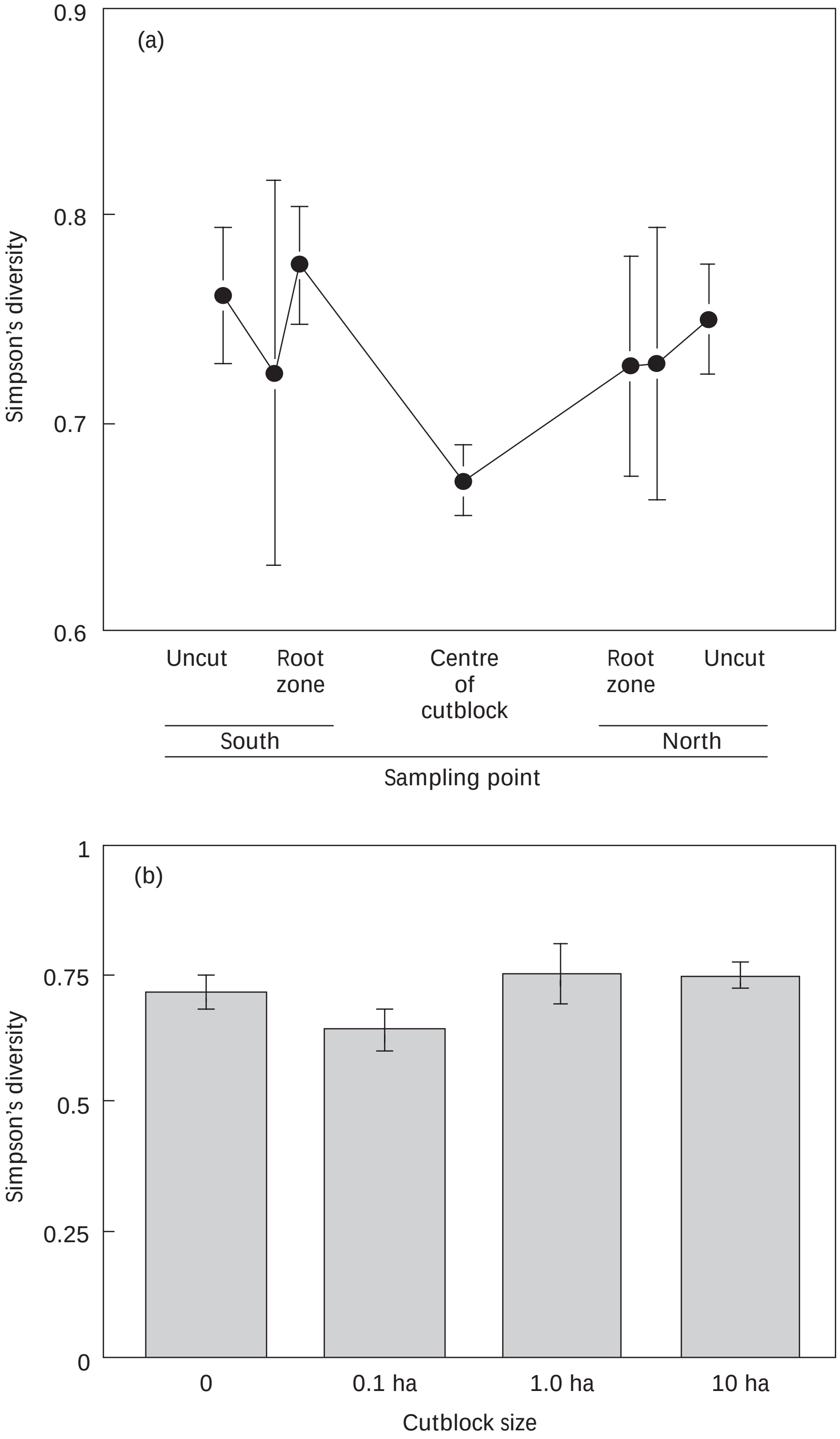

FIGURE 1 *Simpson's diversity of ectomycorrhizal types found in soil cores collected at Sicamous Creek in the August following harvesting. Mean ± SE, n = 3; diversity calculated on 2000 live mycorrhizae examined per sampling location: (a) samples collected in 10-ha cutblocks, and (b) samples collected on control treatment units and in the centres of cutblocks.*

The mycorrhizal types that comprised at least 5% of the roots in any one sampling location following harvest included: *Cenococcum geophilum*, *Amphinema byssoides*, *Cortinarius*-like species, *Hebeloma*-like species, *Laccaria*-like species, two *Lactarius* types, *Piloderma croceum*, *Tuber*-like species, and *Mycelium radicis atrovirens* (dark septate). Not surprisingly, these were the same major types observed before logging. At that time, *Cenococcum geophilum* and the *Hebeloma*-like mycorrhizae comprised 35% and 22%, respectively, of the mycorrhizae examined. In 1995, *Cenococcum geophilum* mycorrhizae were by far the most abundant encountered, forming 42% of the mycorrhizae examined. One interpretation is that these mycorrhizae survive longer than *Hebeloma*-like mycorrhizae following logging. It is possible however, that when dead, these mycorrhizae are more resistant to microbial attack than other mycorrhizal types and thus are more readily classified as "living" than they should be. The use of chemical vitality tests on samples collected in the summer of 1996 should eliminate this uncertainty.

The pre-harvest sampling of hypogeous sporocarps indicated that species richness was highest in July (Table 1), thus the post-harvest

TABLE 1 *Hypogeous sporocarp biomass at the Sicamous site during June, July, and September, 1994*

Fungal taxa	**Sporocarp biomass (g/ha)**[a] **June**	 **July**	 **September**
Chamonixia sp. (HSOUC/94 003)	—	4 (2)[b]	—
Elaphomyces granulatus	5145[c] (200)	—	—
Hydnotrya cubispora	693 (396)	566 (269)	—
Hysterangium setchellii	51 (50)	39 (20)	4 (6)
Hysterangium sp. (HSOUC/94 012)	—	23 (6)	—
Leucogaster sp. (HSOUC/94 006)	—	20 (5)	1 (4)
Rhizopogon ochraceorubens	—	228 (16)	—
Rhizopogon sp. (HSOUC/94 007)	—	—	92 (15)
Rhizopogon sp. (HSOUC/94 008)	—	—	—
Rhizopogon sp. (HSOUC/94 009)	—	10 (2)	—
Thaxterogaster pingue	63 (47)	40 (23)	—
Unknown (HSOUC/94 010)	—	34 (29)	—
Unknown (HSOUC/94 011)	3 (2)	—	—
Total (g/ha)	5955 [810[d]]	964	97
Species richness[e]	5	9	3

[a] Values (g/ha) are estimated from a 468-m^2 sample area.
[b] Values in parentheses are the number of sporocarps collected from a 468-m^2 sample area.
[c] All sporocarps are from one 4-m^2 area.
[d] Value is excluding *E. granulatus.*
[e] Values are numbers of species.

sampling occurred in this month. In the first summer following harvest, production of hypogeous sporocarps was limited to the control plots (Table 2). No sporocarps were found in any size of cutblock. Sporocarp biomass was approximately one-third that of July 1994 (964 g/m). Seven species were collected in 1995 compared to nine in 1994. This may be because the area sampled was slightly less in 1994 (300 m^2 in the unlogged areas in 1995 vs. 468 m^2 in 1994). *Thaxterogaster pingue* formed a more dominant part of the community in 1995 than in 1994, and *Rhizopogon* spp. were less common in 1995. *Hysterangium setchellii* formed a very high proportion of the biomass in Block A, but all of this biomass was from one 4-m^2 plot. Generally, the same species were encountered in the two years.

TABLE 2 *Hypogeous sporocarp biomass on the undisturbed control plots at the Sicamous site in July 1995. Values (g/ha) are estimated from a 100 m^2 per block sample area. Values in parentheses are the number of sporocarps per hectare estimated from a 100 m^2 per block sample area.*

Fungal taxa	**Block A (g/ha)**	**Block B (g/ha)**	**Block C (g/ha)**
Hydnotrya cubispora	21 (100)	128 (900)	22 (300)
Hysterangium setchellii	516 (3100)[a]	—	5 (100)
Rhizopogon sp. 1	—	29 (800)	—
Rhizopogon sp. 1	—	—	7 ()
Thaxterogaster pingue	9 (100)	202 (1000)	16 (600)
Unknown 1	—	9 (100)	—
Total (g/ha)	546 (3300)	369 (2800)	52 (1000)
Species richness[b]	3	4	4

[a] All sporocarps are from one 4-m^2 area.
[b] Values are numbers of species.

SUMMARY AND CONCLUSIONS

Although the study is ongoing, some preliminary conclusions are possible:

- Removal of the trees at Sicamous Creek severely reduced the production of hypogeous sporocarps. No hypogeous sporocarps were found in any of the cutblocks.
- The dominant ectomycorrhizal types found in soils sampled from the cutblocks were similar to those found before harvesting, indicating that most ectomycorrhizal fungi survived into the first summer following logging. The ectomycorrhizal community is expected to change by the second summer following logging.

- Mean ectomycorrhizal diversity was lower in the centre of the 10-ha cutblocks than in soil samples collected up to 25 m from the edge. However, no significant relationship existed between distance from the edge and ectomycorrhizal richness, evenness, or diversity. In addition, no effect of opening size on ectomycorrhizal diversity was observed by the first summer following logging.

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Effects of Clearcutting on Soil Mites and Springtails in the Subalpine Forest of Sicamous Creek

HANNAH NADEL

ABSTRACT

Soil mites and springtails are functionally and numerically important components of decomposition, nutrient cycling, and soil formation in forests. The effect of silvicultural practices on these organisms has received little attention, especially in North America. In this study, substrate preference, abundance, and diversity of soil mites and springtails were examined at three points along a north–south transect in a square 10-ha clearcut and in an uncut control during the first year after cutting. Abundance of mites and springtails was significantly higher in the forest floor than in decayed wood, and lowest in mineral soil. Overall abundance of the soil micro-arthropods did not change significantly in the clearcut when compared to the control populations. However, one common mite family, Tectocepheidae, appeared to increase in parts of the clearcut, and one common springtail family, Hypogastruridae, appeared to decrease in the middle of the clearcut when compared to the control population. Diversity was examined at the family-level only, and did not change significantly in the clearcut.

INTRODUCTION

Sustainability of subalpine forest timber extraction has traditionally been foiled by poor regeneration success. One of the components of forest ecosystems that may affect regeneration is the soil arthropod fauna, including the highly diverse and abundant mites (Acari) and springtails (Collembola). The activities of these soil micro-arthropods, many of which are saprophagous, increase availability and suitability of organic particles for decomposer communities and promote the dispersal of decomposer propagules to new substrates (Norton 1990). They also contribute to nutrient cycling and soil formation (Behan-Pelletier 1993).

The effect of silvicultural practices on forest soil mites and springtails has received little study, especially in North America (Bird and Chatarpaul 1986). However, these groups represent a large, diverse, and functionally important proportion of the soil fauna. Mites are second only to insects in faunal diversity in Canada, and can reach densities of one million individuals per square metre in the organic horizons of soils (Norton 1990; Behan-Pelletier 1993). Springtails are also abundant, with reported densities of 850 000 individuals per cubic metre in well-decayed woody debris in coastal forests of British Columbia (Setälä and Marshall 1994).

Clearcutting significantly reduces abundance of some or all major mite taxa in forest soils (Huhta et al. 1969; Vlug and Borden 1973; Seastedt and Crossley 1981; Bird and Chatarpaul 1986) and may also create shifts in species dominance (Bird and Chatarpaul 1986). Springtails also decline in numbers on clearcuts harvested by whole-tree logging, but not in conventionally harvested plots (all slash, and stems smaller than 9 cm in diameter left on the sites) (Bird and Chatarpaul 1986). In other cases, clearcutting did not have a significant long-term effect on springtail abundance even though numbers rose initially (Huhta et al. 1969; Seastedt and Crossley 1981; Bird and Chatarpaul 1986).

Harvesting may initially provide increased root decomposition and litter, and may also increase soil moisture because the water uptake by trees is eliminated. It may consequently create an initial increase in the nutrients and water available to soil organisms, which allows their populations to rise in the first few years after cutting (Seastead and Crossley 1981). Microclimatic changes associated with tree removal may augment or counteract these favourable conditions. Clearcutting and other silvicultural disturbances increase the forest floor's exposure to solar radiation and air currents and change the quantity and composition of the litter (e.g., Seastedt and Crossley 1981).

Changes in the quality and quantity of various soil substrates following disturbance may also have a profound effect on soil organisms and their relative densities within each substrate. A preliminary study of the upper 3 cm of three substrates in undisturbed mesic sites in the Sicamous Creek Silvicultural Systems Project in British Columbia showed that densities of mites and springtails differ in the forest floor, mineral soil, and decayed wood. More than 50% of soil mites and springtails were found in the forest floor layer, while the remainder were nearly equally distributed between the mineral soil and highly decayed (Class V; Maser et al. 1988) wood (Nadel 1995).

Although densities of mites and springtails are highest in the forest floor, coarse woody debris is a long-term source of energy and nutrients that may also play a critical role in moisture retention in some systems (Shaw et al. 1991). Decayed logs may act as refugia for some forest organisms after tree removal, and may also prove important in the reintroduction of micro-arthropods to the forest floor and mineral soil as forests regenerate.

Changes in faunal abundance, distribution, and community composition are known to follow clearcutting, although not always quickly (e.g., Huhta et al. 1969; Sundman et al. 1978). Because few studies exist on the effects of forest harvesting on soil micro-arthropods, and because soil arthropod response seems variable, there is no clear expectation of the type of trend that will be found on any site.

Because mites and springtails are numerically and functionally significant in regulating decomposition and other soil processes, their abundance and diversity in forest soils should be monitored to determine how they are affected by silvicultural practices. Various substrates must be studied to determine their relative importance for soil arthropods during silvicultural management. In addition, various cut sizes must be examined to determine an optimal size that minimizes the impact on these soil organisms and, ultimately, on the regenerative capacity of the forest under study.

This report summarizes the findings of a study on soil mites and springtails undertaken as part of the Sicamous Creek Silvicultural Systems Project. Three substrates were sampled across a transect in a 10-ha clearcut and in intact forest during the first year after tree removal. The centre of the clearcut was sampled to provide information on the area expected to be most heavily changed by cutting. The clearcut edges were sampled to gain information on canopy openings that retain a living tree rhizosphere and that are, therefore, ecologically analogous to small cuts. This study will be repeated and expanded over the next two years and at five-year intervals in the future.

MATERIALS AND METHODS

Study Site

The Sicamous Creek research area encompasses 1000 ha of old-growth Englemann Spruce – Subalpine Fir (ESSFwc2) zone forest in the Salmon Arm Forest District. It is located between the north fork of Sicamous Creek and Mara Mountain at an elevation of about 2000 m.

Logging occurred during winter of 1994–95 to produce canopy openings of five treatment sizes with a standard 30% wood volume removal in each. These treatments included single-tree selection, 0.1-, 1.0-, 10-ha cuts, and an uncut control. These were arranged in a randomized block design, with three replicate blocks per treatment.

Methodology

Micro-arthropod sampling was conducted across a median north–south transect on a 10-ha cut designated as “B-5.” This block is square and contains soil moisture indices of 01 (mesic) and 06 in nearly equal amounts, and a few small patches of 08.

Sampling occurred on August 1 and September 14, 1995, at three points within the B-5 clearcut and at a control point located 40 m to the north of the cut in undisturbed forest. Within the clearcut, one sampling point was 5 m from the north edge and another 5 m from the south edge within the living tree rhizosphere, and the third was near the centre, about 160 m from the cut edge. All were restricted to pre-mapped mesic (01 soil moisture index) sites. Five soil samples were taken on both days at each point in each of three substrates, for a total of 120 samples.

Samples consisted of cores 5 cm in diameter and 3 cm deep. The upper 3 cm of forest floor, mineral soil, and decayed wood (decay class V) substrates were sampled. Forest floor and mineral soil samples were sectioned from single cores. The moss layer above the forest floor and decayed wood samples was removed. Samples were then placed in polyethylene bags and transported in a cooler to an extraction facility in Victoria.

In the laboratory, micro-arthropods were driven by heat and light into cups of picric acid using high-gradient extraction and preserved in vials of 70% ethanol. Half the samples were extracted immediately, while the rest were first stored at 2 ± 1°c for seven days. The data showed that storage did not affect the extraction procedure (unpublished data). Extraction lasted seven days for each batch of samples. The vials were emptied into plates and the specimens picked out under a dissecting microscope, placed on temporary slide mounts in lactic acid, identified to family under a stereo or compound microscope, and counted.

Biometric Approach

The effect of month on micro-arthropod abundance was tested for each substrate at each transect point by the Mann-Whitney U test (GraphPad Software 1994). Effects of transect point and substrate on abundance of mites and springtails were tested by two-way anova on log-transformed totals from each substrate and point (August and September data pooled), followed by the Tukey-Kramer multiple comparisons test for all pairs of substrates (GraphPad Software 1994).

Substrate preferences of the commoner mite and springtail families, represented in 1995 samples (August and September data were pooled) by at least 100 specimens, were tested by parametric one-way ANOVA followed by the Tukey-Kramer multiple comparisons test (GraphPad Software 1994). To meet the assumptions for anova, and because samples contained zero values, the data were first transformed by $\log(x + 1)$, where x was the observed value.

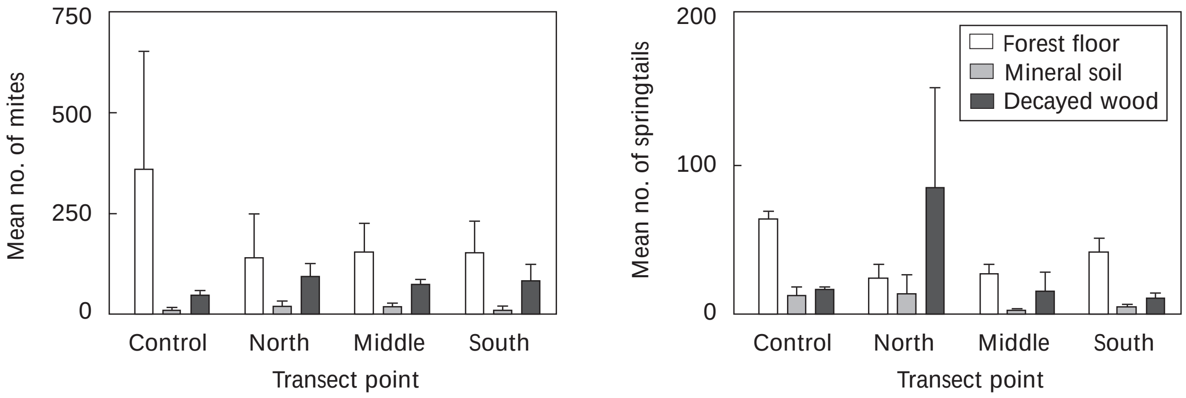
Preliminary comparisons of the diversity of mite communities among transect points were done at the family level, under the assumption that family-level diversity reflects species-level diversity to some degree. Comparisons were made using Spearman's rank correlation on total numbers per family found at each transect point. Differences were deemed statistically significant at the 0.05% level.

RESULTS AND DISCUSSION

Mite and springtail densities were generally higher in September than in August in each substrate and transect point (Figure 1), but were not significantly different between the two months (Mann-Whitney U tests). Because densities and diversity of the number of arthropod families represented were similar in the samples over both months, the data for August and September were pooled for most of the remaining statistical tests.

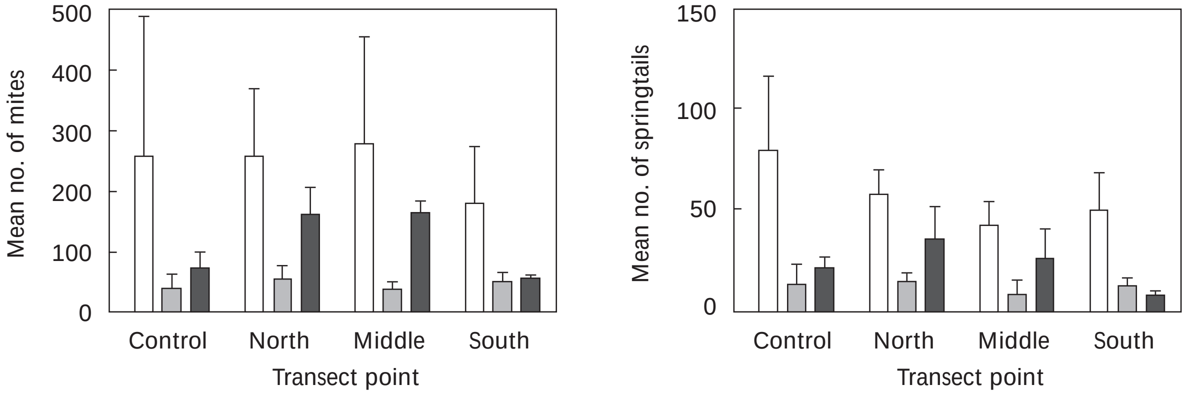
The abundance of micro-arthropods was significantly affected by the type of substrates examined (i.e., forest floor, mineral soil or decaying wood), but the location from which the samples were taken (uncut forest or clearcut point) did not affect micro-arthropod abundance (Table 1). Also, no significant interaction was found between the substrate factor and the transect point factor.

An analysis of mite and springtail densities in the three substrates showed that the forest floor contained significantly higher densities of both groups than the other substrates (Table 2). Mite densities were second highest in decayed wood, and were significantly lowest in mineral soil. Relative springtail densities were not as consistent among the



(a) August

(c) August



(b) September

(d) September

FIGURE 1 *Mean numbers of mites and springtails (± s.d.) found in soil cores in three substrates at each of four transect points (n = 5 per substrate per transect point) during two months in 1995: (a) mites, August; (b) mites, September; (c) springtails, August; (d) springtails, September.*

TABLE 1 *Results of two-way analysis of variance on log (n + 1)-transformed numbers of soil arthropods in core samples*

	August			**September**		
	% of variance	***F***	***P***	**% of variance**	***F***	***P***
Mites						
Substrate	38.5	16.20	< 0.0001 (***)	60.3	40.59	< 0.0001 (***)
Transect point	3.1	0.85	0.4713 (n.s.)	1.4	0.61	0.6121 (n.s.)
Interaction	3.5	0.50	0.8088 (n.s.)	4.5	1.01	0.4320 (n.s.)
Springtails						
Substrate	42.0	20.89	< 0.0001 (***)	32.4	11.87	< 0.0001 (***)
Transect point	6.0	2.00	0.1273 (n.s.)	2.9	0.71	0.5504 (n.s.)
Interaction	4.7	0.79	0.5852 (n.s.)	5.3	0.65	0.6919 (n.s.)

TABLE 2 *Comparison of mean density [a] (± s.d.) of mites and springtails in soil cores (n = 20 samples per substrate per month)*

Substrate	August		September	
Mites				
Forest floor	205.80a	(± 105.3)	244.30a	(± 41.39)
Decayed wood	77.35b	(± 20.89)	113.10b	(± 57.97)
Mineral soil	18.30c	(± 2.13)	44.50c	(± 9.73)
Springtails				
Forest floor	40.30a	(± 17.88)	57.90a	(± 15.66)
Decayed wood	33.25a	(±35.49)	22.90b	(± 11.41)
Mineral soil	8.80b	(± 5.62)	12.10b	(± 2.52)

[a] Means followed by the same letter are not significantly different from each other ($p < 0.05$).

substrates during the two sample months. Densities in forest floor and decayed wood were not significantly different from each other, but were significantly higher than in mineral soil in August. In September, however, the forest floor had higher densities than decayed wood and mineral soil, but the latter two were not significantly different from each other. A trend emerged with springtail densities highest in forest floor and lowest in mineral soil, similar to the trend observed for mites, but more data may be needed to show this statistically.

The presence of a living tree rhizosphere at the edge of the cut did not significantly affect micro-arthropod abundance compared with the middle of the cut, where only severed roots remained (Table 2). This suggests that either root condition does not directly affect mites and springtails or that these effects may be delayed. The abundance of hypogeous mycorrhizal fungi, which depend on living roots and which may provide an important food source for some micro-arthropods, was also not significantly affected in the Sicamous Creek cut sites during the first year after cutting, but is expected to decrease with time (Jones and Durall, this proceedings, page 101). Food resources and soil moisture may be more favourable for certain micro-arthropods directly after tree cutting because of the remaining dying roots and increased litter, and the elimination of water uptake by trees (Seasteadt and Crossley 1981). Because no clear changes in micro-arthropods occurred in the edges or middle of the B-5 clearcut, it is unlikely that any significant changes occurred in the smaller cut sizes during this year.

Thirty-five families of mites in four suborders, and seven families of springtails in two suborders were identified in the samples (Table 3). Despite the general lack of keys to immature mites, all but 13 immatures were identified to the family level by their resemblance or association with the adults, and their numbers are included in all tables and analyses of abundance. When mite communities were compared between transect

TABLE 3 *Distribution and abundance of mites and springtails among transect points in a 10-ha clearcut and an uncut control in 1995. Numbers represent individuals found in soil cores (n = 40 per transect point).*

	Control	North	Middle	South	Total
Mites (Acari)					
Suborder Prostigmata					
Family Nanorchestidae	509	810	622	591	2532
Eupodidae	212	158	234	78	682
Tydeidae	212	113	168	181	674
Alicorhagiidae	66	91	43	90	290
Rhagidiidae	103	26	32	30	191
Scutacaridae	36	10	128	6	180
Tarsonemidae	24	57	4	23	108
Pygmephoridae	24	8	16	35	83
Ereynetidae	40	5	18	1	64
Pachygnathidae	3	1	2	6	12
Raphignathidae	0	1	1	0	2
Suborder Mesostigmata					
Family Zerconidae	170	75	68	81	394
Uropodidae	64	34	61	60	219
Rhodacaridae	23	13	44	18	98
Polyaspididae	26	11	11	4	52
Parholaspididae	6	2	7	12	27
Ascidae	14	7	4	0	25
Parasitidae	4	6	1	4	15
Halolaelapidae	3	2	0	0	5
Digamasellidae	1	0	0	0	1
Suborder Cryptostigmata					
Family Oppiidae	1361	350	237	251	2199
Tectocepheidae	30	585	882	319	1816
Brachychthoniidae	409	503	269	470	1651
Camisiidae	168	452	591	270	1481
Suctobelbidae	130	141	108	85	464
Achipteriidae	7	24	54	8	93
Cepheidae	12	15	2	8	37
Peloppiidae	18	2	1	4	25
Ceratozetidae	5	7	8	3	23
Damaeidae	2	4	4	4	14
Eremaeidae	7	2	0	0	9
Astegistidae	0	2	0	0	2
Palaeacaridae	0	1	0	0	1
Mycobatidae	0	1	0	0	1
Suborder Astigmata					
Family Acaridae	232	127	78	144	581
Springtails (Collembola)					
Suborder Arthropleona					
Family Isotomidae	665	939	528	516	2648
Hypogastruridae	187	121	29	62	399
Onychiuridae	154	79	69	84	386
Tomoceridae	8	3	0	2	13
Entomobryidae	1	0	0	0	1
Suborder Symphypleona					
Family Neelidae	5	24	6	4	39
Sminthuridae	7	5	1	6	19

points, their presence and abundance were very similar across all points (Table 3). Families were usually consistently common or rare in all transect points.

One statistically significant (anova, $F = 11.786$, $p < 0.0001$) exception did appear among one of the dominant mite families. The Tectocepheidae, represented in the Sicamous Creek research area probably by a single species (*Tectocepheius velatus* [Michael]), was significantly more common in the middle of the clearcut than at any other transect point (Tukey-Kramer multiple comparisons test: middle vs. control, $p < 0.001$; middle vs. north, $p < 0.01$; middle vs. south, $p < 0.01$). In the control, this species was very scarce and significantly lower than the middle and the south (control vs. south, $p < 0.05$) cut points, but not significantly lower than the north. It was far less common in the control when compared to the clearcut points (constituting 0.8% and 17.6% of all mites in control and clearcut, respectively). This species was also relatively uncommon in uncut forest during preliminary sampling in 1994 (3.1% of all mites found) (Nadel 1995), and therefore probably represents a species that becomes more dominant in response to tree removal. Although the Oppiidae were far more common in the control than in the cut, the difference was not statistically significant when an anova was performed. This was probably because of high inter-sample variability. This family includes species also known to increase in response to disturbance (V. Behan-Pelletier, Agriculture-Agrifood Canada, pers. comm., 1996).

Springtail distributions among the clearcut points and control were also generally similar (Table 3). Isotomidae and Onychiuridae were not significantly different across the transect points (anova, $p = 0.4733$ and $p = 0.0636$, respectively), but Hypogastruridae showed significant response (anova, $p = 0.0079$) to transect location, stemming from a significantly higher density in the control compared with the middle of the cut (Tukey-Kramer multiple comparisons test, $p < 0.01$). This family, therefore, seems negatively affected by clearcutting, at least away from the edge of the clearcut, and at least during the first year.

Mite and springtail families represented by at least 100 individuals in the samples were categorized by their substrate preferences (Table 4). In all cases densities followed a distinct trend, with forest floor densities greater than decayed wood densities, and decayed wood densities greater than mineral soil densities. The families were roughly divided into those which showed a clear preference for forest floor over the other two substrates, those which showed equal preference for forest floor and decayed wood over mineral soil, and a few families of mites and springtails that showed no clear preference for any substrate.

Nine of the 15 common mite families and one springtail family were represented in decayed wood in numbers as high as in the forest floor and higher than in mineral soil. In future years, as the remaining forest floor changes in quantity and quality in the clearcut, these families in particular may be able to survive in decayed logs remaining in clearcuts and to have the opportunity to recolonize the new forest floor if trees regenerate.

TABLE 4 *Substrate preferences of the common mite and springtail families found in Sicamous Creek soil*

	Family	Substrate
Preference for forest floor		
Mites	Zerconidae	ff[a] dw ms
	Brachychthoniidae	ff dw ms
	Camisiidae	ff dw ms
	Tarsonemidae	ff dw ms[b]
	Tectocepheidae	ff dw ms
	Uropodidae	ff dw ms
	Eupodidae	ff dw ms
Springtails	Isotomidae	ff dw ms
	Hypogastruridae	ff dw ms
Preference for forest floor and decayed wood		
Mites	Acaridae	ff dw ms
	Nanorchestidae	ff dw ms
	Oppiidae	ff dw ms
	Scutacaridae	ff dw ms
	Suctobelbidae	ff dw ms
	Tydeidae	ff dw ms
No clear preference		
Mites	Alicorhagiidae	ff dw ms
	Rhagidiidae	ff dw ms
Springtails	Onychiuridae	ff dw ms

[a] ff = forest floor; dw = decayed wood; ms = mineral soil.
[b] Underlined substrates are not significantly different from each other ($p \leq 0.05$).

CONCLUSIONS

Among the three substrates studied, the forest floor harbours the majority of soil mites and springtails, followed by decayed wood, and with fewest found in the mineral soil. During the first year after clearcutting, the overall abundance of soil mites and springtails showed little change in a 10-ha clearcut at Sicamous Creek compared with an uncut control. One mite family, the Tectocepheidae, increased significantly in the middle and south transect points of the cut when compared to the control, and appears to respond positively to disturbance. In contrast, the springtail family Hypogastruridae decreased significantly in the middle of the clearcut when compared to the control, and appears to respond negatively to clearcutting, at least away from the edge of the cut. A preliminary analysis of the diversity of mite and springtail communities revealed no significant differences between the control and clearcut transect points. Decayed wood contains high numbers of some soil micro-arthropods and may therefore provide refugia for these groups in clearcuts if the forest floor disappears or becomes unsuitable in later years.

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Influence of Forest Harvesting on the Nutrient Status of ESSFwc2 Ecosystems

MICHAEL FELLER

ABSTRACT

In Engelmann Spruce – Subalpine Fir (ESSFwc2) forests in the Clearwater Forest District, I recently found that solution inputs (precipitation) of nitrogen were balanced by solution outputs (mineral soil leachate). Because of the lack of nitrogen in bedrock, the amount of this element present in the ecosystem could only be explained by nitrogen fixation. Whole-tree harvesting by clearcutting results in relatively large nitrogen losses, made even larger by site preparation techniques such as mechanical scarification or slashburning. To see if these results were applicable to ESSFwc2 ecosystems elsewhere, a study was started in 1994 at the Sicamous Creek Silvicultural Systems research area. This study will quantify gaseous nitrogen fluxes and the effects of different harvesting systems on ecosystem nutrient (including nitrogen) status. Results to date suggest, as in Clearwater, that no net input of nitrogen from precipitation occurs in the Sicamous forests, and that clearcutting increases the flux of nutrients leached out of the soil. The effects of timber harvesting at the Sicamous Creek research area on nutrient stocks are tentatively estimated and future work is described.

INTRODUCTION

Nutrient availability is a major determinant of forest productivity, with nitrogen the major growth-limiting nutrient in British Columbia's forests (e.g., Edmonds et al. 1989). Timber harvesting results in nutrient losses from a site primarily by log removal and enhanced leaching through soil. The effect of timber harvesting on nutrient losses and on ecosystem nutrient status has been the subject of considerable research during the last 25 years, particularly with the advent of whole-tree harvesting (e.g., Kimmins 1977; Anon. 1979; Edmonds et al. 1989). Studies have generally shown that the degree of vegetation removal, the severity of subsequent site preparation, rotation length, and inherent site fertility are the major factors determining the significance, from a forest productivity viewpoint, of the effect of timber harvesting on ecosystem nutrient status.

Nitrogen is not only the major growth-limiting nutrient in British Columbia's forests, but it is also the most difficult nutrient to assess because of its involvement in gaseous and microbially mediated transformations in forests (e.g., Clark and Rosswall 1981). Only one study has produced a gaseous nitrogen flux budget in British Columbia and that was for a single-aged Coastal Western Hemlock (CWH) biogeoclimatic zone forest in southwestern British Columbia (Cushon and Feller 1989).

The Engelmann Spruce – Subalpine Fir (ESSF) zone forests in the interior of British Columbia are important for producing wood fibre. Timber harvesting has become a major use of the forests, yet relatively little is known about the effects of such harvesting on nutrients and the long-term productivity of the forests. These forests grow under severe environmental conditions and have relatively slow growth rates (e.g., Farnden 1994). Under such conditions, any declines in productivity induced by harvesting could have major consequences for future timber supplies.

The nutrient status of ESSF forests has been studied relatively little, even in the United States. Arthur and Fahey (1990, 1992) described biomass and nutrient pools and fluxes in an undisturbed Engelmann spruce – subalpine fir forest in Colorado, while Stump and Binkley (1993) quantified nitrogen release from litter decomposition in another such forest in Colorado. Jurgensen et al. (1987, 1991) studied nitrogen fixation in forests of Engelmann spruce – subalpine fir/Douglas-fir/western larch in Montana. Snell et al. (1979) quantified the nutrient content of vegetation and soils in an Engelmann spruce – subalpine fir forest in Colorado, and the nutrient losses in logs that would result from different harvesting systems.

In Canada, Prescott et al. (1989a) quantified the biomass and the dynamics of nitrogen and phosphorus in an undisturbed Engelmann spruce – subalpine fir forest in the southern Rocky Mountains of Alberta. They also examined nutrient fluxes in litterfall and litter decomposition in this forest (Prescott et al. 1989b, 1993). I have studied biomass and nutrients in wetter ESSF forests in the Clearwater Forest District, and the effects on these of forest harvesting and site preparation (Feller and Hamilton 1994). The organic matter and nutrients contained in the Clearwater forests and lost as a result of forest harvesting and site preparation are shown in Table 1.

At Clearwater, nutrient loss was the least with clearcutting only, increased with clearcutting and slashburning, and was highest with clearcutting and mechanical scarification. The magnitude of the losses for a given treatment was generally greater in subhygric than in mesic ecosystems. However, in the case of clearcutting and clearcutting plus slashburning, losses of nitrogen, sulphur, and potassium were relatively greater in the mesic ecosystem; phosphorus and magnesium losses were relatively greater in the subhygric system; and calcium and iron losses were similar in both ecosystems, when expressed as a percentage of the undisturbed forest ecosystem nutrient content (Table 2). Copper losses and contents were too small to allow a reliable comparison between the two ecosystems. In the case of clearcutting followed by a mechanical scarification, which removed relatively similar amounts of organic materials in both the mesic and subhygric ecosystems, nitrogen, phosphorus, sulphur, and potassium losses were greater in the mesic ecosystem, while magnesium, calcium, and iron losses were greater in the subhygric ecosystem (Table 2).

TABLE 1 *Organic matter and nutrients contained in ESSFwc2 forests in the Clearwater Forest District and estimated losses resulting from clearcutting and site preparation*[a]

Treatment/	Organic matter	Nutrients (kg/ha)							
component	(kg/m^2)	N	P	S	K	Mg	Ca	Fe	Cu
				Subhygric Ecosystems					
Undisturbed forest									
Trees - living	32.9	545	95	105	435	70	688	22	2
Trees - dead	5.9	54	5	11	444	17	101	3	0
Understorey vegetation	0.5	46	6	7	28	9	13	1	0
Dead woody material	10.8	89	5	36	36	19	90	4	0
Forest floor	5.1	691	48	65	108	35	250	124	1
Roots (understorey)	0.3	14	2	2	7	2	4	1	0
Mineral soil[b]	17.6	3537	28	358	88	35	168	160	9
Total	73.1	4976	189	584	1146	187	1314	315	13
Loss from clearcutting only									
Tree removal	23.0	380	67	61	232	72	514	14	2
Soil leaching	0.0	3	0	3	2	2	3	0	0
Total	23.0	383	67	64	234	74	517	14	2
Loss from clearcutting and slashburning									
Tree removal	23.0	380	67	61	232	72	514	14	2
Soil leaching	0.0	4	0	11	25	2	10	0	0
To the atmosphere	4.3	139	17	31	27	21	135	18	0
Total	27.3	523	84	103	284	95	659	32	2
Loss from mechanical scarification									
Tree removal	23.0	380	67	61	232	72	514	14	2
Bulldozer scarification	15.9	728	56	83	157	65	345	97	1
Soil leaching	0.0	6	0	1	14	3	11	0	0
Total	38.9	1114	123	145	403	140	870	111	3
				Mesic Ecosystems					
Undisturbed forest									
Trees - living	24.6	440	77	69	346	69	538	6	1
Trees - dead	3.1	31	3	6	24	9	54	2	0
Understorey vegetation	0.5	133	38	17	55	8	40	1	0
Dead woody material	9.3	37	2	15	17	8	38	2	0
Forest floor	3.8	435	23	52	52	20	128	91	0
Roots (understorey)	0.1	3	0	0	1	0	1	0	0
Mineral soil[b]	12.1	1997	23	187	48	23	102	153	1
Total	53.5	3066	166	346	543	137	901	255	3
Loss from clearcutting only									
Log removal	14.2	270	48	42	210	47	349	9	1
Soil leaching	0.0	6	1	12	2	0	3	0	0
Total	14.2	276	49	54	212	47	352	9	1
Loss from clearcutting and slashburning									
Log removal	14.2	270	48	42	210	47	349	9	1
Soil leaching	0.0	10	5	11	29	1	2	0	0
To the atmosphere	3.7	210	14	30	70	12	133	17	0
Total	17.9	490	67	83	309	60	484	26	1

[a] Data are from Feller and Hamilton (1994).

[b] Mineral soil nutrient quantities are total quantities for organic matter, N, and S, and extractable quantities for all other nutrients. Leaching losses are from below the rooting zone over a four-year post-treatment period.

TABLE 2 *Percentage of pre-harvesting ecosystem nutrient content lost through harvesting and site preparation in the Clearwater ESSFwc2 forests*[a]

Ecosystem/ treatment	Organic matter	N	P	S	K	Mg	Ca	Fe	Cu
			Subhygric Ecosystem						
Clearcutting only	31	8	35	11	20	40	39	4	15
Clearcutting and slashburning	37	11	44	18	25	51	50	10	15
Clearcutting and mechanical scarification	53	22	65	25	35	75	66	35	23
			Mesic Ecosystem						
Clearcutting only	27	9	30	16	39	34	39	4	33
Clearcutting and slashburning	33	16	40	24	57	44	54	10	33
Clearcutting and mechanical scarification[b]	45	27	70	35	71	73	55	31	33

[a] Percentages were calculated from data in Table 1. Data are from Feller and Hamilton (1994).

[b] Estimated assuming the same percentage removals of organic matter components as for mechanical scarification in the subhygric ecosystem.

The three significant results from this analysis were:

1. mechanical scarification caused greater losses than did the slashburning;
2. losses of nitrogen tended to be higher in mesic than in subhygric ecosystems, for a given treatment; and
3. leaching losses tended to be relatively minor compared to losses by other pathways.

Therefore, when considering long-term productivity, low-severity treatments in subhygric ecosystems, with relatively large mineral soil nutrient reserves, are less likely a concern than high-severity treatments in mesic ecosystems with lower nutrient reserves. In the case of nitrogen, timber harvesting followed by a typical low- to moderate-severity slashburn, is likely to leave behind 2000–3000 kg/ha of nitrogen reserves in a subhygric ecosystem (data from Feller and Hamilton 1994). In a mesic ecosystem, timber harvesting followed by a low- to moderate-severity slashburn, is likely to leave behind 1000–2500 kg/ha of nitrogen reserves. While such levels should sustain another rotation, growing trees are likely to require 400–800 kg N/ha (data from Feller and Hamilton 1994) leaving little in reserve, and inputs to the system during that rotation will probably be very low. Precipitation inputs of 0.6 kg/ha per year at Clearwater were essentially the same as soil solution outputs. Nitrogen fixation is uncertain, but might not be more than 1 kg/ha per year, given the relative absence of symbiotic nitrogen-fixing plants and the low rates of asymbiotic fixation found elsewhere (e.g., Edmonds et al. 1989). As well, weathering inputs of nitrogen are probably negligible (e.g., Edmonds et al. 1989). A similar loss of nutrients at the end of the next rotation could well leave insufficient nitrogen reserves to sustain a subsequent rotation.

These nitrogen budget data raised questions about the long-term sustainability of timber harvesting in these ESSF ecosystems. If we ignore the issue of site preparation and consider the mesic ecosystems at Clearwater, nitrogen losses from timber harvesting alone were approximately 300 kg/ha. Net annual nitrogen inputs could be less than 1 kg/ha, based on the discussion above. Consequently, these losses are unlikely to be replaced in 300 years, suggesting that rotations of less than 300 years could lead to nitrogen depletion, in the absence of any mitigation strategy.

My major conclusions from the Clearwater study were:

- Mesic ecosystems are more vulnerable to nutrient depletion than subhygric ecosystems.
- If nitrogen inputs are as low as estimated, then to sustain timber production, forest management must aim to minimize nitrogen losses and maximize nitrogen inputs.
- Site preparation treatments that cause relatively large losses of organic matter and nutrients should be avoided. Extensive mechanical scarification, such as that applied in this study area, should definitely be avoided (Feller and Hamilton 1994).

These conclusions are contingent on relatively low net gaseous inputs of nitrogen. They would have major implications for the management of wetter ESSF forests if they applied generally to all such forests. The generality of these results and quantification of gaseous nitrogen fluxes in ESSF forests are currently being studied in the Sicamous Creek Silvicultural Systems research area.

OBJECTIVES

The major objective of the study is to assess the effect of forest harvesting on the long-term nutrient, particularly nitrogen, budgets of wetter ESSF ecosystems.

This major objective will be realized by achieving the following objectives:

- to determine nutrient inputs in precipitation and outputs in soil leachate in both undisturbed and harvested ESSF forests;
- to quantify nutrient inputs by geological weathering into the ESSF forests;
- to quantify gaseous nitrogen fluxes (nitrogen fixation and denitrification) in a chronosequence of ESSF forests; and
- to quantify nutrient outputs in logs or through site preparation.

METHODS

The study is occurring in and around the Sicamous Creek Silvicultural Systems research area in the Salmon Arm Forest District, Kamloops Forest Region. To minimize variability in the field, all measurements are occurring in zonal subalpine fir – azalea – oak fern ecosystems or mosaics of the zonal and subalpine fir – valerian – oak fern ecosystems (Lloyd et al. 1990). A graduate student (Reinhold Posmyk) is conducting the gaseous nitrogen flux component of the study.

Field Methods

The field methods used to achieve the specific objectives of the study are outlined below.

Objective 1: Nutrient fluxes in precipitation and soil leachate Nutrient fluxes in precipitation and soil leachate were, and will be, determined from estimates of water fluxes and chemical analysis of samples of both types of solutions, collected regularly throughout the year for precipitation and during the snow-free portion of the year for soil leachate.

Precipitation water fluxes were obtained from data collected by other workers at the Sicamous Creek study area. Soil water fluxes were estimated from measurements of:

- precipitation, solar radiation, and air temperature, which were recorded at an on-site weather station; and
- throughfall in the mature forest, which was measured using 20 throughfall collectors (36 L plastic storage containers) during the snow-free period in 1994 and again (with 10 collectors) in 1995.

All the weather and throughfall measurements were used to calculate evapotranspiration, on a weekly basis, using a form of the Penman-Monteith equation (McNaughton and Black 1973; T.A. Black, Soil Science Department, University of British Columbia, pers. comm., 1993).

Precipitation samples for chemical analysis were, and will be, taken from both snow and rain. Snow samples were collected by scraping the face of a snow pit with a 1 L polyethylene bottle. Samples were collected in mid-winter and just before snowmelt in early spring. Rain samples were collected using two simple polyethylene funnel/collector systems in a clearing. Precipitation sampling began in March 1994, while throughfall measurements began in July 1994. These measurements will continue for a three-year post-treatment period (i.e., until October 1997).

Soil solution samples were collected using soil water extractors from Soil Moisture Inc., with extracting cups located just below the rooting zone (25 cm). These were evacuated to a vacuum of 0.5 bar and emptied approximately once per month when samples were collected. Ten collectors were located in an undisturbed control forest (block A1). Another 10 collectors were located in a forest that was subsequently clearcut (A4). Sampling began in June 1994. Sampling will continue for a three-year post-treatment period.

Objective 2: Geological weathering nutrient inputs Geological weathering inputs will be estimated using the nutrient flux balance equation:

$$W + P + GI = L + U + \Delta FF + \Delta MS + GO,$$

where: W = weathering inputs;
P = precipitation inputs;
GI = gaseous inputs;
L = leaching outputs;
U = plant uptake;
ΔFF = change in forest floor storage;
ΔMS = change in mineral soil storage; and
GO = gaseous outputs.

This equation will be applied to the undisturbed forests, where it is assumed that changes in forest floor and mineral soil storage are zero. While this may not be true over a long time period, any changes over a two- to three-year period are likely very small and essentially negligible.

Precipitation inputs and leaching outputs were determined as described above, while gaseous inputs and outputs (only of significance for nitrogen) will be determined as described below. Plant uptake will be determined as follows:

Mosses and herbs I assumed that these have no net increase in biomass, and hence nutrient uptake, over a two- to three-year period.

Shrubs Approximately 40 shrubs of each of the four dominant species (*Rhododendron albiflorum*, *Menziesia ferruginea*, *Vaccinium ovalifolium*, and *Vaccinium membranaceum*), were marked and measured for volume and height in 1995 and will be measured again in 1996 and 1997. Biomass of the shrubs will be calculated using regression equations (Feller and Hamilton 1994). Nearby shrubs of each species were destructively sampled and analyzed for nutrient concentrations (five samples per species).

Uptake is estimated as the change in biomass multiplied by its nutrient concentrations. This will be extrapolated on a per hectare basis by estimating the per hectare biomass of the shrubs. The latter was done by measuring the heights and volumes of all shrubs present in rectangular (5 × 2 m) plots, using 10 plots per block (A1 and B2). These measurements will be repeated in block C4.

Trees Uptake is estimated as the increment in tree component biomass over a three-year period, multiplied by the appropriate nutrient concentrations in each tree component. Approximately 30 trees of each species, covering a range of diameter classes, will be cored in 1997. Widths of the most recent three rings will be measured and used to develop a regression equation between increment and tree dbh for each species separately. These equations will be applied to the tree inventory data for undisturbed forest blocks that I collected in 1994 to estimate the dbh of each of the sampled trees in 1997. (The 1994 plots were not marked as they were not intended to be permanent.) Tree component biomass regression equations (Feller and Hamilton 1994) will then be applied to each of the 1994 and 1997 tree dbh data sets to estimate tree component biomass for each of these years. Tree component biomass increment is then calculated as the difference in component biomass between 1994 and 1997.

Uptake is estimated as the change in biomass multiplied by its nutrient concentrations. Nutrient concentrations were measured in tree biomass samples (three samples per species per component roots, stemwood, stembark, live branches, dead branches, foliage) collected in 1995. Additional sampling for chemical analysis occurred in 1996.

Objective 3: Gaseous nitrogen fluxes Gaseous nitrogen fixation and denitrification will be quantified in a forest chronosequence in the vicinity of Sicamous Creek. Four age classes will be studied (1–2, 5–10, 50–100, and > 200 years). Age classes 1–2 and > 200 years occur in the Sicamous Creek Silvicultural Systems study area.

Initially I had planned to measure nitrogen fixation and denitrification simultaneously, using the acetylene-ethylene assay for nitrogen fixation (Hardy et al. 1973) and the acetylene inhibition method for denitrification (Yoshinari et al. 1977). However, these methods have been heavily criticized and it is suggested that ^{15}N tracer methods are superior and more reliable (e.g., Benckiser 1994, Minchin et al. 1994). We will therefore use a tracer method using ^{15}N nitrogen gas to study the net rate of gaseous nitrogen input into the forests. Materials from the forest floor, mineral soil, foliage, and the bark of two tree species, three classes of decaying wood, and plant species that are known to fix nitrogen will be assessed monthly (from June through October). The nitrogen-fixing species include some lichens (*Peltigera*, *Stereocaulon*, and *Pannaria* spp.) and lupines. Assessments began in 1996 and will finish in 1997.

Samples of each stratum were collected for incubation with ^{15}N. When these incubations are complete, sample dry mass is determined and fixation and denitrification rates expressed on a unit area basis using the estimated mass of each stratum present per unit area. Forest floor mass was obtained by destructive sampling. Mineral soil mass (surface 20 cm only) was obtained from data collected by other workers at the Sicamous Creek study area or from destructive sampling from sites not in the Sicamous Creek study area. Tree foliage and bark mass is estimated as described above under objective 2. Decaying wood mass was obtained from estimates of coarse woody debris I made for the research area. We also assessed study sites outside this research area to obtain estimates of coarse woody debris. Surveys will determine the mass and percentage cover of nitrogen-fixing lichens and lupines. Also some destructive sampling will take place to estimate biomass in relation to cover.

Objective 4: Nutrient outputs in logs and through site preparation

Nutrient losses in harvested trees are determined by subtracting the nutrient content of trees and woody material present after harvesting from that present before harvesting. Losses from slashburning are estimated by subtracting the post-burn nutrient quantities in slash, forest floor, and vegetation from the pre-burn values.

The pre-harvesting nutrient content of trees is determined by multiplying estimates of tree component biomass (obtained by applying biomass regression equations [Feller and Hamilton 1994] to tree inventory data I collected in 1994) by their appropriate nutrient concentrations (determined as described above under objective 2). The pre-harvesting nutrient content of coarse woody debris is determined by multiplying estimates of its biomass (which I obtained in 1994 [Feller et al. 1994]) by its nutrient concentrations, on a diameter size class and species/decay class basis. Six samples of each size class for each species/decay class were collected and analyzed in the laboratory. Average chemical concentrations will be used to calculate nutrient content.

The post-harvesting nutrient content of trees and woody material is determined by multiplying the biomass of these materials by their nutrient concentrations. The biomass was estimated from measurements of stumps and slash made the first year after harvesting (1995). Stump biomass was estimated from field measurements of stump volumes and measurements of stem relative densities, on a species-specific basis (using densities I obtained at Clearwater [Feller and Hamilton 1994]).

Slash biomass is estimated using:

- the line intersect method (Van Wagner 1968) and estimates of slash material relative density I obtained for materials greater than 1 cm in diameter (Feller and Hamilton 1994; Feller et al. 1995); and
- destructive sampling of 1-m^2 plots for materials less than or equal to 1 cm in diameter (including needles).

The destructive sampling occurred using 10 randomly located plots per block. Sampled materials were oven-dried (70°c) then weighed.

Forest floor biomass was obtained as described above under objective 3. Post-burn biomass is estimated from measurements of forest floor density and depth of burn, as described in Feller and Hamilton (1994).

Understorey vegetation biomass is estimated for shrubs, as described under objective 2 above, and for herbs and ferns by destructive sampling of 10 randomly located 1-m^2 plots.

Stump nutrient content is estimated by multiplying its mass (on a species-specific basis) by its nutrient concentrations (obtained as described above under objective 2). Slash nutrient content is estimated by multiplying its mass (on a size- and species/decay-class basis) by its nutrient concentrations (obtained as described above for coarse woody debris). Forest floor nutrient content is estimated by multiplying its mass by its nutrient concentrations, obtained by analyzing the samples collected for mass determination. An additional 10 randomly located samples per burn site will be collected immediately after burning for post-burn nutrient concentration analyses. Shrub nutrient content is obtained as described above under objective 2. Herb and fern nutrient content is obtained by multiplying their mass by their nutrient concentrations. Pre- and post-burn understorey nutrient contents are estimated as described in Feller and Hamilton (1994).

Laboratory Methods

All precipitation and soil water samples are analyzed for pH and electrical conductivity (standardized to 25°c) as soon as possible after collection, using pH and conductivity meters, respectively. If necessary, the samples are stored in a freezer before being analyzed for potassium, magnesium, and calcium by atomic absorption spectrophotometry, and for ammonium, nitrate, sulphate, and phosphate using colorimetric methods on a Technicon traacs 800 continuous flow analyzer. Organic phosphorus and organic nitrogen are measured using an alkaline persulphate digestion technique (D'Elia et al. 1977).

For chemical analysis, all plant material samples are oven-dried at 70°c and ground to pass a 1-mm mesh sieve. A portion of each sample is subjected to a Parkinson and Allen digestion (Parkinson and Allen 1975), then the digestion solution is analyzed for nitrogen (as ammonium) and P (as phosphate) on the traacs 800 instrument, and for potassium, magnesium, and calcium by atomic absorption spectrophotometry. Another portion of each sample is being analyzed for total sulphur using a Leco sc-32 sulphur determinator.

For objective 1, data for the 1993–94 water year (1 October–30 September) are not yet available because of problems in estimating snowpacks. However, the 1994–95 water year data for Sicamous Creek suggest that precipitation inputs (Table 3) are similar to those in the ESSF at Clearwater, although metallic cation (potassium, magnesium, and calcium) inputs are somewhat higher. As the Sicamous data are for one year and the other data are for longer time periods (an average of five years in the case of Clearwater) it is probably too soon to make comparisons.

Nutrient outputs in soil leachate (Table 4) suggest elevated levels following harvesting, although the increases are relatively low (probably all ≤ 12 kg/ha) when compared to likely losses in biomass removal. Of interest is the nitrogen flux in the undisturbed forest which, as at Clearwater, is essentially the same as the estimated precipitation input (Table 3).

Thus, the initial conclusions about nutrient fluxes in precipitation and soil leachate are:

- Nitrogen inputs in precipitation at Sicamous Creek are balanced by soil leachate outputs, as at Clearwater. This would suggest that the Sicamous Creek ESSF forests, like those at Clearwater, would depend on nitrogen fixation for their accumulation of nitrogen.

TABLE 3 *Annual nutrient inputs in precipitation (kg/ha) at the Sicamous Creek research area and other areas in British Columbia*[a]

Area	Precipitation quantity (mm)	K	Mg	Ca	NO_3-N	NH_4-N	Organic-N	Total-N	P
ESSF, Sicamous	1130	0.9	0.2	1.2	0.1	0.0	0.3	0.4	0.1
ESSF, Clearwater	1290	0.3	0.1	0.9	0.2	0.2	0.2	0.6	< 0.05
CWH, Vancouver Island	1650–2360	0.6–2.1	0.6–1.3	1.2–3.7	0.1–0.2	0.2–0.3	1.8–4.5	2.2–5.0	0.0–0.2
CWH, Lower Mainland	2020–2960	1.3–2.2	0.9–1.6	1.3–3.9	2.7–5.1	1.5–5.6	1.6–3.9	8.0–11.6	0.0

[a] ESSF, Sicamous data are for the 1994–95 water year from the present study. Other data are in Feller and Hamilton (1994).

TABLE 4 *Annual nutrient fluxes in soil leachate (kg/ha) at the Sicamous Creek research area for the 1994–95 water year*[a]

Area	Quantity (mm)	K	Mg	Ca	NO_3-N	NH_4-N	Organic-N	Total-N	P
Undisturbed forest	660	2.7	0.7	1.5	0.0	0.0	0.3	0.4	0.0
Clearcut	860	15.4	2.9	7.0	1.7	0.1	0.7	2.5	0.0

[a] For nitrogen, differences between total nitrogen and the sum of its constituents are due to rounding errors.

- Losses of nutrients in soil solution induced by harvesting are relatively low and comparable to those found elsewhere.

Additional data are needed before these conclusions are confirmed.

Work to address objectives 2, 3, and 4 (i.e., geological weathering; gaseous nitrogen fluxes; and nutrient outputs in logs and through site preparation) only began in 1995 and I have insufficient data to present any reliable results.

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Coarse Woody Debris in Forests: An Overview of the Coarse Woody Debris Study and the Sicamous Creek Study Area

MICHAEL FELLER

ABSTRACT

The characteristics and quantity of coarse woody debris (CWD) are being determined in the Sicamous forests pre- and post-harvesting and post-manipulation for a small mammal study. This is part of a larger study of coarse woody debris in British Columbia forest ecosystems. Quantities of coarse woody debris in the forests at Sicamous are similar to those found in other Engelmann Spruce – Subalpine Fir (ESSFwc2) zone forests, averaging around 6.5 kg/m², but are approximately half those found in old-growth Coastal Western Hemlock (CWH) zone forests. Most of this mass comes from the largest size class coarse woody debris, while most debris pieces come from the smallest size class. Coarse woody debris covers only 14% of the ground surface in the Sicamous Creek forests. It remains on the ground in these forests for an average of 320–390 years, intermediate between that for Interior Douglas-fir (IDF) zone forests (average of at least 80–190 years) and CWH forests (average of at least 780–810 years). The influence of this debris on small mammals, vegetation, and soils is being studied in three separate, but related, studies at the Sicamous Creek Silvicultural Systems research area.

INTRODUCTION

The importance of coarse woody debris (CWD) in a forest has been stressed by numerous people, both for its beneficial effects on forest productivity (e.g., Harvey et al. 1981; Maser et al. 1988; Perry et al. 1989) and as wildlife habitat (e.g., Elton, 1966; Thomas 1979; Maser et al. 1988; Ruggiero et al. 1991). A definitive review of the ecological role of coarse woody debris in forests, based on literature published to 1986, is presented by Harmon et al. (1986). They concluded that CWD is an important functional component of temperate forest ecosystems, but that our understanding of its true importance was rudimentary. They also considered that greater scientific attention should be given to its function in forest ecosystems.

Since 1986, some attention has been paid to CWD from a wildlife perspective (e.g., Ruggiero et al. 1991). However, only a few studies have quantified coarse woody debris and related it to the abundance of small mammals. Even less attention is paid to the relationships between CWD and forest soils and considerable uncertainty still exists about this topic. Thus, Harvey and co-workers (e.g., Harvey et al. 1981, 1989) have repeatedly emphasized the importance of decaying CWD to site productivity in drier sites, while Krajina (1969) suggested that in wetter coastal areas of the province, increased soil podzolization and loss of soil nutrients could occur under accumulations of decaying CWD.

Coarse woody debris is known to provide plant habitat, but little information exists on which plants are associated with it (Harmon et al. 1986). Although it is known to be an important habitat for *Tsuga heterophylla* in coastal western North America (e.g., Krajina 1969) and *Picea engelmannii* and *Abies lasiocarpa* at higher elevations in interior British Columbia (e.g., Smith 1955), the value of CWD as a substrate for all tree regeneration in western North America has been questioned (e.g., Dobbs 1972; Alexander et al. 1984). For at least some tree species growing on coarse woody debris, the ability to survive apparently depends on their roots forming an association with mycorrhizal fungi (Harmon et al. 1986). Relationships between understorey plant species and CWD are less well studied than those between tree species and CWD, although some lists of species are available (e.g., Harmon et al. 1986).

In summary, coarse woody debris is considered an important functional component of forest ecosystems, which influences soils, plants, and small mammals. However, our current knowledge of the inter-relationships between CWD and these other ecosystem components is incomplete. This is particularly so in British Columbia, where even the quantity present has been poorly documented. Harmon et al. (1986) presented no data for the province, while a more recent review by Trofymow and Beese (1990) found such data only from two unpublished theses. In addition, some other workers and I have some unpublished data.

Forest managers in British Columbia are under increasing pressure to maintain both long-term site productivity and biological diversity. They are also under similar pressure to modify harvesting and slash treatment practices. Such practices affect coarse woody debris, which in turn may affect long-term site productivity and biological (plant and small mammal) diversity. Understanding the inter-relationships between coarse woody debris and soils, plants, and small mammals is critical if forest managers are to prescribe practices which maintain long-term site productivity and biological diversity. Our current inadequate knowledge of these inter-relationships has provided the impetus for this study, which quantifies the amount and type of CWD and its influence on soils, plants, and small mammals in a variety of forest situations in the province.

OBJECTIVES

The general objective of the study is to improve our understanding of the ecological role of coarse woody debris in some interior British Columbia forest ecosystems to help forest managers maintain long-term site productivity and biological diversity.

Specific objectives of the study are:

- to quantify the amount and type of CWD in different forest ecosystems;
- to determine the influence of CWD on forest floor and mineral soil morphology and physical and chemical properties in the study areas;
- to determine the influence of CWD on the distribution and abundance of different plant species in the study areas;
- to determine the relationships between CWD and the abundance and diversity of small mammals in the study areas; and
- to determine how CWD influences the inter-relationships between soil properties, plant species distribution and abundance, and abundance and diversity of small mammals in the study areas.

METHODS

The four facets to this study and researchers involved in each are:

1. Quantification of CWD: M. Feller
2. Soil properties: G. Kayahara and K. Klinka
3. Vegetation: X. Song and M. Feller
4. Small mammals: V. Craig and T. Sullivan.

The study is occurring along two biogeoclimatic gradients, one in the interior and one on the coast. In the interior, the Opax Mountain Small Business Program Alternative Silvicultural Systems area was used for the Interior Douglas-fir (IDF) zone study area; the Sicamous Creek Silvicultural Systems research area was used for the Engelmann Spruce – Subalpine Fir (ESSF) zone study area; and an area near the Sicamous Creek site was used for the Interior Cedar-Hemlock (ICH) zone study area. The coastal transect includes the Coastal Douglas-fir (CDF) (Victoria's watersheds), the Coastal Western Hemlock (CWH) (Vancouver's watersheds), and the Mountain Hemlock (MH) (Cypress Bowl) biogeoclimatic zones. In all areas, zonal (mesic) or close to zonal ecosystems are being studied. The size and nature of the areas being studied varies according to the facet of the study. Sampling began in 1993 and will be completed in 1997.

The soil and vegetation studies took place only in undisturbed forests in all six biogeoclimatic zones. The soil study also assessed forests in the Sub-boreal Spruce (SBS) and Boreal White and Black Spruce (BWBS) zones. The small mammal study used both undisturbed forests and recent clearcuts in the IDF and ESSF zones only. Vanessa Craig discusses the results of this study separately at this workshop (see this proceedings, page 243). Coarse woody debris was quantified in all the soil, vegetation, and small mammal study plots.

Measurements

Amount and type of coarse woody debris In all study plots, CWD mass was quantified by size class and decay class using the line intersect method (Van Wagner 1968), and by wood density from each size and decay class. The line intersects involve three equilateral (30-m sides) triangles per plot.

Estimated wood volumes were converted to masses using wood relative densities. Relative densities were measured from the volume and dry mass of samples of each size class (1–3, 3–5, 5–7, 7–12, and > 12 cm diameter) for each wood material type (fresh spruce, fresh fir, decay classes I, II, and III). Decay class III material was rarely present for size classes less than 7 cm in diameter. Discs were cut from at least eight logs per size class and wood material type combinations. These discs, or sectors from the larger ones, then had their volumes (water immersion technique) and dry masses (oven drying at 100°c) determined. Porous materials were wrapped in plastic film before immersion in water for volume measurement.

The ground surface occupied by CWD, forest floor, rock, and mineral soil was determined using the transect lines established to estimate debris mass. The length of transect occupied by each of these materials was recorded and converted to a percentage. At the Sicamous Creek research area, a total of 270 m of transect line in each of six blocks (for a total of 1620 m) was used.

To determine the length of time that CWD remains separate from the forest floor, four samples of the most well-decayed CWD (decay class IV of Maser et al. 1988 or decay class III of the present study), were collected from the study plots. These samples were carefully collected from the outermost 5 mm of wood in areas where the wood was not penetrated by plant roots. Radiocarbon (^{14}C) dating of these samples was conducted by accelerator mass spectrometry at the Iso Trace Radiocarbon Laboratory, University of Toronto. Two subsamples from each sample were analyzed. The sample ages were determined after calibration with the bidecadal data set intcal93.14c of the uncalibrated conventional radiocarbon dates using the Libby ^{14}C mean life of 8033 years.

Decay classes used in the present study are:

- Fresh: Tree has recently fallen and is essentially intact with fine twigs (0.5 cm in diameter) present. Dead foliage may be present.
- Decay class I: Bark is intact; branches 2 cm in diameter are present; wood is original colour; and tree retains its original shape. No fine twigs or foliage are present.
- Decay class II: Bark is loose or has partly fallen off; few branches, only those > 5 cm in diameter are present; wood is original colour or faded, and may be partly soft; and tree retains its original shape.
- Decay class III: Bark is absent; wood is at least partly soft and generally red brown to dark brown; no branches are present; tree has at least partly lost its original shape; and is at least partly embedded in the forest floor.

These decay classes compare to those of Maser et al. (1988) as follows:

Present study	Maser et al.
Fresh	Decay class I
Decay class I	Decay class I–II
Decay class II	Decay class II–III
Decay class III	Decay class IV–V

The decay classes of the present study were preferred to those of Maser et al. because fewer classes would be easier for inexperienced workers to recognize. The five-class system also presented problems when applied to forests with long-lived western redcedar and yellow-cedar because a given log could belong to decay class II or V depending on the criterion considered.

Coarse woody debris and soils In each biogeoclimatic zone, three study sites were used. At each site, 12 soil pedons of 1 × 1 m were located systematically. Each pedon consisted of a decay class III log, which was at least 30 cm in diameter and at least 50% incorporated into the forest floor on one side, and a forest floor without any large coarse woody debris accumulation on the other. A soil pit was dug through the pedon to expose the surface 10 cm of mineral soil.

Forest floors and mineral soils were described, samples taken of each from both sides of the pedon, as well as from the decaying log. Samples were air-dried, and forest floor samples were ground to pass through a 2-mm sieve to remove coarse fragments. All samples were analyzed for pH (in water), total carbon, total nitrogen, and mineralizable nitrogen. Mineral soil B horizon samples were also analyzed for three different forms of each of iron, aluminum, and silicon. Forest floor samples were also analyzed for lipids and humic and fulvic acid content. Data were analyzed by analysis of variance to assess the effect of coarse woody debris on the chemical and physical properties of the soils.

Coarse woody debris and vegetation The vigour and percentage cover of all plant species, both tree regeneration (up to 2 m high) and understorey species on different rooting substrata (decaying CWD and forest floor) were recorded using small (approximately 0.5 m^2), homogeneous (with respect to rooting substratum) plots (at least 10 small plots per substratum per study plot).

Coarse woody debris and small mammals Methods are described by V. Craig (this proceedings, page 243).

INITIAL RESULTS

Amount and Type of Coarse Woody Debris

Coarse woody debris mass in the Sicamous Creek forests was dominated by the largest size class and decay class II logs (Table 1). However, the smallest size class and decay class II materials dominated the numbers of pieces of CWD lying on the ground (Table 2). Decay class III materials constituted only 13% of the number of pieces, but 28% of the mass, as they were present only in the larger size classes. Decay class III material, however, comprised 50% of the CWD that covered the ground surface (i.e., material that is not suspended above the surface, but lies on the surface) (Table 3). Coarse woody debris covered only 14% of the ground surface in the mesic-subhygric forests, however. This surface was composed primarily (85%) of forest floor (Table 3) .

TABLE 1 *Mass (kg/m^2) of coarse woody debris in the undisturbed forests in the Sicamous Creek research area*

	Coarse woody debris type				
Size class (cm)	**Fresh**	**Decay class I**	**Decay class II**	**Decay class III**	**All types**
1.0–3.0	0.01 (0.01)[a]	0.04 (0.02)	0.07 (0.02)	0.00 (0.00)	0.12 (0.03)
3.1–5.0	0.01 (0.01)	0.03 (0.01)	0.04 (0.01)	0.00 (0.00)	0.08 (0.01)
5.1–7.0	0.00 (0.01)	0.02 (0.01)	0.04 (0.02)	0.00 (0.01)	0.06 (0.03)
7.1–12.0	0.02 (0.04)	0.08 (0.04)	0.15 (0.04)	0.03 (0.01)	0.28 (0.09)
> 12.0	0.20 (0.41)	1.39 (0.64)	2.56 (1.15)	1.77 (0.32)	5.92 (1.48)
Total	0.24 (0.40)	1.56 (0.67)	2.85 (1.10)	1.80 (0.28)	6.45 (1.21)

[a] Standard deviations are in parenthesis ($n = 54$).

TABLE 2 *Average number of coarse woody debris pieces per 100 m of transect line in the undisturbed forests in the Sicamous Creek research area*

	Coarse woody debris type				
Size class (cm)	**Fresh**	**Decay class I**	**Decay class II**	**Decay class III**	**All types**
1.0–3.0	3	21	36	0	60
3.1–5.0	0	3	6	0	9
5.1–7.0	0	1	2	0	3
7.1–12.0	1	2	4	1	8
> 12.0	1	4	13	13	31
Total	5	31	61	14	111

TABLE 3 *Ground surface cover (%) in the Sicamous Creek research area forests*[a]

			Coarse woody debris				
Forest floor	**Rock**	**Mineral soil**	**Fresh**	**Decay class I**	**Decay class II**	**Decay class III**	**Total**
85 (5)[b]	1 (1)	0 (0)	0 (1)	2 (2)	5 (3)	7 (4)	14 (5)

[a] Data represent the average cover determined from 270 m of transect lines in each of blocks A1, A4, B2, B5, C3, and C4.

[b] Standard deviations are in parenthesis ($n = 6$).

The mass of CWD in the Sicamous Creek forests is very similar to its mass in other ESSFwc2 forests assessed in the Clearwater Forest District (Table 4). No major differences occur in CWD mass between mesic and subhygric ecosystems in the ESSFwc2. The forests assessed at Sicamous Creek were essentially mosaics of these two ecosystems.

When comparing CWD in different biogeoclimatic zones, a trend of increasing mass, moving from IDF to the ICH to the ESSF (Table 5) is apparent in the interior transects. The same trend is not apparent in the coastal transects, where maximum CWD mass occurs in the middle elevation CWH zone (Table 5). The ESSF forests had relatively little fresh CWD when compared to the other zones sampled. Overall, the driest zone sampled (IDF) had the least amount of CWD, which may be a function of climate and fire history.

The length of time that large CWD logs remain on the ground in the forests in the six biogeoclimatic zones studied is indicated by the radiocarbon dating results in Table 6. The trend in length of time parallels the trend in CWD mass, with length of time increasing with mass. The correlation coefficient between date of death in Table 6 (using the average of the range for each of the ICH, ESSF, and CDF) and the total CWD mass in Table 5 is a notably high 0.99. More intensive sampling would be required to confirm this. In the IDF zone, the least amount of CWD mass lasts for the shortest period of time, while the greatest amount and the longest-lasting material (presumably western redcedar) occurs in the CWD zone. Coarse woody debris in the Sicamous Creek forests is intermediate between these extremes, with large logs remaining separate from the forest floor for an average of 320–390 years (range 190–580 years).

TABLE 4 *Mass (kg/m^2) of coarse woody debris in undisturbed ESSFwc2 forests in the Sicamous Creek research area and in the Clearwater Forest District*

Size class (cm)	**Sicamous Creek (*n* = 18)[a]**	**Clearwater**	
		Subhygric ecosystems (*n* = 17)	**Mesic ecosystems (*n* = 23)**
1.0–3.0	0.12 (0.03)[b]	0.17 (0.02)	0.15 (0.01)
3.1–5.0	0.08 (0.01)	0.07 (0.01)	0.09 (0.01)
5.1–7.0	0.06 (0.03)	0.05 (0.01)	0.08 (0.01)
7.1–12.0	0.28 (0.08)	0.18 (0.02)	0.28 (0.05)
> 12.0	5.92 (1.23)	6.05 (0.58)	5.80 (0.67)
Total	6.45 (1.21)	6.52 (0.56)	6.40 (0.67)

[a] *n* = number of stands assessed.
[b] Standard deviations are in parenthesis.

TABLE 5 *Mass (kg/m^2) of coarse woody debris in undisturbed forests in zonal ecosystems in six biogeoclimatic zones in British Columbia*

Biogeoclimatic zone	**Coarse woody debris type**				
	Fresh	**Decay class I**	**Decay class II**	**Decay class III**	**All types**
IDF (*n* = 9)[a]	1.28 (2.57)[b]	0.07 (0.18)	0.74 (0.63)	0.65 (0.66)	2.74 (2.71)
ICH (*n* = 9)	0.59 (1.17)	0.85 (0.75)	1.63 (1.47)	1.70 (0.98)	4.76 (1.95)
ESSF (*n* = 18)	0.24 (0.47)	1.56 (0.67)	2.85 (1.10)	1.80 (0.28)	6.45 (1.21)
CDF (*n* = 12)	1.12 (1.04)	1.00 (0.73)	4.13 (4.75)	1.03 (0.44)	7.28 (5.24)
CWH (*n* = 9)	1.39 (1.49)	0.31 (0.44)	11.25 (13.77)	3.39 (3.45)	16.34(15.50)
MH (*n* = 9)	1.05 (1.01)	1.59 (1.22)	3.84 (2.95)	1.82 (1.55)	8.30 (3.52)

[a] *n* = number of stands assessed.
[b] Standard deviations are in parentheses.

TABLE 6 *Radiocarbon dating of death of well-decayed coarse woody debris samples from the Opax Mountain (IDF zone), Sicamous Creek (ESSF zone), and other study areas in British Columbia along a coastal and an interior biogeoclimatic zone transect*

Area/biogeoclimatic zone	**Sample no.**	**Date of death (years before present)**	
Interior			
Interior Douglas-fir	1	90 or 100[a]	
(Opax Mountain)	2	70, 170, 270 or 290	
	3	70, 170, 270 or 290	
	4	90	$\bar{x}$ = 80–190
Interior Cedar-Hemlock	1	200, 210 or 320	
	2	200, 210 or 320	
	3	190, 220 or 320	$\bar{x}$ = 195–320
Engelmann Spruce – Subalpine Fir	1	340	
(Sicamous Creek)	2	190, 210, or 320	
	3	190, 210, or 320	
	4	580	$\bar{x}$ = 320–390
Coast			
Coastal Douglas-fir	1	180, 260, or 300	
	2	180, 260, or 300	
	3	180, 260, or 300	
	4 (*Thuja plicata*)	710	$\bar{x}$ = 310–400
Coastal Western Hemlock	1	1200	
	2	550	
	3	600, 660 or 670	$\bar{x}$ = 780–810
Mountain Hemlock	1	200, 210 or 320	
	2	720	
	3	370, 390 or 480	
	4	360, 450 or 460	$\bar{x}$ = 410–490

[a] When two or three dates are given for the same sample, the radiocarbon dating method was unable to distinguish between them.

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Ecological Niches of Seedling Establishment in High-elevation Forests

PETER BRANG

ABSTRACT

The concept of the ecological niche for regeneration of tree seedlings in high-elevation forests is discussed. This concept seems equally useful for high-elevation forests in the European Alps and in North American mountainous ranges. It helps to explain the clumpy stand structure of many high-elevation forests. Mainly based on a Swiss example, the variations of the suitable microsite over space, time, and developmental stages are discussed. Since survival and growth of tree seedlings vary between microsites, microsites should be considered in forest research and management.

INTRODUCTION

Switzerland has a history of over 100 years of clearcut-free forestry. In that time, existing selection-cutting systems were refined, and new selection systems developed. These systems allow for continuous harvesting while maintaining continuous forest cover. Without continuous regeneration, the Swiss high-elevation forests could not protect people and assets against avalanches, landslides, and floods. Regeneration therefore is a key silvicultural issue in most Swiss high-elevation forests. About 73% of the Swiss forests are in mountainous regions, with a high proportion in protected forests, and about 27% are located in high-elevation subalpine forests (Eidgenoessische Anstalt fuer das forstliche Versuchswesen 1988). Since silvicultural practices in a given stand depend on various ecological and socioeconomic factors, solutions can not be transferred to another location. However, some of the general concepts applied to Swiss high-elevation forests might be useful in similar areas of British Columbia. One of these concepts is that of the ecological niche.

The Ecological Niche

An ecological niche is a set of environmental conditions that meet the demands of a given species. It thus represents a subset of the environmental conditions present in a given ecosystem. Similar terms include *suitable* or *favourable microsite*, *safe site* (Harper et al. 1965), and *regeneration niche* (Grubb 1977). The term "microsite" focuses more on the visible features of an ecological niche and less on environmental conditions. A description of an ecological niche could include minimum and maximum temperatures, whereas a microsite description might include micro-topographic features or forest structures associated with those temperatures.

The environmental conditions important for seedling establishment include many factors such as precipitation patterns, soil and air temperatures, light conditions, snow cover, and competing vegetation (e.g., Farnden 1994; Rochefort et al. 1994). By reducing the complexity of the environment to only two factors, moisture and temperature, the meaning of the "ecological niche" can be visualized (Figure 1).

Depending on the species in question and on the ecosystem, the ecological niche can be wide, narrow, or even absent. The ecological niche for short-term survival of a given species on a given site is often wider than the niche for long-term survival (Figure 1). In many lowland and montane forests, ecological niches for one or several dominant tree species are wide, at least during the regeneration phase (Burschel et al. 1985). Most of the forest floor is available for tree seedling establishment. Often, a seedling bank of shade-tolerant tree species exists on the ground (Burschel et al. 1985). In contrast to lowland and montane conditions, the ecological niche for tree seedling establishment above the treeline is absent. No microsites meet the physiological requirements of tree seedlings (e.g., because the growing period is too short) (Tranquillini 1979). However, many microsites are available for herbs and shrubs that are adapted to this harsh environment.

High-elevation forests should not be seen as homogeneous environments, but as a mosaic of suitable and unsuitable microsites for trees (Bischoff 1987). The environmental conditions on some microsites are close to those above the treeline and are therefore too harsh for tree seedling establishment (Ott 1995).

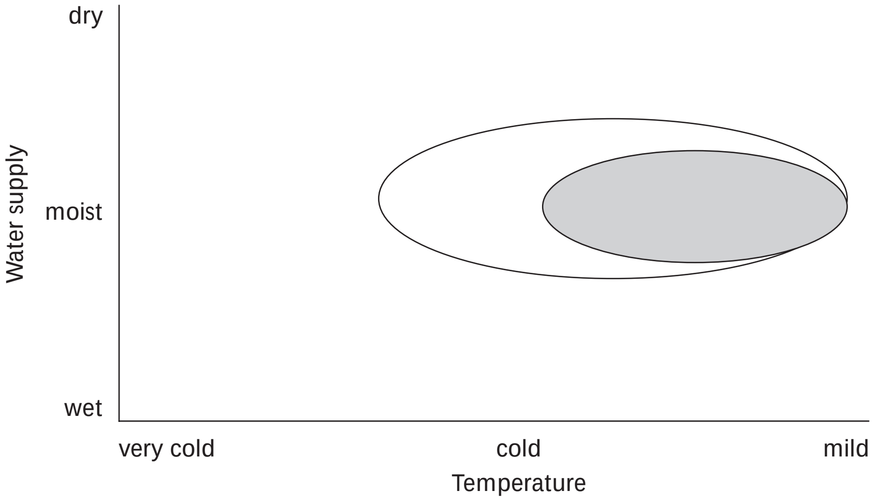

FIGURE 1 *Simplified ecological niche for a species in a ecosystem. Only two of the numerous ecological factors are presented, namely water supply and temperature. The ecological niche for short-term survival (blank) is often wider than the niche for long-term survival (grey).*

Why Are Subalpine Forests Clumpy?

The mosaic of suitable and unsuitable microsites for tree establishment translates into the clumpy stand structure found in most subalpine forests (Alexander 1986; Ott et al. 1991). Seedlings successfully establish only on suitable microsites (Shea 1985), and thus appear in aggregations or clumps (Penev 1961). The clumpyness usually increases with elevation (Ott et al. 1991). In high-elevation forests, we can not expect the same stocking and spatial eveness as in lowland and montane forests. A clumpy structure has important long-term benefits, since clumps are more likely to resist storm and snowload impact (Schoenenberger et al. 1990).

While suitable microsites for tree seedlings in high-elevation ecosystems may cover only 10% of the forest floor, this does not translate into similarly low stand density because the crown of mature trees extends beyond the surface of the original microsite (cf., Shea 1985). This is also true in any ecosystem where regeneration depends on rotten logs.

Unsuitable microsites that cover an area larger than the diameter of one tree crown create "edaphic gaps" (Lertzman et al. 1996). Edaphic gaps are associated with an identifiable edaphic or topographic condition that makes seedling establishment very unlikely (Lertzman et al. 1996). The presence of such gaps can not be entirely explained by the death of the gapmakers.

The spatial mosaic of suitable and unsuitable microsites is often a stable ecosystem feature. Simply, trees grow where trees have been growing. This is the case in high-elevation ecosystems where regeneration does not rely primarily on rotten logs. The spatial distribution of rotten logs changes slowly over several tree generations.

Homogeneous Stand Structures in High-elevation Ecosystems

After centuries of using deforested high-elevation areas for grazing, thousands of hectares have been reforested in Switzerland in the last 100 years to reduce peak waterflows (Leibundgut 1982). Most of these areas were planted in regular grids, regardless of microsite variation. Some attempts at reforestation failed in the seedling establishment phase. The pattern of seedling growth observed in one high-elevation trial is very instructive: seedling growth and survival was highest on the lee side of ridges and lowest in hollows (Schoenenberger and Frey 1988). Other attempts in less rugged terrain were initially successful, but later in the sapling stage the stands became unstable (Schoenenberger et al. 1990). These unstable planted stands now pose major silvicultural problems, except in cases where a major storm or snow event has solved the problem by creating a more stable clumpy stand structure. In other cases, it is too late to do anything—the risk of complete stand destruction because of the short-term destabilizing effect of any stand intervention is deemed to outweigh the potential benefits. Some younger stands can still be thinned with a clump-creating intervention. This experience shows that simply leaving open strips between clumps during planting would have been much more successful and less costly. It also suggests that homogeneous stand structures should not be created in environments that promote heterogeneity.

Why Microsite Information Is Important

There are two main applications for knowledge about microsites. The first is in reforestation of cutblocks. In many high-elevation ecosystems, mounds are suitable microsites for seedling establishment (Farnden 1994). Beneficial effects of mound planting on seedling growth have been reported several times (Bassman 1989; Letchford and Hawkins 1994). Natural or artificial mounds can be used. Other possibilities include planting near stumps and logs (Farnden 1994). This has recently become common practice in Swiss high-elevation forests.

While Farnden (1994) already advocates relaxed spacing requirements for tree planting, I would go further and suggest that the spatial layout of planting should ensure the long-term stand clumpyness where the original stand was clumpy (Schoenenberger et al. 1990). This applies equally to artificially created mounds.

The second application of knowledge about microsites is in the design of cuts to promote natural regeneration. Cuts can create suitable microsites and improve growing conditions on existing suitable microsites. However, no general guidelines exist to explain how these cuts should look. Cuts generally have an influence on light conditions, precipitation patterns, snowmelt, and vegetation development. In wet-cold ecosystems, for instance, cuts should improve light conditions, thus enhancing soil temperature, without increasing snowpack considerably, and without promoting a rapid vegetation development. On steep north-facing slopes in the Swiss Alps where avalanches must be prevented, Norway spruce (*Picea abies* [L.] Karst.) is successfully regenerated with slit-shaped openings that are a third of a tree height wide and two tree heights long (Ott et al. 1991). The Swiss experience shows that in high-elevation forests a single-tree selection system is inappropriate because light conditions are not sufficiently improved (Ott 1995).

SUITABLE MICROSITES FOR NORWAY SPRUCE ESTABLISHMENT IN SWISS SUBALPINE FORESTS

In the European Alps, Norway spruce ranges over several climatic zones from oceanic to continental. Suitable microsites for seedling establishment are not identical throughout these zones. In the climate intermediate between oceanic and continental in some regions of the Swiss Alps, the following microsites are beneficial for Norway spruce establishment (Sorg 1980; Imbeck and Ott 1987; Ott et al. 1991; Brang 1996):

- raised microsites (rotten wood, stumps, rocks);
- microsites receiving sufficient direct sunlight in summer (at least one and a half to two hours daily potential duration of sunlight; diffuse light is not sufficient for seedling establishment);
- microsites without vegetation or at least without dense vegetation;
- microsites with early snowmelt;
- microsites in gaps; and,
- microsites with mineral soil, especially on south-facing slopes.

The following factors contribute to these microsite patterns (Frey 1983; Imbeck and Ott 1987; Brang 1996):

- soil temperature (dependent mainly on direct radiation);
- low light levels (normally coupled with low soil temperature, rainfall interception, and/or competing vegetation);

- drought, especially (a) during germination, and (b) under canopy due to interception;
- snow press, snow gliding;
- snow mould (*Herpotrichia* sp.), especially on microsites with late snowmelt; and
- competition by herbs and shrubs (by smothering and by lowering the soil temperature).

The case of Norway spruce in the Swiss Alps is not unique, but resembles patterns found in other high-elevation ecosystems. Low soil temperature is considered limiting for root development in several north American studies (Delucia and Smith 1987; Binder et al. 1989; Day et al. 1989; Coates et al. 1991). Snow is identified as an obstacle for seedling establishment (Koppenaal and Mitchell 1992). Many investigators stress the adverse effect of competing vegetation on tree seedlings (Eis 1981; Coates et al. 1991; Balisky and Burton 1995). Drought is detrimental to seedling establishment on duff (Eis 1965; Nobel and Alexander 1977; Knapp and Smith 1982).

Despite all these similarities between high-elevation ecosystems, differences also exist. The effect of the environmental factors that define the suitability of microsites varies over space, time, and developmental stages. These variations are discussed below.

Spatial Variation of Suitable Microsites

Four spatial scales are recognized: microsite, gap, aspect, and regional (Figure 2). Environmental factors create a mosaic of suitable and unsuitable microsites on a very small scale (decimetre to metre), around a stump, for example. Patterns of direct radiation, precipitation, snow accumulation and melt vary on the gap scale. A stump and its immediate surroundings may be a favourable microsite for tree seedlings on a south-facing stand edge, but may not receive enough direct light on a north-facing stand edge (Imbeck and Ott 1987; Brang 1996; for light patterns see Chen et al. 1995).

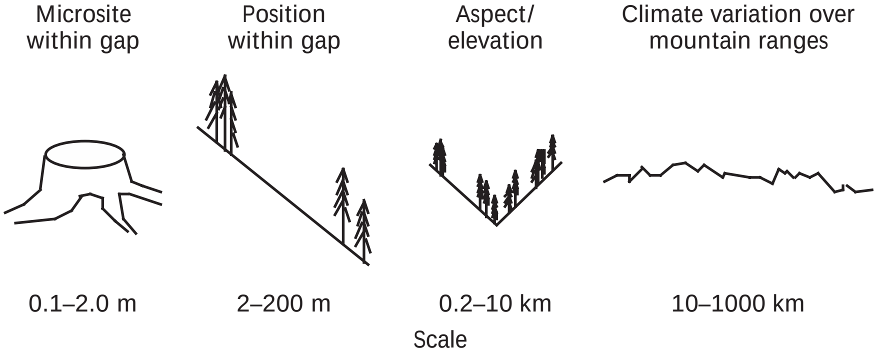

FIGURE 2 *Variation of suitable microsites for tree seedlings over spatial scales. Suitable microsites for a given species depend on micro-topographical features such as stumps, stand features such as stand edges, topographical features such as aspect, and climate variation across mountain ranges.*

The next scale refers to ecological differences between aspects or elevations. While organic layers are often suitable microsites for tree seedlings on north-facing slopes, they are prone to desiccation on south-facing slopes (Noble and Alexander 1977; Brang 1996). While a high amount of direct radiation (up to six hours potential sunshine per day in summer) is beneficial for Norway spruce seedlings in an intermediate climate on a north-facing slope, more than two hours daily is already detrimental on duff on a south-facing slope (Brang 1996). Favourable microsites for tree seedlings generally become less frequent with increasing elevation, reflecting the increasingly harsher environment (Ott 1995).

On an even larger regional scale, the factors defining suitable microsites change with changes in climate. For instance, Norway spruce seedlings avoid microsites under canopy in intermediate to continental climate because of insufficient water supply (with up to 1400 mm annual precipitation) (Ott et al. 1991). In contrast, the same species is almost confined to microsites under shelter of canopy trees in an oceanic climate with ample precipitation (over 2000 mm annually) and long-lasting snowpack in gaps (Ott et al. 1991).

Temporal Variation of Suitable Microsites

The environment of a tree seedling changes over time. An obvious temporal variation of this environment is the annual variation in meteorological patterns (e.g., occurrence of drought periods and summer frost, timing of snowmelt, and length of growing period). The outcome of seeding experiments are highly dependent on year-to-year variations in weather, as shown by Noble and Alexander (1977) in the Rocky Mountains. Another temporal microsite change occurs when competing vegetation quickly encroaches on suitable microsites created by a gap-forming event (Imbeck and Ott 1987; Brang 1996).

Weather episodes are more important than long-term meteorological averages. A sequence of ten summers with temperatures above average may offer a suitable microsite "in time" (cf., Payette and Filion 1985; Rochefort et al. 1994). The long-term existence of some high-elevation forests may even result from the infrequent occurrence of such microsites "in time." Weather episodes can also negatively affect regeneration. Norway spruce regeneration on whole slopes was killed in a single winter with long-lasting snowpack that offered favourable conditions for pathogenic black snow mould (Ott 1995). Other episodic meteorological events include drought periods, summer frost, and storms. Storms, as typical episodic weather elements, can create gaps containing suitable microsites, or they can thin out crowns of surrounding canopy trees, in both cases enhancing light conditions.

Since extreme stress events occur only rarely, we are not aware of all types of stress events, and only rarely are we able to observe the response of seedlings to extreme stress. Short-term regeneration success over 2–5 years should therefore only be extrapolated with caution over longer time periods. In ecosystems where the overall conditions are harsh and natural regeneration relies on a few suitable microsites, it is risky to disregard the pathways that have proven successful over centuries of natural establishment of seedlings. Suitable microsites should therefore be considered during harvesting and planting.

Variation of Suitable Microsites Over Stand Developmental Stages

Germinants, seedlings, and saplings vary in resistance to environmental stress. For example, the susceptibility to drought and smothering by vegetation or litter is highest for germinants (Brang 1996), the susceptibility to browsing is highest for seedlings, and the susceptibility to snow movements on slopes is highest for saplings (Schoenenberger 1978).

These differences in microsite suitability can reduce the number of suitable microsites over developmental stages. This applies to Norway spruce on north-facing slopes in intermediate climate in and around small gaps. While germinants are found in great densities all over the forest floor, later stages are confined to raised sites in gaps, with sufficient direct sunlight (Sorg 1980; Brang 1996). In contrast to the north aspect, on the south aspect the germinant stage in Norway spruce is most vulnerable, in this case by summer drought. The earliest stage therefore already defines the suitable microsite, while further stages are more resistant to environmental stress (Brang 1996).

CONCLUSIONS

The harsh microclimatic conditions above the treeline encroach into high-elevation forests, creating a mosaic of favourable and unfavourable microsites for tree seedling establishment. Seedlings successfully establish on favourable, and fail on unfavourable microsites. If we want to promote natural regeneration in high-elevation forests, we should try to take the viewpoint of a seedling and thus "think in microsites."

The mosaic of favourable and unfavourable microsites is reflected in the clumpy stand structure of many high-elevation forests. The gaps in the forest cover should not be mistaken as sites calling for improvement. These gaps partly reflect environmental conditions unsuitable for seedling establishment.

Direct radiation is much more important than diffuse for successful seedling establishment in Swiss high-elevation forests (Imbeck and Ott 1987, Brang 1996). The role of diffuse and direct radiation requires testing in British Columbia's high-elevation forests, to confirm the conclusions of the Swiss findings.

Since trees are close to their ecological limits in high-elevation forests, slight variations in key ecological factors are likely to be highly influential on seedling establishment. This creates silvicultural possibilities for promoting regeneration. On the other hand, inappropriate silvicultural practices can also negatively affect regeneration in the long term.

The importance of rare meteorological events for tree seedling establishment makes it difficult to explain or predict regeneration success or failure. Forest management practices that rely on artificially created patterns may seem successful for decades before a single extreme event can cause a major failure. This is why a close imitation of natural regeneration patterns that have been successful for centuries is advocated in high-elevation forests.

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Regeneration of Engelmann Spruce and Subalpine Fir from Seed in ESSFwc2 Forests

MICHAEL FELLER

ABSTRACT

A study investigating the influence of seedbed, light, ecosystem, and competing vegetation on germination, and initial (three years) germinant survival and growth of subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) was conducted in the northern Monashee wet cold Engelmann Spruce – Subalpine Fir biogeoclimatic variant (ESSFwc2) in the Clearwater Forest District. The study was conducted as part of the Sicamous Creek Silvicultural Systems Project of the Kamloops Forest Region. Small (approximately 1 m^2) plots in mesic and subhygric ecosystems were established in closed forest, open forest, and clearcuts. Within each ecosystem/light combination, three seedbed types were created—undisturbed forest floor, mineral soil, and burned forest floor. Half of the plots had all competing vegetation removed. Each ecosystem/light/seedbed/competition combination was replicated 12 times, for a total of 432 plots. Plots were seeded in the fall of 1991. Excessive seed predation required that the first year (1992) results be rejected and the study be started again. Seeding occurred again in the spring of 1993, with 100 seeds per species per plot, and germinant survival and growth was assessed for the first two growing seasons (1993 and 1994). Survival was also assessed for the third growing season (1995).

After three growing seasons, the number of living germinants of both fir and spruce were greater in subhygric than in mesic ecosystems and where forest floor disturbance increased (greatest numbers occurred on mineral soil seedbeds). The numbers of fir germinants increased as the amount of light increased (greatest numbers in clearcuts). The same trend occurred for spruce germinants during the first growing season, but higher mortality of spruce germinants in clearcuts during the second growing season resulted in slightly greater numbers of surviving germinants in open forest areas. This trend persisted through the third growing season. The presence of competing vegetation reduced the number of fir germinants, but had no significant effect on the number of surviving spruce germinants.

After two growing seasons, surviving fir germinants were larger (height, basal diameter, and crown volume) and were generally less influenced by seedbed, ecosystem, and competition, than spruce germinants. Light was the only factor studied which significantly influenced fir germinant

growth. This growth was greater in clearcut than in forested areas, but only in mesic ecosystems. Spruce germinant growth tended to be:

- greatest for undisturbed forest floor seedbeds and least for mineral soil seedbeds, this tendency increasing as the amount of light increased;
- greater in mesic than in subhygric ecosystems, but only in clearcut areas; and
- greater in clearcut than in forested areas.

The presence of competing vegetation significantly affected (reduced) only the basal diameters of spruce germinants.

Partially cut ESSFwc2 forests can be naturally regenerated to both spruce and fir as long as sufficient seed falls and escapes predation. The optimum treatment to naturally regenerate ESSFwc2 forests with fir, would involve clearcutting and exposing mineral soil. The situation for spruce is more complicated. The optimum treatment to encourage germination would involve providing partial shade and exposing mineral soil. However, to promote germinant growth, clearcutting with minimal forest floor disturbance is optimal.

INTRODUCTION

Engelmann Spruce – Subalpine Fir (ESSF) zone forests are gaining importance for supplying timber as less timber is available in lower-elevation forests. In addition to their increasing value for timber, these forests are valued for recreational and nature conservation purposes. These competing demands require more diversified forest management. Such management is likely to have objectives which put greater emphasis on natural regeneration and the use of partial cuts or small clearcuts, instead of large clearcuts.

Upland conifer forests dominate the ESSF zone. Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) are the major climatic climax species, with lodgepole pine (*Pinus contorta*) as a seral species, which is more important in drier areas (Lloyd et al. 1990; Meidinger and Pojar 1991). These forests were traditionally harvested, then regenerated by clearcutting methods. However, future management is likely to emphasize alternatives to clearcutting.

In a review of regeneration of American forests in 1972, Franklin and DeBell (1973) concluded that "available literature indicates that for most forest types and species on most sites, foresters have wide latitude in choosing regeneration cutting methods insofar as ecological requirements of tree species are concerned . . . there is no ecological necessity for large patch or continuous clearcuttings to regenerate most types, species, and sites." The use of alternatives to clearcutting has proceeded in the United States in recent years, particularly in the Pacific Northwest. Hopwood (1991) considered it possible to extrapolate American forest regeneration results and practices to the coastal and southern interior portions of the province because of the ecological similarities between British Columbia and the United States Pacific Northwest. However, the central interior has fewer ecological similarities with this area and it is uncertain the degree to which forest regeneration information from the United States can be extrapolated to British Columbia.

Because our present understanding of the optimum conditions for germination and initial growth of Engelmann spruce and subalpine fir in British Columbia is incomplete, a study to determine these conditions in an important ESSF unit commenced in 1991. The unit—the northern Monashee wet cold (ESSFwc2) variant—is the most extensive ESSF unit in the Kamloops Forest Region (Lloyd et al. 1990). It is probably a distinctively British Columbian unit because American Engelmann spruce – subalpine fir forests are likely to have warmer growing seasons, while those in Alberta are drier. Full details and results of the study are available in Feller (1997). Here I provide a summary of the methods and some of the results.

STUDY AREA

An area of forests and adjacent clearcuts in the vicinity of kilometre 30 on the Otter Creek road in the Clearwater Forest District was selected for this study. This area was chosen because:

- I was already using it to study some ecological effects of slashburning, thus simplifying logistics;
- it contained representative examples of the two most common ecosystems in the ESSFwc2 forests; and
- it was one of the very few areas in the ESSFwc2 containing clearcuts with untreated slash.

When the study began in 1991, the Sicamous Creek Silvicultural Systems project was still at a conceptual stage and appropriate study sites at Sicamous Creek did not exist.

The area is at an elevation of 1400–1600 m on a ridge between the Adams and North Thompson rivers. It receives 1200–1440 mm of precipitation annually, 60% of which was snow during 1988–1993. The two most common ecosystems consist of the subalpine fir – azalea – oak fern plant association on mesic sites, and the subalpine fir – valerian – oak fern plant association on subhygric sites (Lloyd et al. 1990).

METHODS

Small plots, each approximately 1 m^2 in area, were established in mesic and subhygric ecosystems, both in forests and clearcuts. Within each ecosystem, plots were located in three light conditions—full sunlight (clearcut), light shade (open areas within open forest), and heavy shade (areas within denser forest). Within each ecosystem/light combination, three seedbed types were created: undisturbed forest floor, mineral soil (created by removing overlying forest floor), and burned forest floor. The latter seedbed was created by piling woody debris on plots, covering with a plastic sheet, then burning in late summer (August 28–29, 1991).

The forest floor depths of burn were monitored using three depth-of-burn pins per plot on a representative number of plots. Average depths of burn were 0.6–1.7 cm (Table 1), which are typical of low-severity slash-burns in the area (Feller and Hamilton 1994).

Twenty-four plots were established for each ecosystem/light/seedbed combination. Twelve of these plots had no further treatment, while the other twelve had all competing vegetation regularly removed during the subsequent growing seasons.

TABLE 1 *Forest floor depths of burn for the small plots with burned forest floors*

	Depth of burn (cm)				
Plot type	**Mean**	**SD**[a]	**Maximum**	**Minimum**	***N***[b]
Mesic ecosystems					
Full sunlight	1.4	1.1	4.5	0.1	36
Light shade	0.7	0.5	2.4	0.1	71
Heavy shade	0.6	0.7	3.4	0.1	40
Subhygric ecosystems					
Full sunlight	1.7	1.6	7.4	0.1	57
Light shade	1.6	1.0	3.9	0.1	44
Heavy shade	1.2	0.9	4.2	0.0	56

[a] SD = standard deviation of the mean.
[b] *N* = number of measurements.

All plots were numbered and were surrounded by wood and wire cages to exclude bird and mammal seed predators. Each plot was divided into halves. Eighty subalpine fir seeds (Seedlot No. 02503) were sown on one-half and 100 Engelmann spruce seeds (Seedlot No. 25602) were sown on the other half (during September 28–October 17, 1991), after all the burn plots had been thoroughly wetted with rain. Seeds were obtained from the B.C. Ministry of Forests Seed Centre. After sowing, each cage was covered with fibreglass screening to exclude any additional seeds from entering. This screen was removed in the spring (May 30) of 1992. There was no sign that tree seeds had fallen onto the plots during the August 30–October 17 period.

The experimental design thus involved plots randomly located between obstacles, such as trees or decaying logs. These plots were used to assess the effects of ecosystem, light condition, seedbed type, and plant competition, on the germination and initial growth and survival of Engelmann spruce and subalpine fir germinants. The total number of plots established was:

$$2 \text{ (ecosystem)} \times 3 \text{ (light)} \times 3 \text{ (seedbed)} \\ \times 2 \text{ (competition)} \times 12 \text{ (replicates)} = 432.$$

When the fibreglass screen was removed in 1992, approximately 20% of the wood and wire cages had been damaged by the winter snowpack, allowing access to small mammals. In addition, a number of the cages had lost their seal to the soil because of erosion by water. Seed predators had eaten many seeds, particularly those of subalpine fir. As the extent of seed predation could not be quantified, the 1992 seed germination results would be inconclusive. I decided to continue monitoring seed germination during 1992, after repairing and resealing damaged screens, and to resow the plots with seeds for the 1993 growing season. The results for the 1992 growing season would be meaningful only if confirmed by results for the 1993 growing season. Such confirmation did not occur so the 1992 results were rejected.

At the end of the 1992 growing season (October) all germinants were removed from the plots. All fresh litter was removed from the mineral soil and burned forest floor plots. The latter was partly accomplished by burning the plots using a hand-held drip torch. All predator-exclusion screens were thoroughly sealed and the fibreglass seed-exclusion screens were refitted to the plots.

On June 2–3 1993, shortly after snow had left the plots, they were reseeded with (100 seeds per species per plot) Engelmann spruce (Seedlot No. 26212) and subalpine fir seeds (Seedlot No. 2503) obtained from the B.C. Ministry of Forests Seed Centre. Although the seeds were stratified by the Seed Centre, the fir seeds were not completely stratified because of problems at the Centre.

At the time of sowing, sunflower seeds were scattered beside the plots in an attempt to minimize seed predation within the plots. A total of 45 kg of sunflower seeds was used. Sunflower seeds were also scattered beside the plots during monitoring visits in June and July (33 kg of seeds in each of June and July).

Monitoring consisted of counting the numbers of living and dead germinants every two to four weeks during the growing season (from June 1 through August 26 1992; June 18 through October 12 1993; and June–October 1994, as well as in June–September 1995). Predator exclusion screens were removed from all plots in early August 1993. In the clearcut areas, the plots which were to experience plant competition were selected using random number tables. All other plots were weeded by hand at each monitoring period, beginning on June 18. In the forest areas, the plots which were to experience competition were allocated to two light regimes—light shade and heavy shade. The degree of shading of each plot was determined by measuring the amount of photosynthetically active radiation (PAR), expressed as a percentage of that incident on the canopy, received by each plot. This was done using a Decagon Devices, Model SF-40 Sunfleck Ceptometer during June 1993 and confirmed, by repeating the measurements in July 1993. The plots were separated into light shade (open forest) and heavy shade (closed forest) categories, based on the PAR received by the plots. Within each category, the plots to experience plant competition were again selected using random number tables. All other plots were weeded by hand. In 1992, suitable weather (continuous cloud cover is desired) did not occur and the ceptometer was not available. Therefore, measurements were completed for the mesic forest, but not for the subhygric forest plots. Consequently, mesic forest plots assigned by random number tables as free of plant competition were weeded, beginning on June 17. However, no subhygric forest plots were assigned as free of plant competition, and all such plots were left unweeded throughout the growing season. Successful use of the ceptometer in 1993 resulted in weeding in both ecosystems during 1993 and 1994.

The PAR used to separate the light shade and heavy shade categories varied between ecosystems, but to a lesser extent between seedbed types within an ecosystem (Table 2). The higher percentages of incident radiation for the mesic ecosystem indicate that the mesic forest plots were located in more open conditions than the subhygric forest plots.

TABLE 2 *Photosynthetically active radiation, expressed as a percentage of that incident on the forest canopy, used to delineate light shade (open forest) from heavy shade (closed forest) conditions*

Seedbed type	**Ecosystem** **Mesic**	 **Subhygric**
Forest floor	24.4	13.0
Burned forest floor	27.5	17.5
Mineral soil	26.0	17.0

Soil temperatures in the germinant rooting zone (considered to be 2 cm below the soil surface) were measured on August 30 and September 11, 1995, using Fisher Scientific NIST digital thermometers. Three to six measurements were made in each plot, and an average calculated for each plot. Measurements were made from 10 am to 5 pm and corrected to 12 noon from curves relating temperatures to time of day. These curves were obtained from hourly measurements of 2 cm deep soil temperatures in areas considered characteristic of each of the three light conditions in each of the two ecosystems. Thus, six temperature/time curves were generated for each day of sampling and each measured average plot temperature was corrected to a 12 noon temperature using the appropriate curve for the light condition/ecosystem combination of each plot. The August day was cool and cloudy, whereas the September day was warm and sunny. The August and September temperatures were averaged and statistical analyses conducted on these average plot temperatures. Using some of my unpublished data of continuous soil temperature measurements throughout the growing season in the study area, I considered these average values to reasonably approximate growing season temperatures.

Analysis of variance was undertaken on the germination data from each collection period. In some instances, visual inspection suggested that data sets were not normally distributed or Bartlett's tests indicated variances were not homogeneous, lowering confidence in the ANOVA results. Data were subjected to arcsine transformations and the transformed data were again subjected to ANOVA. Transformations improved normality and homogeneity of variances, although variances were occasionally still not homogeneous. Germinant growth and plot temperature data were checked for normality and homogeneity of variances before analysis of variance. Logarithmic or square root transformations were needed to improve normality and homogeneity of variances, but some data sets could not meet the assumptions of ANOVA. All analyses were conducted using SYSTAT software (Wilkinson 1986).

RESULTS

Seed Viability

Seed viability was tested using a germination chamber in a laboratory. The 1991 and 1993 seeds of both species had similar viabilities. Spruce seed had greater viability than fir seed (Table 3). The low viability of fir seed is well known. Burns and Honkala (1990) reported that the average germinative capacity of fir seed is 34% and that it is often less than 30%. However, an incomplete stratification of the 1993 fir seeds at the B.C. Ministry of Forests Seed Centre might also have contributed to the lower results for 1993.

TABLE 3 *Viability (% germination) of the stratified seeds used in the study*

	% Germination	
Year	**Engelmann spruce**	**Subalpine fir**
1991	71	20
1993	77	17

Seed Germination

Ecosystem After three growing seasons, more living germinants were found in subhygric than in mesic plots, although the difference was statistically significant only for spruce (Table 4). Survival was probably influenced by soil moisture, but no evidence relates drought as a major cause of germinant mortality (Feller 1997). There was a significant second-order interaction for spruce, in which the number of surviving germinants was greater in subhygric than in mesic ecosystems in forest floor and mineral soil seedbed plots, but not in burned forest floor plots.

Light Spruce and fir exhibited similar trends, with numbers of germinants increasing with photosynthetically active radiation (i.e., germinant numbers were generally greatest in the clearcuts and least in the closed forests) (Table 4). Heavy mortality of spruce during the summer of 1993 reduced the initially higher numbers of germinants in clearcuts so that no statistically significant differences existed between clearcut and open forest during most of the study. Although significant second-order

TABLE 4 *Average number of living germinants per plot at the end of three growing seasons, and results of analyses of variance*

Environmental parameter	Spruce	Fir
Ecosystem (E)		
Mesic	6.2 (0.7)B[a]	1.9 (0.2)
Subhygric	8.1 (0.7)A[b]	2.3 (0.3)
Significant second-order interactions	S × E	none
Seedbed (S)		
Mineral soil	10.9 (0.9)A	4.2 (0.5)a
Burned forest floor	8.0 (0.9)B	1.3 (0.2)b
Undisturbed forest floor	2.6 (0.4)C	0.9 (0.1)b
Significant second-order interactions	S × E, S × L	S × L
Light (L)		
Closed forest	4.6 (0.6)B	0.9 (0.2)b
Open forest	8.8 (0.9)A	1.8 (0.2)b
Clearcut	8.2 (0.9)A	3.6 (0.5)a
Significant second-order interactions	S × L	S × L
Competition (C)		
Without	7.5 (0.6)	2.5 (0.3)A
With	6.9 (0.7)	1.7 (0.2)B
Significant second-order interactions	none	none

[a] Standard errors are given in parentheses.

[b] For a given environmental parameter and a given species, different letters indicate significantly different mean values ($p < 0.05$). Uppercase letters are for data sets which conform to the assumptions of ANOVA, lowercase letters are for data sets which do not conform to these assumptions. An absence of letters indicates no significant differences.

interactions are sometimes complications, they do not alter the conclusion that heavily shaded areas are the least desirable for germination of both species in the study area. This suggests that shading adversely affected fir germination and initial survival and that fir germinants were relatively tolerant of the exposed conditions of the clearcuts. In the case of spruce, however, substantially greater mortality occurred in clearcut than in forested plots in both of the first two summers, but not in the third summer (Feller 1997). Thus, the exposed conditions of the clearcuts favoured germination, but were less beneficial to the initial survival of germinants. This was attributed to solarization, a condition that occurs when photosynthesis is inhibited because of high light levels, which ultimately leads to death. After growing for two years, the germinants became adapted to high-light levels.

Seedbed Undisturbed forest floor had the lowest number of living germinants of both species, while mineral soil had the most (Table 4). Again, however, significant second-order interactions between seedbed and light, ecosystem, and competition complicate interpretation of the results. In the case of spruce in mesic ecosystems, germinant numbers on burned seedbeds were slightly greater than those on mineral soil seedbeds, and both were substantially greater than those on undisturbed forest floor seedbeds. Considering the interaction with light, spruce germinant numbers in open forests and clearcuts were similar to the overall trend. In closed forests, however, germinant numbers were greater on burned seedbeds than on the other two seedbed types. In the case of fir, light was the only factor which interacted with seedbed type. In clearcuts and open forests, germinant numbers were similar to the overall trend. In closed forests, however, germinant numbers were greater on burned seedbeds than on the other two seedbed types, as with spruce. Thus, with low-light levels, burned seedbeds appeared to support the greatest numbers of germinants of both species.

Competition The presence of competing vegetation tended to decrease the number of living germinants of both species (Table 4) although, in the case of spruce, this decrease was not significant at the end of the third growing season. Because of the relatively small differences in germinant numbers of each species between plots with and without competition, it is reasonable to conclude that competition had less effect on the number of living germinants than did seedbed or light.

Germinant Growth

After two years, fir germinants had generally grown more than spruce germinants for each measure of growth used—height, basal diameter, and crown volume (Figures 1–4). Fir growth was less influenced by the different environmental parameters studied than was spruce growth. Fir germinant growth was only significantly affected by light, whereas spruce germinant growth, for at least one of the growth measures used, was significantly affected by seedbed, ecosystem, light, and competition. These results are discussed in greater detail in Feller (1997); a summary follows.

Seedbed Spruce height and crown volume were greatest for undisturbed forest floors and least for mineral soil seedbeds (Figure 1). However, significant interactions between light and seedbed occurred. These interactions suggest that this tendency increased as the amount of light increased. Spruce basal diameters were not significantly affected by seedbed and neither were any of the growth measures for fir germinants.

Heineman (1991) also found better growth of interior spruce seedlings on forest floor than on mineral soil materials in the Sub-boreal Spruce (sbs) zone. She attributed this to higher temperatures in the forest floor materials. Data from the study plots suggest that soil temperatures can not explain all of the growth results.

Ecosystem Spruce tended to grow better in the mesic ecosystem, although this was statistically significant only in the case of crown volume (Figure 2). Fir growth was unaffected by ecosystem (Figure 2). For spruce crown volume, ecosystem interacted with light in determining growth. Thus, crown volumes in mesic ecosystems were greater than those in subhygric ecosystems for clearcut plots, but were not significantly different from those in subhygric ecosystems for open or closed forest plots. Why ecosystem should affect growth only in clearcut areas is not immediately obvious, although it may have been related to seedbed temperatures (Feller 1997).

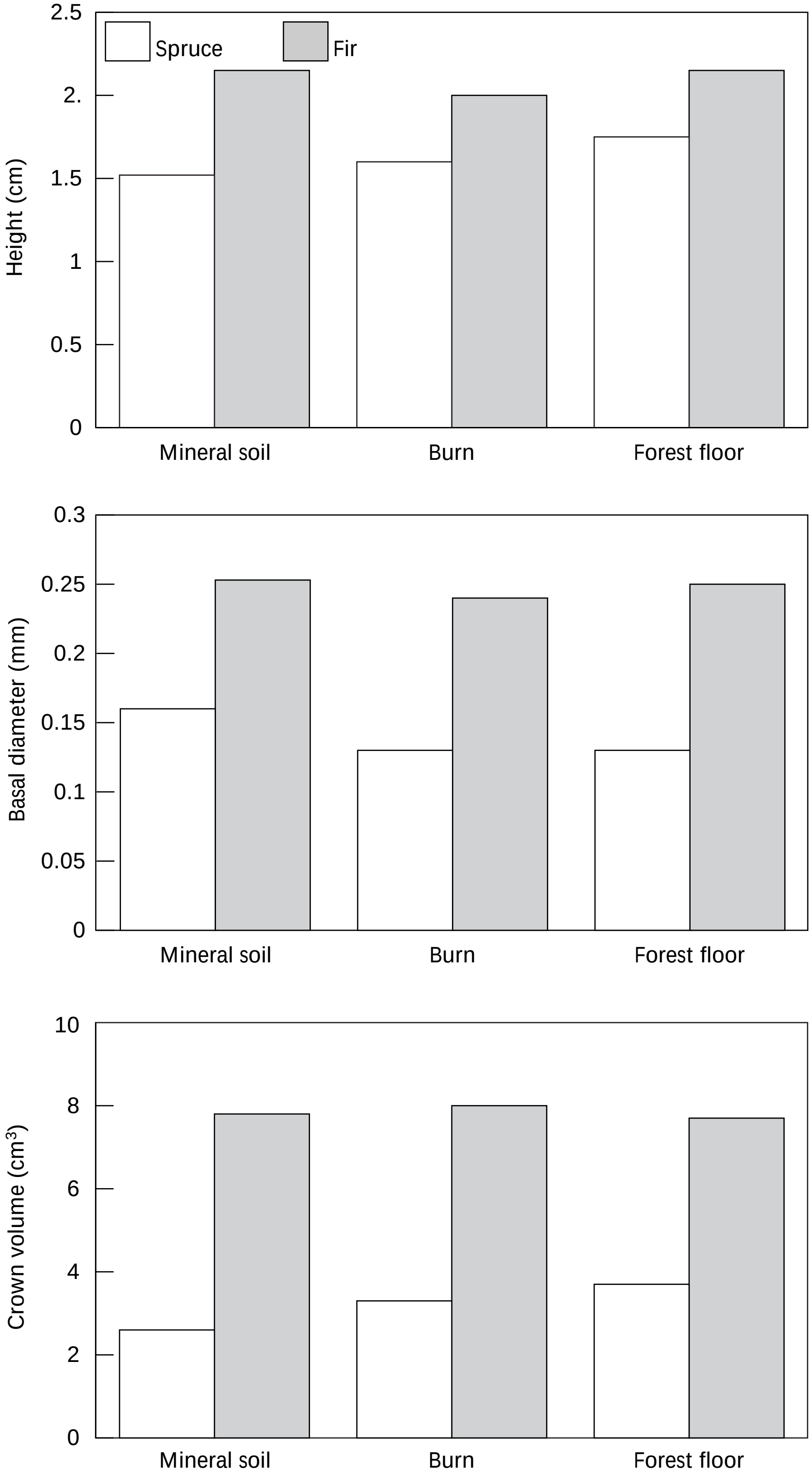

FIGURE 1 *Mean growth of living spruce and fir germinants at the end of the second growing season, as a function of seedbed.*

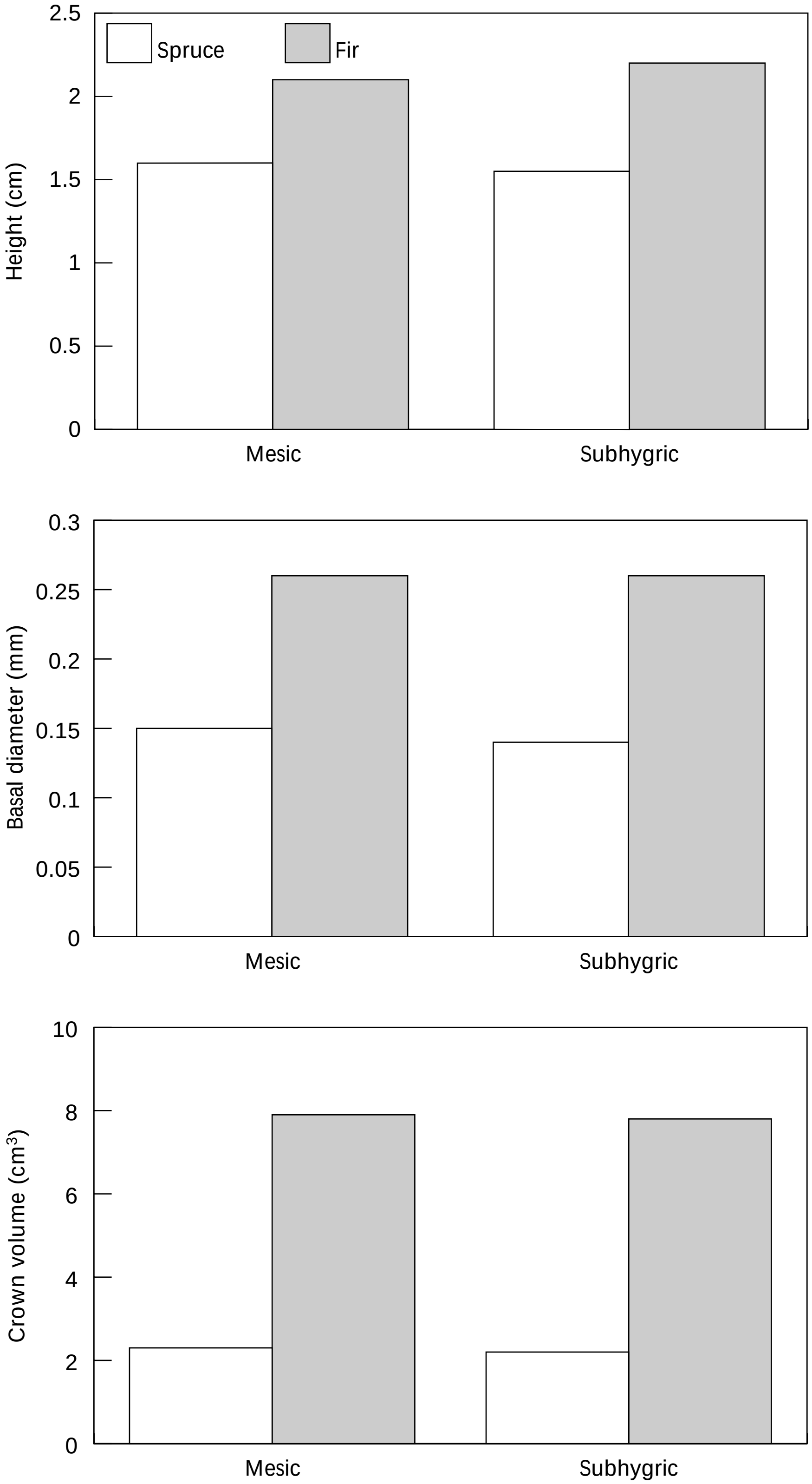

FIGURE 2 *Mean growth of living spruce and fir germinants at the end of the second growing season, as a function of ecosystem.*

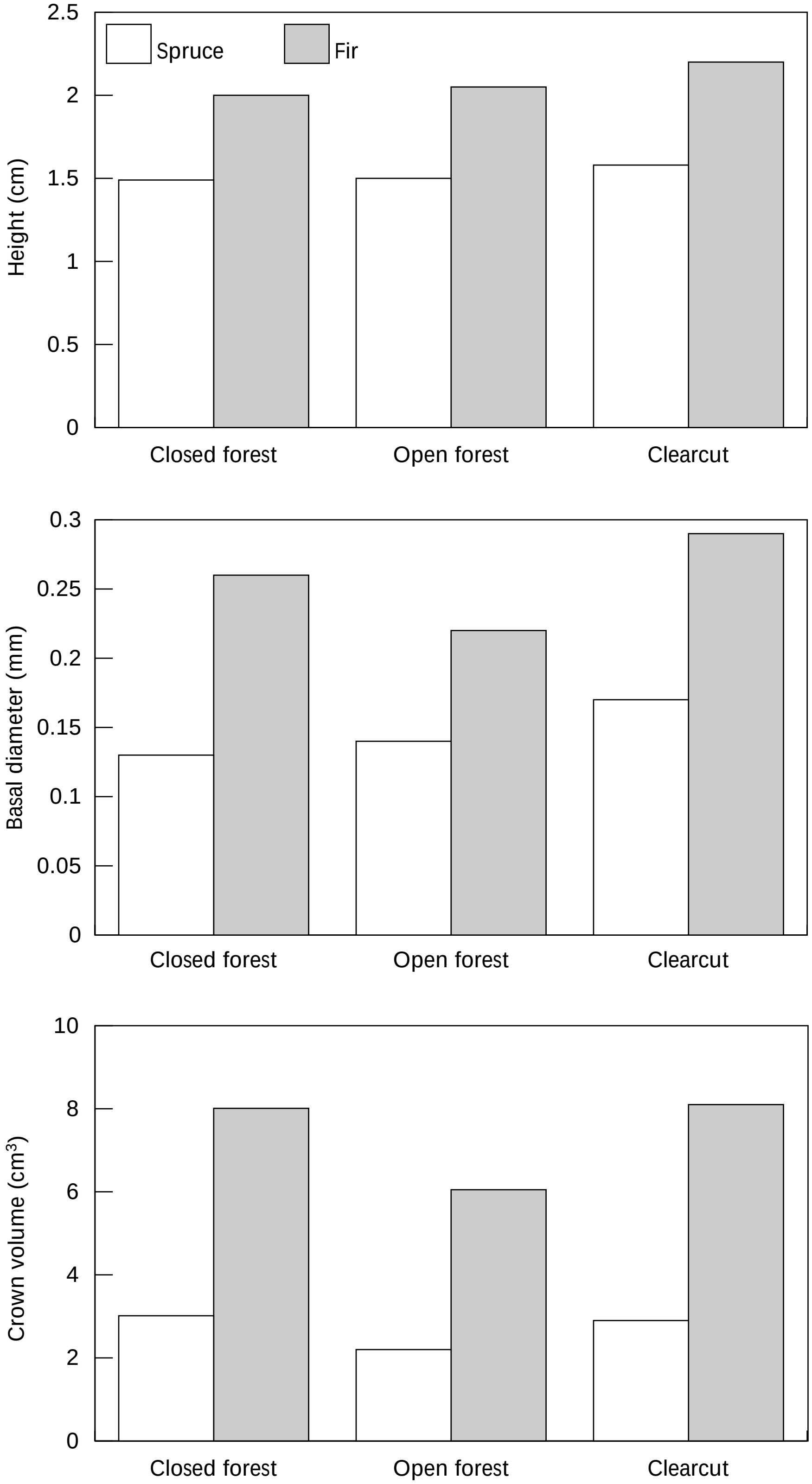

FIGURE 3 *Mean growth of living spruce and fir germinants at the end of the second growing season, as a function of light.*

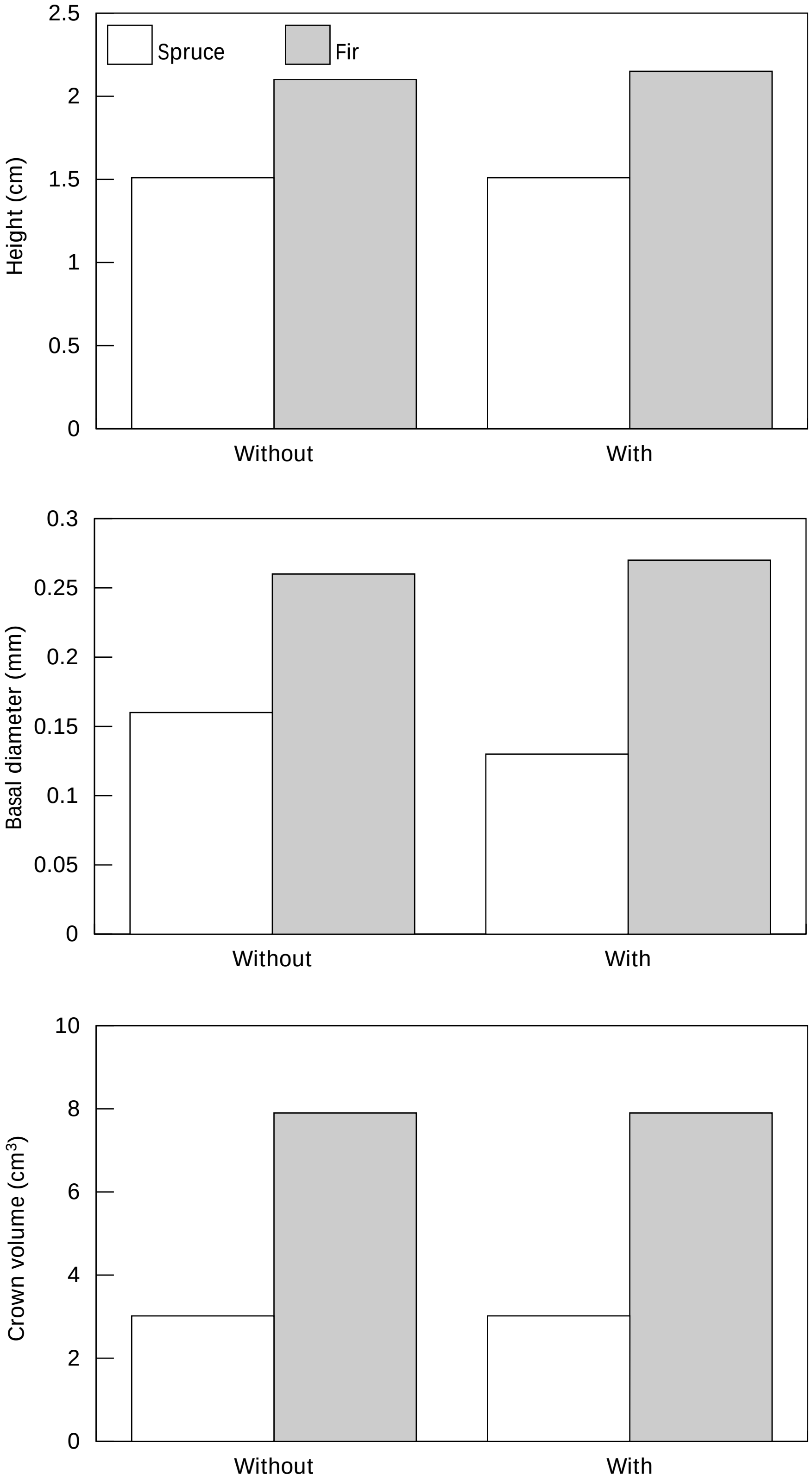

FIGURE 4 *Mean growth of living spruce and fir germinants at the end of the second growing season, as a function of competition.*

Light Of the environmental parameters studied, light had the greatest effect on germinant growth (Figure 3). All three fir growth measures, and two of the three spruce growth measures (height and crown volume) exhibited statistically significant effects of light. In all instances, growth was best in the clearcut plots, with no significant differences between the open and closed forest plots (Figure 3). This is consistent with the results of other studies (e.g., Caza 1991; Comeau et al. 1993) that found better Engelmann spruce growth at higher light levels. In all five situations (species/growth measures) where statistically significant differences occurred, light was also found to significantly interact with ecosystem. It also significantly interacted with seedbed in influencing spruce height and crown volume, and with competition in influencing fir basal diameter.

These interactions suggest that both spruce and fir seedlings grew better in clearcut plots, although light did not significantly affect fir growth in subhygric ecosystems. Light also did not significantly affect spruce crown volumes on burned forest floor seedbeds or fir basal diameters in the presence of competing vegetation. The explanations for these exceptions are not clear.

Competition Competition had relatively little effect on seedling growth—fir being unaffected and spruce exhibiting only greater basal diameters in the absence of competition (Figure 4). This latter result only occurred for undisturbed forest floor seedbeds, where competition was generally greatest. For mineral soil and burned forest floor seedbeds, competition had no significant effect on basal diameters.

CONCLUSIONS

After three growing seasons the number of living germinants of both spruce and fir was greater in subhygric than in mesic ecosystems. Numbers of living germinants generally increased in the first growing season as the relative amount of photosynthetically active radiation increased. Shading adversely affected fir germination and initial survival, and fir germinants were relatively tolerant of the exposed conditions of the clearcuts. These conditions also favoured spruce germination, but were less beneficial to germinant survival. Partial shading, as in the open forest areas, enhanced germinant survival.

Disturbance to the forest floor (scraping away to mineral soil or burning) increased the numbers of living germinants, with the more severe disturbance (scraping away to mineral soil) producing the greatest numbers. The presence of competing vegetation had no major effect on the number of living spruce germinants, or on living fir germinants in forests, but decreased this latter number in clearcuts.

After two growing seasons, surviving fir germinants had grown larger, and were generally less influenced by seedbed, ecosystem, and competition, than spruce germinants.

Spruce germinant height and crown volume were greatest for undisturbed forest floor seedbeds and least for the most disturbed (mineral soil) seedbeds. This tendency increased as the amount of light increased. Fir germinant growth was unaffected by seedbed. Spruce germinant growth was greater in mesic ecosystems than in subhygric ones, but only in clearcut areas. Ecosystem had no effect on spruce germinant growth in forests or on fir germinant growth in forests or clearcuts.

Growth of both spruce and fir germinants was greater in clearcut than in forested areas, although fir germinant growth was unaffected by light in subhygric ecosystems. The presence of competing vegetation had relatively little effect on seedling growth. The only significant effect of competition was in reducing spruce basal diameters in undisturbed forest floor seedbeds.

If ESSFwc2 forests are to be naturally regenerated, clearcutting with exposure of mineral soil is the optimum treatment for promoting fir germination and initial growth, as long as a seed source is present and sufficient seed survives predation. In the case of spruce, partial shade enhances germinant survival, but decreases growth compared to open clearcut conditions. Mineral soil exposure enhances spruce initial survival, but reduces growth when compared to undisturbed forest floor. Thus, the optimum treatment to promote spruce germinant survival is different from that which promotes spruce germinant growth.

Partially cut ESSFwc2 forests can be naturally regenerated to both spruce and fir as long as sufficient seed falls and escapes predation. Germinant numbers, but not growth, are enhanced by disturbing the forest floor, particularly by exposing mineral soil.

ACKNOWLEDGEMENTS

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Pathology of Conifer Seed and Seedlings on Natural and Disturbed Forest Floors in the Engelmann Spruce – Subalpine Fir Zone

JIAN WEN ZHONG AND BART VAN DER KAMP

ABSTRACT

This study was designed to determine whether overwinter survival of conifer seed on natural and disturbed forest floors in the Engelmann Spruce – Subalpine Fir (ESSF) zone is significantly reduced by fungal pathogens of seed. Engelmann spruce and subalpine fir seed was placed on a number of seedbeds in nylon mesh bags just before snowfall in 1995 and recovered at snowmelt in 1996. Subsequent germination tests and culture of surface sterilized seed on a variety of culture media was used to measure seed viability and to identify putative pathogens. Controls consisted of seed stored moist and at near 0°C in the laboratory.

INTRODUCTION

The renewed interest in silvicultural systems other than clearcutting that has arisen in the last few years has led to a need for better understanding of the processes involved in natural regeneration. Current studies of natural regeneration usually consider seed production patterns, the physical characteristics of seedbeds, microclimate, competition, and seed predation by animals as the major determinants of success. Differences between tree species are interpreted as adaptation of these species to different seedbed and microclimatic conditions (niches).

Diseases of seed and seedlings in natural and disturbed forest floor seedbeds have received little attention. Almost all studies on conifer seed and seedling pathology have been conducted in the context of bareroot and container nurseries. However, diseases may play a major role in natural regeneration. For instance, it is well known that when cones come into prolonged contact with forest floors, they are likely to become contaminated with one or more pathogenic fungi, particularly the seed fungus *Caloscypha fulgens* (Pers.) Boudier (Sutherland et al. 1987). These pathogens are then transferred to the seed before or during the extraction process and may enter the nursery as seed-borne pathogens. Ascocarps of *C. fulgens* are common in Engelmann Spruce – Subalpine Fir (ESSF) zone forest floors immediately following snowmelt, and disappear within about two weeks. Seed that overwinters and germinates on such forest floors is also likely to become infected.

A further reason for the lack of attention to these diseases is their ephemeral nature. Seed that falls on the forest floor and dies because of invasion by pathogens is never noticed, and diseased seedlings may escape attention unless a special effort is made to detect such losses.

Natural regeneration of spruce (*Picea engelmannii* Parry) and subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) in mature ESSF stands is largely restricted to rotting wood, and does not usually survive following clearcutting because of drying or disturbance of the substrate. Following partial cutting, much of this advanced regeneration is also lost. In addition, because of its patchy spatial distribution, advanced regeneration can not provide satisfactory stocking. Natural regeneration in clearcuts and smaller strip or patch cuts is usually restricted to exposed mineral soil. The problem, then, is very slow natural regeneration in the ESSF leading to long regeneration periods, increased brush competition on certain sites, increased concern about landscape values, and reduced productivity because of long rotation periods.

Perhaps the most detailed study of spruce regeneration in the high-elevation forests of western North America is that of Alexander (1987) in Colorado. This study documents a slow recruitment of established seedlings resulting from severe losses of seed and of seedlings during the first few years following germination. Although the phenomena of slow regeneration is well known, the causes are not clear. Johnson and Fryer (1966) examined survival of Engelmann spruce in Alberta and found considerable predation of seed by rodents and insects. Seed that escaped predation lost viability within two seasons. Japanese scientists investigating the high-elevation forests of Hokkaido have shown that *Racodium therryanum* Thuem., which causes dark snow-blight, is also a common inhabitant of forest floors and responsible for substantial losses of overwintering conifer seed. Typically these losses ranged from 80 to 100% (Cheng and Igarashi 1988; Cheng 1989). In North America, the fungus *Caloscypha fulgens,* and perhaps others, may play a similar role. However, the pathology of naturally shed seed in North America has not received detailed study.

RESEARCH METHODOLOGY AND RESULTS TO DATE

The current study seeks to identify pathogens of seed and germinants that affect naturally shed seed which overwinters under snow on forest floors, and to assess the effect of such pathogens on natural regeneration. This will be achieved by placing seed of Engelmann spruce and subalpine fir in nylon mesh bags on various natural and disturbed forest floors in the fall, recovering the seed in the spring, and determining seed viability and the identity of pathogens on dead seed (controls: seed stored wet and cold in lab). Overwintered seed will be sown in the seedbeds on which they were placed in the fall and the causes of loss of germinants (if any) determined. We anticipated that the work will lead to a fuller understanding of the causes of failure of natural regeneration on undisturbed forest floors, and that in turn will lead to better practice.

In the summer of 1995, a survey was conducted to determine the distribution of Engelmann spruce and subalpine fir seedlings (i.e., trees < 0.5 m high) on various seedbeds. In undisturbed stands at the Sicamous Creek Silvicultural Systems study site, Englemann spruce seedlings occurred only on rotten logs covered with *Dicranum* moss. On such logs, spruce seedling density was 0.59 seedlings per square metre. These seedbeds covered 14.9% of the total area. Subalpine fir was also very common on rotten logs (1.28 seedlings per square metre), but occurred on normal forest floors at 1200 seedlings per hectare.

Caloscypha fulgens ascocarps were common on forest floors. They appeared within two weeks of snowmelt and disappeared a few weeks later. Thus one of the main suspected pathogens of spruce and subalpine fir seed was present and widely distributed in the experimental area.

During the summer of 1995, a series of site treatments were installed. Engelmann spruce and subalpine fir seed in fine nylon mesh bags (100 seeds per bag) was placed on these treated areas in the fall just before the first snowfall. Site treatments were:

- in undisturbed stands: (1) removal of the forest floor to expose mineral soil; (2) forest floor mixed with the 10 cm of the mineral soil; (3) undisturbed forest floor.
- in selectively cut areas: (4) mineral soil exposed by mechanical scarification; (5) undisturbed forest floor.
- in 1 ha-clearcuts: (6) mineral soil exposed by mechanical scarification; and (7) undisturbed forest floor.

Six replicates of each treatment were installed. Controls consist of seed stored moist and near 0°c in the laboratory.

Bags were recovered immediately after snowmelt in 1996 to determine seed survival. Thirty seeds from each bag were sown on the sites on which they were overwintered, and the remaining seed taken to the laboratory at the University of British Columbia to determine germination and to isolate, on a series of special media, possible pathogenic fungi.

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Advanced Regeneration in Engelmann Spruce – Subalpine Fir Forests

PASI PUTTONEN, JAANA KAIPAINEN, AND ALAN VYSE

ABSTRACT

Advanced regeneration, as a potential source to renew a site after the final harvest, may have several advantages over clearcutting in high-elevation forests. This study examines what factors lead to a successful post-release growth of Engelmann spruce and subalpine fir advanced regeneration in an Engelmann Spruce – Subalpine Fir biogeoclimatic zone (ESSF) ecosystem. In the field season of 1995, 288 permanent 10-m^2 inventory plots were established at the Sicamous Creek research site. At every plot, a number of stand and tree characteristics were recorded. Preliminary results are available on the density of advanced regeneration pre- and post-harvest, number of germinants, height and height distribution, growth rate, stem form, tree vigour, logging damage, and stocking status at the site. The density of advanced regeneration ranged from understocked 10-ha openings with 100 stems per hectare to well-stocked control plots with 2700 stems per hectare. The number of germinants varied from 800 to 3770 stems per hectare; in control plots the density of germinants was five times greater compared to the 0.1-ha and 1-ha openings. Most of the germinants (90%) became established on old logs and stumps. About 65% of seedlings were lost during the logging. When estimating future stocking status, 50% of the advanced regeneration was deemed inadequate as crop trees, mostly because of poor vigour or stem deformities.

INTRODUCTION

Advanced regeneration, or advanced growth, is the regeneration established in the understorey of a stand before final logging (Herring and McMinn 1980). It forms a potential source of material to renew the site after final harvesting. For example, renewal cost and lag are reduced because of a head start, brush problems are lessened, soil protection is possibly better achieved, and aesthetic quality and wildlife habitat may be maintained (Smith 1986). However, our understanding of the processes that lead to a successful post-release growth of advanced regeneration trees is poor for many of our forest ecosystems.

Some studies indicate that growth from advanced regeneration will be merchantable decades before natural regeneration (Herring and McMinn 1980). However, suggested use of advanced regeneration can have several constraints. First, an adequate *quantity* of advance growth of one or two species ecologically suited to the site must exist to meet stocking requirements. Second, the advanced growth must be of good *quality* to ensure vigorous, disease-free response. Third, *protection* of advanced growth from destruction must occur during overstorey harvesting. Fourth, post-logging *sanitation* by removal of residual trees may be required, increasing slash accumulation. Trees, especially subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) are prone to decay by Indian paint fungus (*Echinodontium tinctorium*).

Past studies and experience show that many important factors influence post-release growth. These are typically factors associated with pre-release growth (for true firs see, for example, Seidel 1977, 1980; Standiford 1978; Ferguson and Adams 1980; Helms and Standiford 1985). These factors include: pre-release height growth, live crown ratio, patterns of height growth, canopy conditions, tree size, and site climate. We assume that pre-release live crown ratio and height growth are prime indicators of tree vigour and release potential (Helms and Standiford 1985). The pattern of pre-release height growth—decreasing, constant, or increasing annual height growth—is important (Hatcher 1964). Tree age seems unimportant (also DeLong 1995).

The overall purpose of this study is to gain an understanding of the acclimation processes that lead to a successful post-release growth of advanced regeneration in Engelmann spruce (*Picea engelmannii*) and subalpine fir. The hypothesis is that pre-release height growth (e.g., 5-year growth), live crown ratio, tree size, and site series (e.g., site moisture) will explain most of the variation in post-release growth in Engelmann spruce and subalpine fir. The research objectives are:

- to determine density, spacing, and spatial distribution of advanced regeneration pre- and post-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 determine acclimative changes in released advanced regeneration; and
- to compare growth characteristics of advanced regeneration among species of Engelmann spruce and subalpine fir.

Applied objectives are:

- to develop predictive guidelines for the post-release growth of advanced regeneration in Engelmann spruce and subalpine fir;
- to identify the site, tree, and stand characteristics useful for determining post-release growth; and
- to determine density and spatial attributes for acceptable advanced regeneration.

Study Site

The Sicamous Creek research site is about 15 km east from the city of Sicamous. The site is 1550–1750 m in elevation, with a predominantly northern aspect. The forest is mostly mature, old-growth Engelmann spruce and subalpine fir on gentle terrain. Soils are sandy loams and several areas exist with shallow soil and rock outcrops, and with scattered pockets of saturated soils.

Sampling Design and Plot Layout

The overall purpose of the Sicamous Creek studies is to evaluate the effects of several forest management treatments on a range of forest resources in the Engelmann Spruce – Subalpine Fir biogeoclimatic zone (ESSF). The overall design at the Sicamous site provides a sampling population that consists of different opening sizes and site treatments. The research area at Sicamous is divided into three blocks according to the altitude. Each block is divided into five different treatment units. The following five treatments were created at Sicamous in the winter of 1994–1995: 0.1-ha opening size, 1-ha opening size, 10-ha opening size, selectively cut area, and untreated controls (see Vyse, this proceedings, page 4). A site series mapping was completed before the harvesting in the winter of 1994–95. The treatment units of the ESSFwc2 biogeoclimatic subzone include site series 01 (subalpine fir – azalea – oak fern), 04 (subalpine fir – rhododendron – heron's-bill), 06 (subalpine fir – valerian – oak fern), 08 (subalpine fir – horsetail – sphagnum), and 10 (sedge – sphagnum).

The five treatments within the three blocks were stratified into the different site series because site series (site moisture) may explain some of the variation in post-release growth in Engelmann spruce and subalpine fir. Openings for each site series and treatment were randomly selected from the population of openings in each block. The treatments were sampled by using systematic sampling and circular 10-m^2 plots. Systematic sampling is frequently used in surveys and inventories in British Columbia (Forest Productivity Councils of B.C. 1993; Provincial Cruising Advisory Committee 1996). This method of locating sample units is easier and cheaper than random sampling. However, the formulae for random sampling are used to calculate errors in the systematic survey (Freese 1962).

In the 0.1-ha openings, skid trails ran through the middle of the plot centre. Therefore, the plots were located randomly in the opening by using random numbers for the direction and the distance from the opening's centre. In the 1-ha openings, nine circular plots were systematically located with 33 m between plots. As only one 10-ha opening occurred in every block, every opening was included in sampling. The plot layout was the same as for 1-ha openings with nine plots. The distance between plots was 105 m. In addition, four local controls were established outside every 1- and 10-ha opening at both ends of the outermost plot lines. Nine control plots were established in the untreated control area in each of the three blocks. Finally, 30 plots were measured in an area close to the research area which was planted in 1987. These plots were also established systematically to compare the growth of Engelmann spruce and subalpine fir.

Measurements

When possible, all designs and measurements have attempted to meet the provincial standards for the establishment and remeasurement of permanent sample plots (Forest Productivity Councils of B.C. 1993) and for appraisal cruising (Provincial Cruising Advisory Committee 1996). The plot size of 10 m^2 (radius of 1.78 m) maintains accuracy in regeneration surveys because counting errors increase with an increase in plot size (Pohtila 1977). Each plot was marked with a rebar stake and every tree taller than 10 m with a metal number tag. For every plot, slope, aspect, and basal area of the stand were recorded and the amount of slash estimated. The altitude of each plot was determined from detailed topographic maps. A small plot of 1 m^2 was established at the centre of the sampling unit to count the number of germinants (i.e., seedlings < 10 cm high). All germinants were counted, but not identified by tree species.

The following were measured for every tagged tree: height (cm), leader growth of 1995 and five previous year's growth (accurate to within 0.5 cm), the length and width of the live crown (cm), diameter at the base and at breast height (mm). The distance of trees to the plot centre was determined to estimate stand spacing (clumpyness). The distance to the nearest tall tree, as well as its diameter were also measured. Stem form was classified according to the severity of deformity. Data on seedling microsite, damage, the colour of the needles, and finally the overall vigour of the tree were also collected. To estimate the age of advanced regeneration, core samples were taken from trees taller than 2 m.

Additional measurements include a survey in the summer of 1996 to estimate winter damage from snow press. In the fall of 1996 (two years after harvesting), branch samples that are taken across the distribution of tree sizes to measure the needle area and weight (specific leaf area) should enable estimates of adaptive changes in leaves (Tucker and Emmingham 1977). These samples will show changes in specific leaf area (SLA: m^2/g) two to three years before and two years after the harvesting. In the fall of 1999, a follow-up survey will be carried out including a free-growing survey (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1995).

Analyses

The intensive sampling data at the Sicamous Creek and Opax sites will provide data to formulate a model of post-release growth that will be both precise and site specific. Growth characteristics were analyzed using an analysis of variance model of randomized incomplete block design (Sokal and Rohlf 1981). The factors were: block (random, three levels), opening size (fixed, five levels), site series (fixed, five levels), and their interactions. In the following section we report preliminary results of this study. The results for the control plots include both the actual control blocks and the local control plots established in 1-ha and 10-ha openings.

Density of Advanced Regeneration

The density of advanced regeneration ranged from understocked 10-ha openings with 100 ± 81 (standard error of the mean) stems per hectare to well-stocked control plots with 2700 ± 580 stems per hectare. The density was 1430 ± 360 stems per hectare in the 0.1-ha openings, 1390 ± 270 stems per hectare in 1-ha openings, and 2130 ± 260 stems per hectare in the planted stand (Figure 1). The numbers of Engelmann spruce seedlings were significantly higher in the control plots when compared to the 10-ha openings ($p < 0.05$). In the control plots, subalpine fir were more significantly numerous than in the 1- and 10-ha openings ($p = 0.01$).

The scarcity of advanced subalpine fir and Engelmann spruce seedlings in the 10-ha openings was likely caused by site preparation completed before the plots became established. Compared to the control areas, all Engelmann spruce and 94% of subalpine fir seedlings in 10-ha openings were lost during harvesting and site preparation. In the other treatments, the survey was finished before the site preparation, and therefore the loss of seedlings was not that large. In the 1-ha plots, seedling losses were 50% for both Engelmann spruce and subalpine fir, and 33% for Engelmann spruce and 56% for subalpine fir in the 0.1-ha openings.

In most treatments, subalpine fir was the dominant species accounting for 60% of trees in the 0.1-ha openings, 72% in the 1-ha openings, and 100% in the 10-ha openings. In the control plots, however, the proportion of Engelmann spruce was slightly higher with 52% versus 48% (see Figure 1).

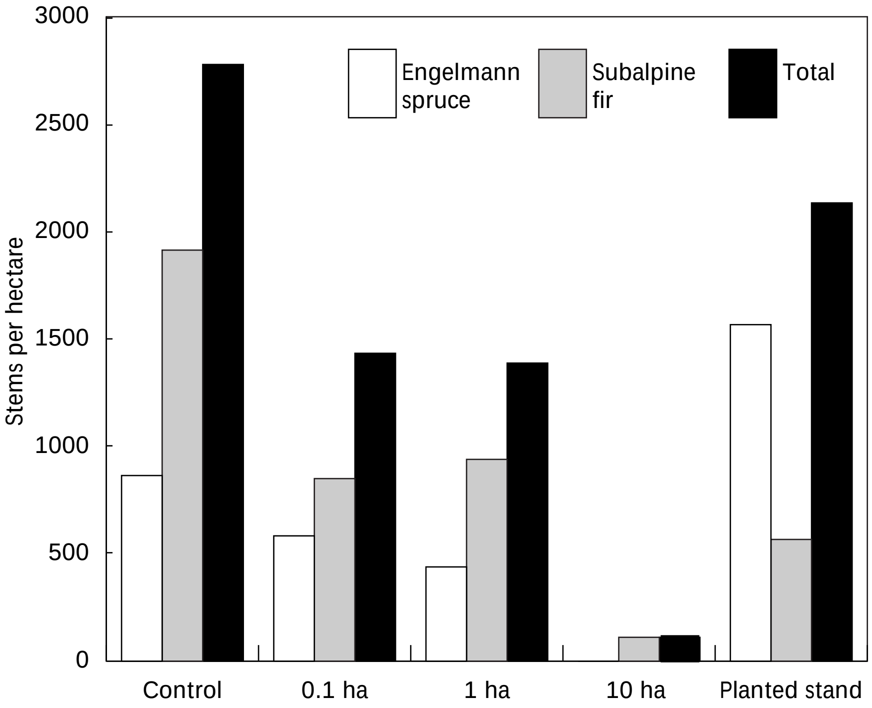

FIGURE 1 *Density of advanced regeneration in different treatments.*

Number of Germinants

The number of germinants in the control plots were significantly higher when compared to the 0.1-, 1-, and 10-ha openings ($p < 0.05$) (Figure 2). The density of germinants was 3770 ± 1050 stems per hectare in the control plots and 800 ± 300 stems per hectare in 0.1- and 1-ha openings. We found no germinants in 10-ha openings. Information about germinants in the planted stand is not yet available.

Most of the germinants (90%) have become established on old logs and stumps; only about 10% of the germinants grow on humus and most of these on humus mounds (Table 1).

Height and Height Distribution

As expected, the Engelmann spruce and subalpine fir seedlings in the planted stand were significantly taller than the seedlings in other treatments (Figure 3). For Engelmann spruce, the height ranged from 0.67 ± 0.11 m in planted area to 0.36 ± 0.07 m in the openings and controls. Compared to the Engelmann spruce seedlings, subalpine fir was taller in all treatments:

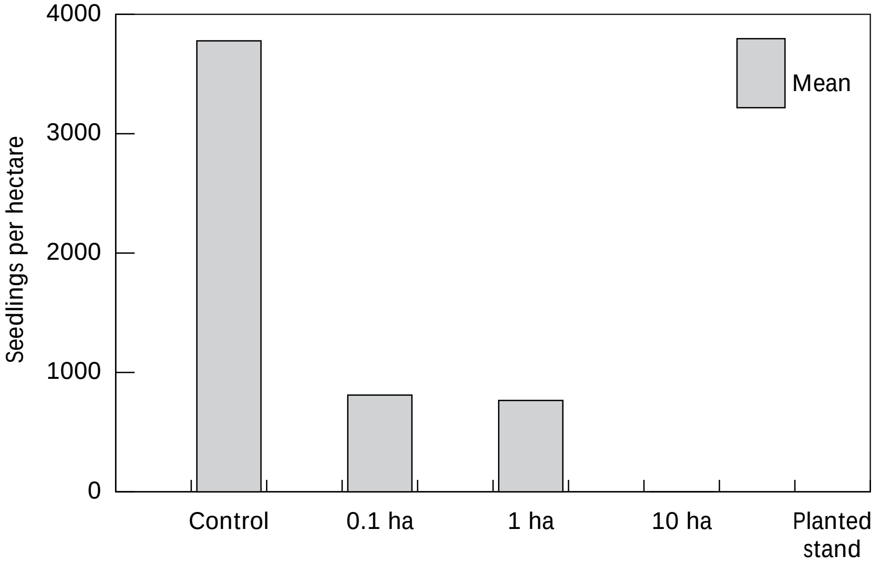

FIGURE 2 *Number of germinants in the different treatments.*

TABLE 1 *Percentage distribution of germinants on the microsites*

Treatment	**Old stump**	**Log**	**Humus, flat**	**Humus, mound**
Control	12.5	87.5	0.0	0.0
0.1 ha	18.2	72.7	0.0	9.1
1 ha	9.1	78.8	3.0	9.1
10 ha	0.0	100.0	0.0	0.0
Average	11.1	79.6	1.9	7.4

0.62 ± 0.05 m in the control plots, 0.57 ± 0.09 m in the 0.1-ha openings, 0.56 ± 0.05 m in the 1-ha openings, 0.28 ± 0.05 m in the 10-ha openings, and 1.17 ± 0.20 m in the planted stand (Figure 3).

The height distribution was similar for both species: 48% of Engelmann spruce and 36% of subalpine fir seedlings were smaller than 20 cm and 90% of Engelmann spruce seedlings and 80% of subalpine fir seedlings were smaller than 0.8 m (Figure 4).

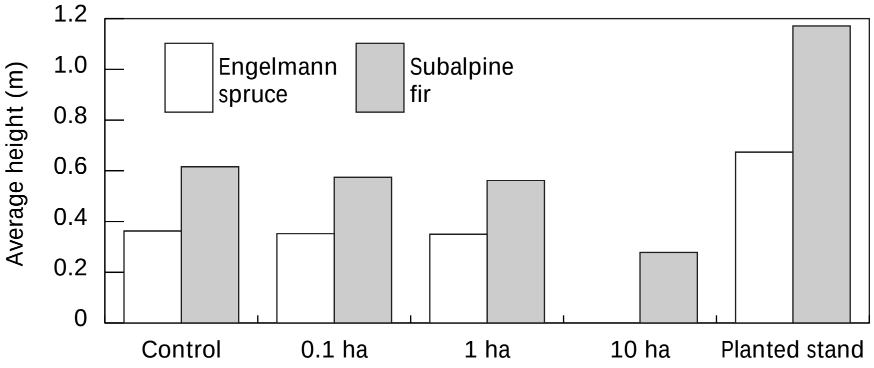

FIGURE 3 *Average height of advanced regeneration in different treatments.*

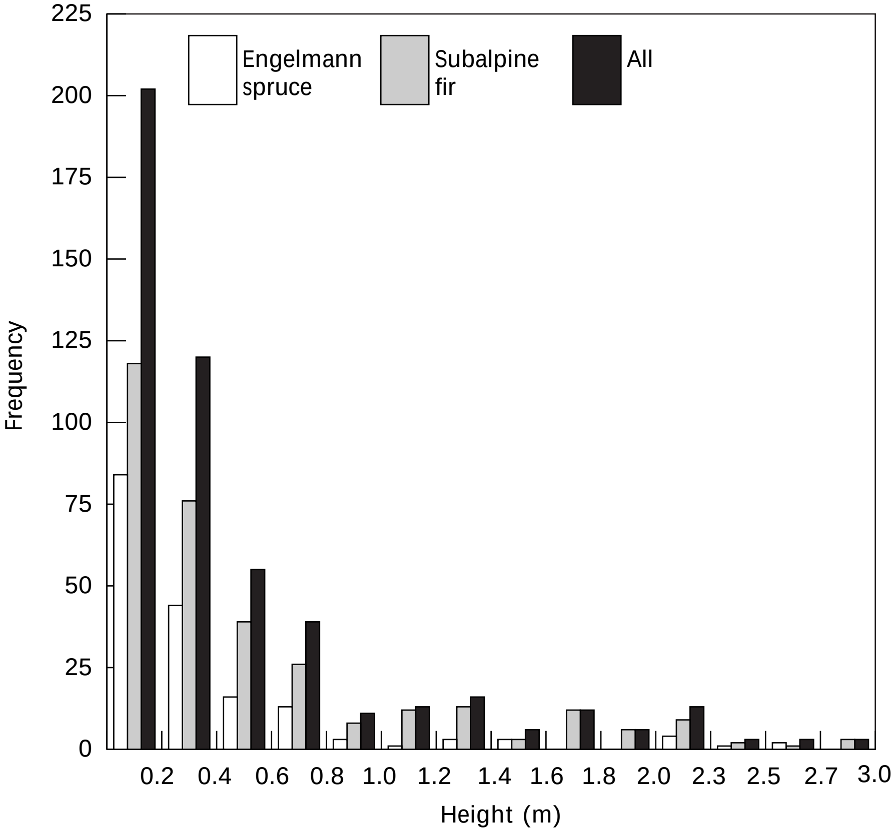

FIGURE 4 *Height distribution of advanced regeneration of Engelmann spruce and subalpine fir, and their total.*

Relative Growth Rate

The relative annual height growth of Engelmann spruce varied from 1.3 ± 0.14 cm to 2.1 ± 0.15 cm without any significant differences among the treatments (Figure 5). Subalpine fir also showed no significant differences between treatments; the growth rate varied from 0.5 ± 0 cm to 3.0 ± 0.13 cm (Figure 6). In the planted stand, height growth was significantly higher for both species; for Engelmann spruce the annual growth rate ranged from 10.8 ± 1.19 cm to 12.0 ± 1.60 cm and for subalpine fir from 11.4 ± 2.00 cm to 17.1 ± 2.93 cm.

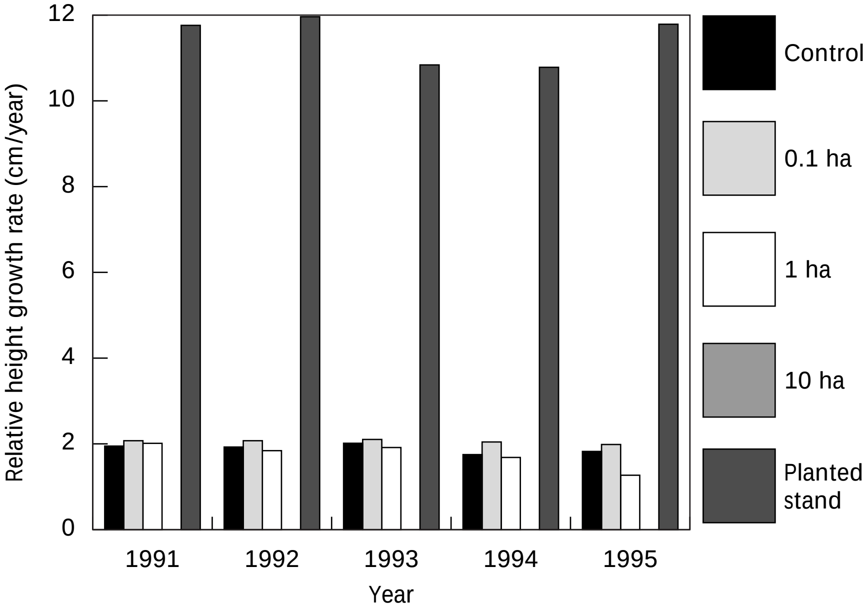

FIGURE 5 *Relative height growth of advanced Engelmann spruce regeneration.*

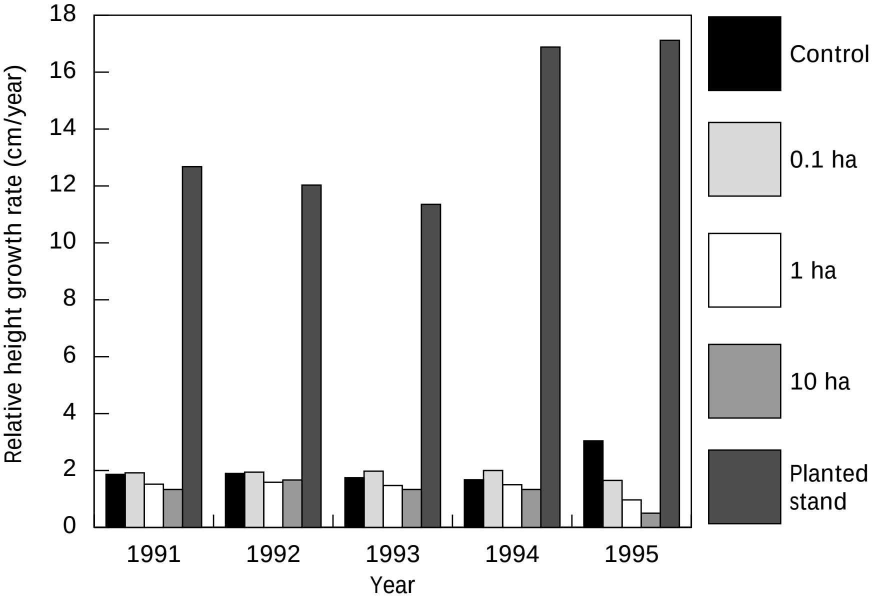

FIGURE 6 *Relative height growth of advanced subalpine fir regeneration.*

Diameter at Base

The average base diameter of Engelmann spruce in the study area ranged from 4.8 to 17.1 mm. In the control plots it was 6.1 ± 0.8 mm, in the 0.1-ha openings 7.5 ± 2.1 mm, in the 1-ha openings 4.8 ± 0.9 mm, and in the planted stand 17.1 ± 3.0 mm (Figure 7). The average diameter of subalpine fir was larger in all treatments: 18.6 ± 3.5 mm in the controls, 15.7 ± 2.7 mm in the 0.1-ha openings, 11.9 ± 1.2 mm in the 1-ha openings, 5.6 ± 1.5 mm in the 10-ha openings, and 29.6 ± 5.9 mm in the planted stand (Figure 7). No differences existed among treatments; only seedlings in the planted stand were significantly thicker.

Live Crown Ratio

The live crown ratio was relatively high in all treatments. The crown ratio of Engelmann spruce was 0.73 ± 0.01 in the control plots, 0.70 ± 0.03 in the 0.1- and 1-ha openings, and 0.88 ± 0.02 in the planted stand (Figure 8). The crown ratio of subalpine fir was lower in all treatments, ranging from 0.47 ± 0.02 in the 1-ha openings to 0.87 ± 0.03 in the planted stand (Figure 8).

Stem Form and Tree Vigour

Most of the Engelmann spruce seedlings assessed had relatively good form: 40% of the seedlings were straight and 32% of seedlings had only minor sweeps (Table 2). Less than one-third of the Engelmann spruce stems had basal sweeps: 19% had moderate sweeps and 9% extreme

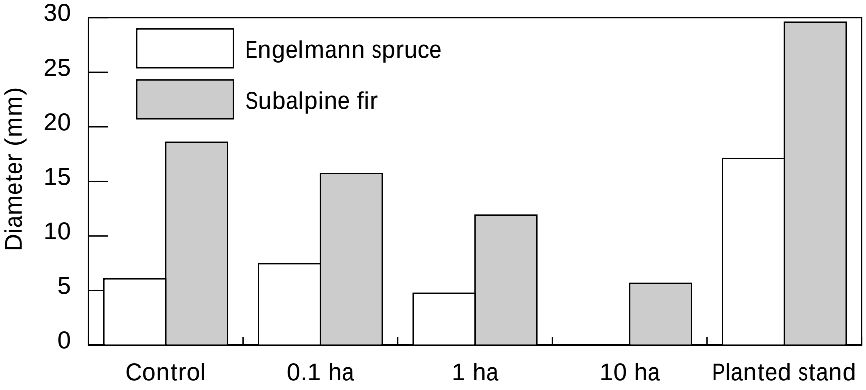

FIGURE 7 *Average diameter at base in different treatments.*

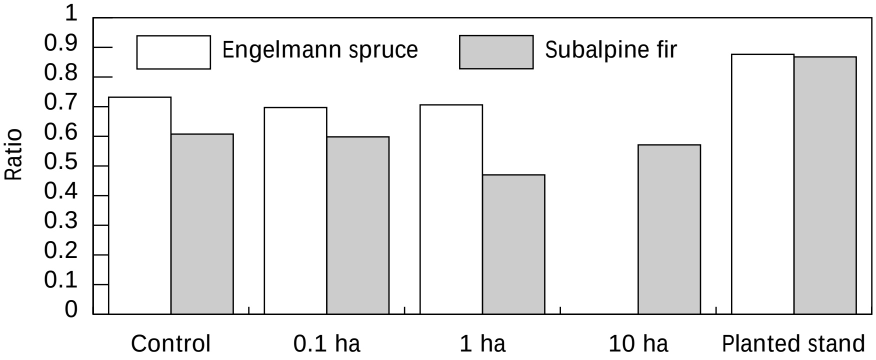

FIGURE 8 *Live crown ratio in different treatments.*

sweeps. For the subalpine fir seedlings, the stem form was worse. Less than one-third (28%) of seedlings had straight stems, 29% had minor sweeps, 23% moderate sweeps, and 21% extreme sweeps (Table 2).

About 44% of Engelmann spruce seedlings were class 1 trees with relatively good vigour, while 23% of seedlings had minor damages (Table 3). About one-fifth of the seedlings appeared weak and the remainder had poor vigour ratings. One-third of the subalpine fir seedlings were assessed in good vigour, about 25% had minor damages, 18% were weakened, and as many as 25% had poor vigour (Table 3).

TABLE 2 *Number and percentage of seedlings in different stem form categories*

		Engelmann spruce				**Subalpine fir**			
Treatment		**Straight**	**Minor sweep**	**Moderate sweep**	**Extreme sweep**	**Straight**	**Minor sweep**	**Moderate sweep**	**Extreme sweep**
Control	No.	25	26	14	8	55	51	30	27
	%	34.3	35.5	19.2	11	33.7	31.3	18.4	16.6
0.1 ha	No.	8	9	15	3	15	19	14	5
	%	22.8	25.7	42.9	8.6	28.3	35.9	26.4	9.4
1 ha	No.	15	17	7	4	22	29	29	34
	%	34.9	39.5	16.3	9.3	19.3	25.44	25.44	29.82
10 ha	No.	—	—	—	—	1	0	2	1
	%	—	—	—	—	25	0	50	25
Planted	No.	32	11	2	2	4	3	5	5
	%	68.1	23.3	4.3	4.3	23.5	17.7	29.4	29.4
Total no.		80	63	38	17	97	102	80	72
Average		40.4	31.8	19.2	8.6	27.6	29.1	22.8	20.5

TABLE 3 *Number and percentage of seedlings in different tree vigour categories*

		Engelmann spruce				**Subalpine fir**			
Treatment		**Good**	**Minor damages**	**Weakened**	**Poor**	**Good**	**Minor damages**	**Weakened**	**Poor**
Control	No.	34	18	16	5	77	40	19	27
	%	46.6	24.7	21.9	6.8	47.2	24.5	11.7	16.6
0.1 ha	No.	18	10	4	3	24	9	9	11
	%	51.4	28.6	11.4	8.6	45.2	17	17	20.8
1 ha	No.	8	12	13	10	8	29	29	48
	%	18.6	27.9	30.2	23.3	7.1	25.4	25.4	42.1
10 ha	No.	—	—	—	—	0	2	1	1
	%	—	—	—	—	0	50	25	25
Planted	No.	27	5	6	9	6	5	4	2
	%	57.5	10.6	12.8	17.2	35.3	29.4	23.5	11.8
Total no.		87	45	39	27	115	85	62	89
Average		43.9	22.7	19.7	13.7	32.7	24.2	17.7	25.4

Logging Damage

Only 17 stems in the study plots showed logging damage, which is only 3% of all stems. One-half of the damages resulted from falling trees and one-third by rubbing during the skidding. Most of the wounds were relatively small: 1–5 cm long and wide and 2–3 cm deep. Most of the trees (75%) were wounded below 1.3 m height.

Although the proportion of wounds caused by falling trees and skidding was relatively small, the actual amount of logging damage was higher. About 1800 seedlings per hectare (i.e., 65% of seedlings) were lost during logging.

Stocking Status

Stem form is an important aspect of second-growth management because it affects future merchantability, but many of the minor deformities are outgrown relatively quickly. To estimate future stocking status (i.e., the number of preferred and acceptable trees), trees with extreme and moderate stem deformities and those with poor vigour were not included in the calculation. As a result, about 50% of the existing advanced regeneration was estimated to not establish as crop trees. Only slight differences occurred between Engelmann spruce and subalpine fir: 47% of Engelmann spruce and 55% of subalpine fir were not likely to become crop trees.

The final density of advanced regeneration ranged from 100 stems per hectare in the 10-ha openings to 1670 ± 400 stems per hectare in the control areas. The minimum stocking standard is 700 stems per hectare and the target stocking standard is 1200 stems per hectare (Figure 9). The density in the other treatments was under the minimum requirements: 500 ± 230 stems per hectare in the 0.1-ha openings and 460 ± 110 stems per hectare in the 1-ha openings. The density in the planted stand was 1330 ± 230 stems per hectare, which is significantly higher than in the treatment areas (Figure 9). Significantly more Engelmann spruce in the controls were assessed as crop trees compared to the other treatments ($p < 0.01$).

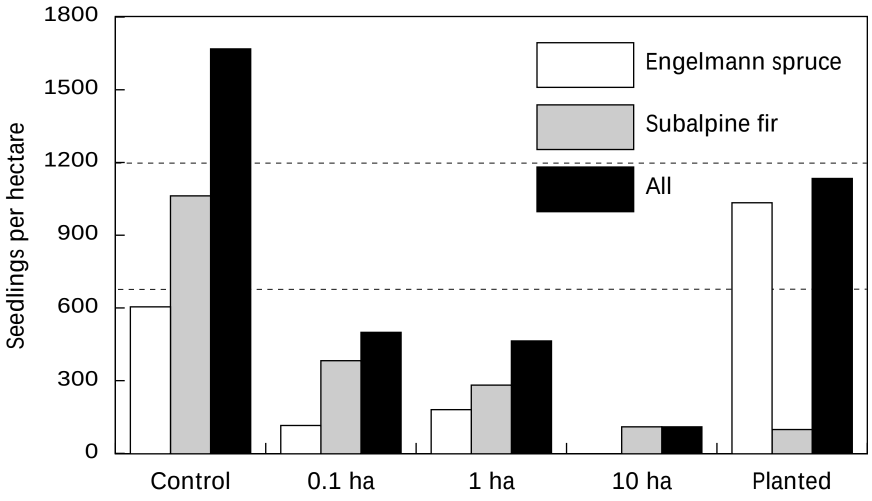

FIGURE 9 *Estimates of crop tree density; lines indicate minimum (700 stems per hectare) and target (1200 stems per hectare) stocking standards.*

CONCLUSIONS

The number of advanced regeneration seedlings (2700 stems per hectare) and germinants (3700 stems per hectare) in control treatments clearly shows the potential for using natural regeneration. Harvesting, however, destroyed over 60% of advanced regeneration and about 80% of germinants. Improving the results of harvesting for protection of advance regeneration may be accomplished by using cut-to-length harvesting systems, which concentrate slash and wood piles. This option should be studied.

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Faunal Biodiversity Studies at Sicamous Creek: Background and Rationale for the Choice of Species Indicator Groups

WALT KLENNER AND DAVE HUGGARD

ABSTRACT

The approach and rationale for faunal biodiversity studies at Sicamous Creek are discussed in the context of recent legislative changes, and the habitat structures and patterns created by forest management activities. The species being monitored at Sicamous Creek represent a diverse range of taxa that use different habitat structures, and that perceive the habitat patterns created by the experimental treatments at different scales because of their life history requirements. Other studies on flora, tree regeneration, and soil organisms complement the vertebrate and invertebrate species being monitored in the faunal biodiversity program.

INTRODUCTION

Maintaining biodiversity has become a recognized and 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 fifty years, but with the implementation of the Forest Practices Code, maintaining biodiversity has become an important obligation for forest managers. To maintain species, genetic, and ecosystem diversity, the *Biodiversity Guidebook* (B.C. Ministry of Forests and Ministry of Environment, Lands and Parks 1995) recommends 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 a manageable surrogate for maintaining biodiversity, versus the unwieldy alternative of managing each of the thousands of species and ecological processes on an individual basis. However, little direct evidence exists to indicate that the habitat targets specified in the biodiversity guidelines are adequate, or if they are too conservative and thus represent an excessive and unnecessary constraint to the timber industry. These are important issues, since not achieving biodiversity objectives and unnecessary constraints both have a high cost.

Before the management of our forests for timber commodities, a wide range of natural disturbances (e.g., wildfire, insect attack, windthrow, etc.; 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; Spies and Franklin 1991; Mladenoff et al. 1993); and within stands, an abundance of habitat structures such as snags and downed wood was maintained (Franklin and Spies 1991a, b; Spies and Franklin 1991). This complex and heterogeneous mosaic of habitat structures and patterns is important to maintain 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 mosses and lichens (Seastsdt 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 1991a, b; Spies and Franklin 1991; Lee et al. 1995a, b; Roy et al. 1995; Schieck and Nietfeld 1995). The numerous large declining green trees, snags, and abundant downed wood characteristic of old-growth forests will decrease in managed forests unless special practices are implemented to maintain these features (Swanson and Franklin 1992). Trees, snags, and downed wood are not the only habitat structures that are likely to diminish in managed forests. Other habitat structures, such as a thick layer of forest floor litter and terrestrial mosses or lichens, support a diverse arthropod and invertebrate fauna that is important in food webs and nutrient cycling (Powers 1989). The effects of canopy removal through partial or clear cutting, or the effects and duration of site preparation disturbances on the forest floor litter layer and its associated fauna are largely unknown. In addition, the effects of a reduction in 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 effects of forest harvesting on some forest-dependent species may be exacerbated by the biota (e.g., nest predators) that flourish in early seral habitats (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; Saunders et al. 1991; Lehmkuhl and Ruggiero 1991; Mladenoff et al. 1993). These changes will benefit some species and be detrimental to others, as the foraging, breeding, or cover capabilities of the habitats are modified.

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

1. identify the habitat structures and patterns that are likely to diminish or change as a result of forest management activities;
2. document the magnitude of these habitat changes at the Sicamous Creek forest harvesting trials;
3. 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 which are viewed as surrogates for the overall objective of managing for faunal biodiversity; and
4. develop landscape models that incorporate the results from the Sicamous Creek research, as well as other related studies into temporally and spatially explicit habitat supply models.

Six faunal indicator groups were chosen and are being monitored at Sicamous Creek. These indicator groups were chosen for two reasons: (1) information on their natural history suggests that they require specific habitat or landscape attributes, which will likely diminish in managed forests, and (2) the groups or species within a group are likely to perceive habitat patterns at different scales (Table 1; e.g., a home range of 0.1 ha for shrews vs. 100 ha for small carnivores). The six groups being monitored in relation to forest management practices span a diverse range of taxa and reflect a variety of trophic interactions (plant-herbivore-predator). Shifts in the population density, demography, home range, habitat use, and community structure of the monitored groups should provide reliable indicators of change in overall faunal biodiversity at Sicamous Creek. Ongoing studies by T. Dickinson on passerine birds (this proceedings, page 254), and by D. Lloyd and T. Goward on plants and lichens, respectively (this proceedings, pages 57 and 67), complement our approach and will be combined to provide a more complete picture of changes in biodiversity arising from the experimental treatments. Large, wide-ranging vertebrates, such as grizzly bear (*Ursus arctos*), cougar (*Felis concolor*), or goshawk (*Accipiter gentilis*), were not encountered frequently enough at the Sicamous Creek trials, hence the effects of the research treatments on these and other wide-ranging species can only be inferred from changes in the preferred habitat and prey populations we are monitoring. The experimental harvesting and site preparation treatments at Sicamous Creek form a unique opportunity to evaluate the response 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 partial cut and 0.1-ha patch 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

TABLE 1 *Indicator groups being monitored to assess the effects of forest management on faunal biodiversity at Sicamous Creek*

Taxon	Home range size (ha)	Potentially important habitat structures	Potentially important habitat patterns	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
Shrews Masked Montane Vagrant	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 Heather vole Red-backed vole Long-tailed vole Meadow vole Northern bog lemming 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 leave strips; habitat dilution within home range	Competition within the group; predation by small carnivores
Grouse Spruce grouse	1.0–10	Conifers; canopy cover; topographic features	Habitat edges; use of forested corridors; habitat dilution within home range	
Cavity-nesting birds Three-toed woodpecker	10–100+	Snags; conifers; downed wood	Habitat edges; use of forested corridors; habitat dilution within home range	
Small carnivores (winter only) Short-tailed weasel Marten	10–1000	Canopy cover; downed wood; conifers; snags	Habitat edges; use of forested corridors; habitat dilution within home range	Predation on small mammals

management practices in Engelmann Spruce – Subalpine Fir (ESSF) zone forests.

These silvicultural systems trials are important from another perspective: project infrastructure (e.g., roads, GPS locations of study sites, co-ordination of treatments and studies, etc.) will be important in the long-term monitoring of these sites. In the ESSF biogeoclimatic zone, harvesting initiates a series of seral changes that proceed towards the establishment of a mature coniferous forest. The acute, 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 the effect is likely to be species specific and is not well known. For example, shrew abundance and diversity may return to pre-harvest levels on the treated sites within 10 years, while cavity-nesting birds may not find suitable habitat structures for 80 years or

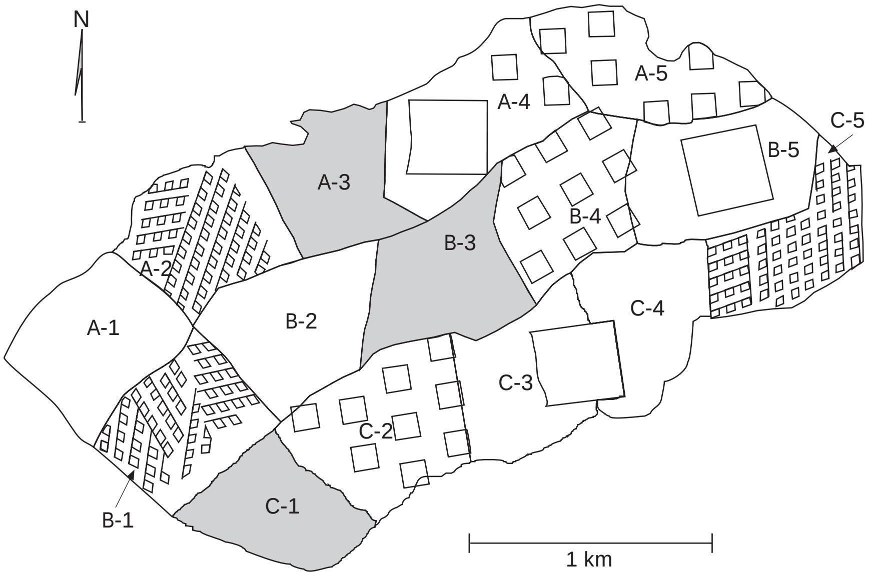

figure 1 *Schematic diagram of the Sicamous Creek research site indicating locations of control (A-1, B-2, C-4), uniform partial cut (A-3, B-3, C-1), 0.1-ha patch cut (A-2, B-1, C-5), 1.0-ha patch cut (A-5, B-4, C-2), and 10-ha clearcut (A-4, B-5, C-3) treatments.*

more. The study site infrastructure of the Sicamous Creek silvicultural systems trials will facilitate and encourage the long-term monitoring of habitat changes and testing of habitat suitability models over time.

The rationale for choosing each of the monitored indicator groups is presented in the series of reports that follow. Two approaches are being used to study these species. Initially, the monitoring effort will focus on evaluating habitat suitability models for habitat structure and pattern. 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 mice and voles indicator group, the habitat suitability monitoring program is being conducted in conjunction with a hypothesis-testing manipulation of downed wood. This approach is used because other studies suggest that the abundance and distribution of downed wood has a strong influence on mouse and vole populations. A similar approach will be applied to the other indicator species as habitat suitability models are developed.

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Terrestrial Arthropod Sampling at Sicamous Creek: Progress Report

DAVE HUGGARD AND WALT KLENNER

ABSTRACT

A multiple-scale, multiple-year sampling design is used to determine the habitat and landscape relationships of terrestrial arthropods and to ascertain how this group is affected by forest management. Specimens are classified using a "taxonomic hierarchy" approach: all arthropods are identified to class; insects are identified to order; beetles and a few other groups are identified to family; and carabid beetles are identified to species. Classification is in the preliminary stages, but variability at several scales is apparent even in the preliminary pre-harvest results.

INTRODUCTION

Maintaining arthropod communities in managed forests requires understanding the relationships between habitat and landscape features and arthropods, and how this group is affected by forest management. Determining habitat relationships and the effect of forest management is particularly challenging for terrestrial arthropods for two reasons:

1. Arthropods are the most diverse group of organisms in the province, with over 50 000 species (Pojar 1993). Terrestrial arthropods are a large component of this diversity, including centipedes, millipedes, mites, spiders, and many orders of insects.
2. Abundance of terrestrial arthropods can be highly variable, with large differences in population sizes and community composition in different forest types, at different sites within a forest type, and in particular, microhabitats; populations and communities also change seasonally and annually, and in response to weather events and forest succession.

To handle the first source of complexity—the huge diversity of terrestrial arthropod species—we used a "taxonomic hierarchy" approach. All terrestrial arthropod specimens were identified to class, insects were identified to order, beetles were identified to family, and carabid beetles were identified to species. In some studies, classification to broad taxonomic group (class or order) has proven adequate (Williams and

Gaston 1994), but these studies have generally focused on broad-scale patterns of diversity. Individual species of carabid beetles show distinct habitat preferences and responses to disturbances (Holliday 1991; Niemala et al. 1992), suggesting that coarse taxonomic classification may be an insufficient or misleading indicator of responses to the local effects of forest management. The taxonomic hierarchy approach will allow us to test whether broad taxonomic groups are adequate for studying the effects of forest management, or whether detailed species identification is required. We will also be able to address the question of whether one taxonomic group can be used as an indicator of the local effects of forest management on the entire diversity of terrestrial arthropods (cf., Pearson and Cassola 1992).

To accommodate the second source of complexity—the high variability of terrestrial arthropod populations and communities—we use a multiple-scale, multiple-year sampling design. The design allows us to measure several sources of variability and also to factor out those that we can not manage (e.g., annual variability) to assess more precisely the factors that we can manage (e.g., harvest treatments, habitat, and landscape changes). The multiple-scale design also allows us to estimate the relative importance of different variance components—for example, whether or not harvesting effects are large compared to the natural annual and seasonal variability in arthropod abundances, or whether landscape relationships are important compared to microhabitat relationships.

A large number of terrestrial arthropod samples have been collected at Sicamous Creek. We are beginning to classify the specimens to broad taxonomic group, and initiating classification of carabid beetles to species. This report summarizes progress to date, and initial patterns of arthropod abundance.

METHODS

We have sampled terrestrial arthropods using pitfall traps. Five individual pitfall traps are arrayed in a 4 m radius "trap circle"; three circles 50 m apart are used in each replicate block. Traps were sampled for two, two-week sessions in late summer in 1992, 1993, 1994 (pre-treatment), and 1995 (post-treatment). Additional early summer sessions were added in 1996. Winter trapping, using insulated "chimneys" to access traps without disturbing the snow cover, was conducted in 1992–93 and 1993–94 (pre-treatment) and 1995–96 (post-treatment).

We established 1 m radius plots around each pitfall trap and 5.65 m radius plots around each trap circle, measuring a variety of habitat variables at each scale. The pitfall traps and habitat plots are the same as those used in the shrew study at Sicamous Creek. (See Huggard and Klenner [this proceedings, page 200] for details of the habitat measurements and sampling design.)

We are identifying all arthropod specimens to class (e.g., Diplopoda, Arachnida, Insecta); insects are identified to order (e.g., Diptera, Coleoptera); beetles are identified to family, and carabid beetles are identified to species. Other obvious families within some insect orders are also separated (e.g., Formicidae within Hymenoptera), and we plan to identify one additional family to the species level. We initiated a subsampling scheme in which only two of five samples per trap circle are processed to provide preliminary estimates of treatment effects and variance components more quickly.

PROGRESS AND RESULTS TO DATE

A total of 1680 summer samples and 243 winter samples were collected before harvesting, and 660 summer and 198 winter samples have been collected after harvesting. We have classified 850 summer pre-treatment samples (51%) and 153 (63%) winter pre-treatment samples to broad taxonomic group. No post-treatment samples have been classified yet. Initial identification of carabid beetle species collected at the site is being conducted by S. Lavallee (University of British Columbia, Department of Zoology).

Summaries of the preliminary results confirm that several sources contribute to high variability in arthropod abundances. Some taxa show high variability among individual traps. For example, some traps caught over 100 spiders per session, while adjacent traps caught none (Figure 1). The traps with high numbers of a particular taxon sometimes show consistent results between sessions or years, suggesting that microhabitat characteristics at the trap site may be important, but in other cases, high numbers are only recorded in one session or one year, suggesting less predictable local variability.

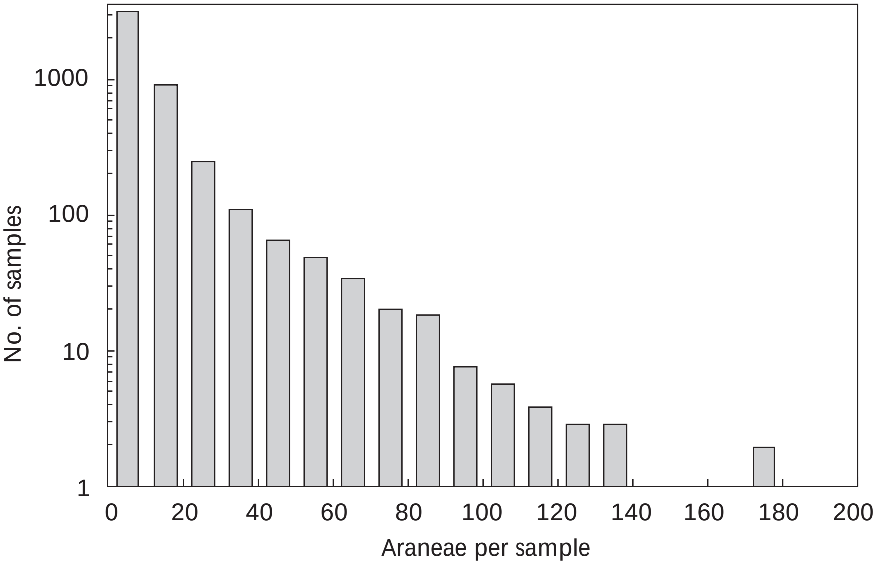

FIGURE 1 *Histogram of number of Araneae per sample, Sicamous pre-treatment collections; note log scale on Y-axis.*

Annual variation and variation between sessions within a year is obvious for some taxa (e.g., Carabidae, Chilopoda; Figure 2). Differences between the lower-, middle-, and upper-elevation replicates is sometimes

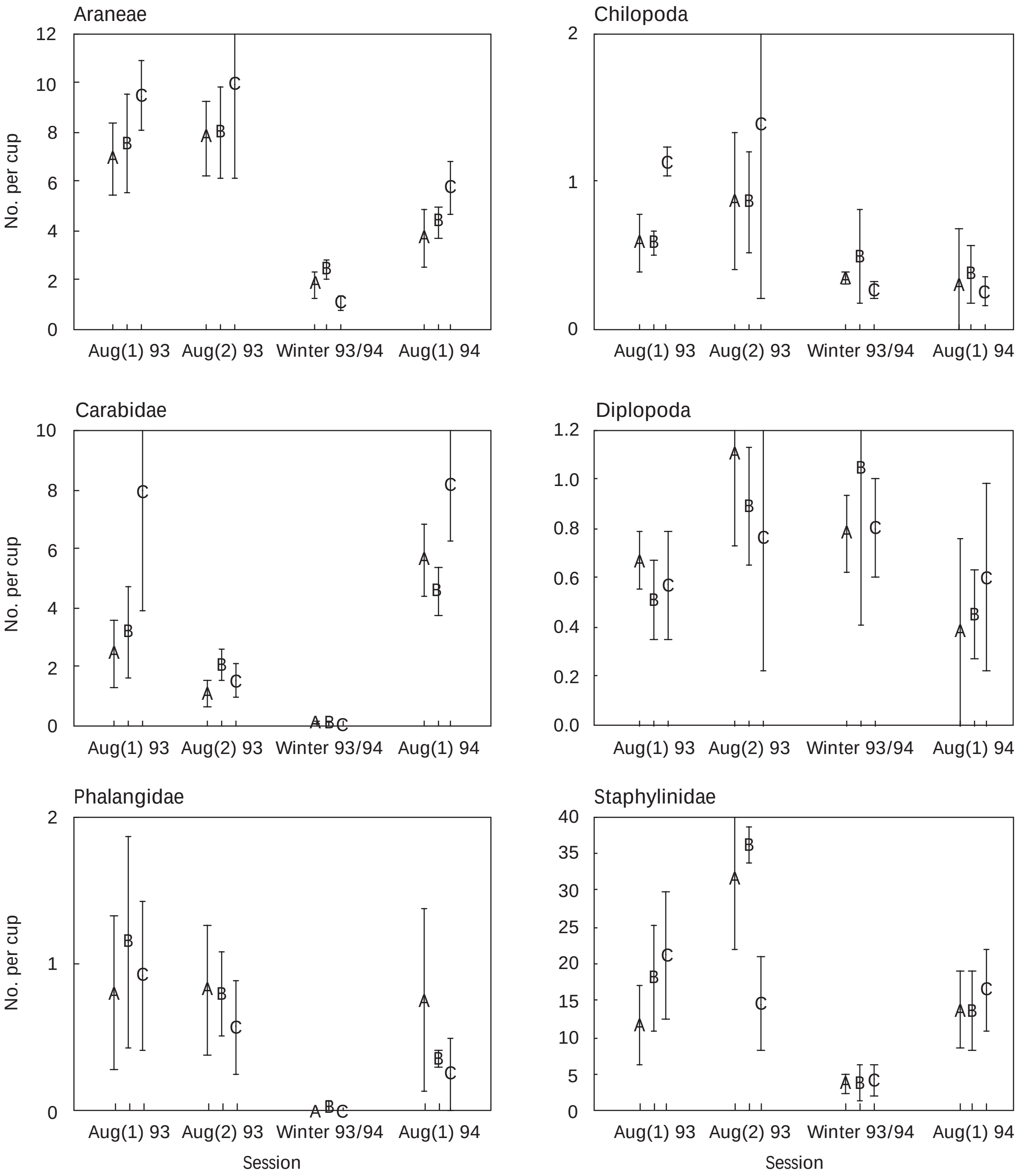

FIGURE 2 *Initial results for several terrestrial arthropod taxa. Two sessions from August 1993, one from winter 1993–94 and one from August 1994 (all pre-treatment) have been classified to date. Only parts of some sessions have been classified to date. A = lower, B = middle, and C = upper replicate blocks. Error bars are 2 SE, based on between-block variances.*

pronounced for certain groups, while others do not show this effect (Figure 2). Variation between blocks within the A, B, or C replicate blocks is also occasionally high (indicated by wide error bars in Figure 2), even though all results to date are from uncut forest before harvesting. When a more complete set of samples has been classified, we will use variance partitioning techniques to determine the relative importance of these different sources of variability for different taxa, and regression techniques with the two scales of habitat variables to determine how much of this variation can be explained by relationships with measured habitat variables. When post-treatment samples are classified, we will also be able to assess the relative importance of harvesting effects compared to this observed natural variability.

Traps set under the snow in the winter showed that certain taxa were abundant and active at this season (e.g., Diplopods, Chilopods; Figure 2), while other groups that are common in the summer are rarely captured in winter (e.g., Carabidae, Phalangidae; Figure 2). Some generally rare groups, such as Grylloblatids and wingless Dipterans, were only found in winter samples. Little is known about arthropod communities under the snow, even though high-elevation sites such as Sicamous Creek have snow cover for more than eight months of the year.

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Initial Effects of Harvesting on Shrews, Immature Rodents, and Forest Floor Habitat at Sicamous Creek

DAVE HUGGARD AND WALT KLENNER

ABSTRACT

We used a multiple-scale design of pitfall traps and habitat plots at Sicamous Creek to assess the effects of harvesting on shrews and other small mammals, and to develop models of the habitat relationships of these species. Three pre-harvest and one post-harvest year of sampling have shown large annual variation in abundance of these species. There were no clear differences in abundance between uncut forest, edges, partial cuts, and clearcuts in the first post-harvest summer, but the annual variability, short time since harvesting, and predominance of immature animals in the collections limit our confidence in these initial results. Most habitat characteristics, which are potentially important to shrews, were changed by harvesting, but coarse woody debris volume, size, and profile class composition were not significantly different between uncut and harvested sites.

INTRODUCTION

Maintaining populations of shrews should be an objective in managed forests, both because these species are a component of biodiversity and because they play several roles in forest ecosystems. Shrews prey on forest insects, which may assist in maintaining a diverse insect community and reduce the risk and severity of outbreaks of forest "pest" insects (Hanski 1987). Shrews are prey for larger forest carnivores, sustaining their populations when other small mammals are at low densities (Korpimäki and Norrdahl 1989).

The effects of forest harvesting on shrew populations are poorly understood. Negative effects are possible because shrews are associated with several forest floor characteristics that are affected by harvesting. With small bodies and high metabolic rates, shrews are thought to need moist forest floor conditions and moderate surface temperatures (Getz 1989). Forest harvesting removes or reduces the canopy which leads to more extreme high and low temperatures on the ground and may reduce surface moisture through increased evaporation (Chen et al. 1993). During harvesting and site preparation activities, the moss and duff layers are reduced, thereby reducing the foraging habitat for shrews (Terry 1981). Shrews are also associated with coarse woody debris (Craig 1995), which can be reduced by harvesting and silvicultural activities. Background on the ecological roles and habitat requirements of small rodent species is provided in Craig et al. (this proceedings, page 243).

Determining habitat relationships for shrews and small mammals, and how populations of these species will respond to forest management is difficult, because small mammal populations are highly variable (Henttonen et al. 1989). Part of this variation results from variation in habitat and landscape features, and can be managed through silviculture and harvest planning. Other sources of variation include annual differences in weather and certain severe weather events (Pankakoski 1985), spatial heterogeneity, and ecological interactions with other species. These sources of population variability can not be controlled through management. However, we must account for these sources of variability otherwise this added variation will mask the habitat and landscape relationships which are useful for management. Determining habitat relationships and harvesting effects is made even more complicated because the relationships may change from year to year depending on weather conditions and population densities, and they will differ between species and forest types.

The shrew study at Sicamous Creek is one of three shrew studies we are conducting in different biogeoclimatic zones. These studies all use a multiple-year, multiple-scale sampling design. This design allows us to measure many of the sources of variability in shrew populations, to provide reliable information on the habitat and landscape relationships of shrew species, and to understand the effects of forest management. This requires extensive field sampling and data analysis, which has not yet been completed; this paper outlines our methods and summarizes results to date. The pitfall traps used in the study also sample immature voles. This work therefore complements an extensive small mammal live-trapping program at Sicamous Creek, which catches mainly larger voles. Initial results for the immature rodents are also presented here, although these are intended to be integrated with the live-trapping study, rather than used separately.

METHODS

Field Sampling and Specimen Identification

We sampled shrews and immature voles using pitfall traps. Pitfall traps are plastic cups (500 ml) set in the ground with their top flush with the ground surface. A 30 × 30-cm cover board, which keeps rain and debris out of the sample, is held 10 cm above the trap by three stakes. Traps are set by pouring 100 ml (3 cm deep) of propylene glycol, a non-toxic, non-volatile liquid preservative, into the cup. Each trapping “session” lasts for two weeks, after which the contents of the trap are strained out and the trap closed. Arthropods are stored in ethanol for future identification (see summary of terrestrial arthropod study, this proceedings, page 195). We weighed the shrews and determined their sex and reproductive status based on internal reproductive organs. Species identification is based on dental characteristics, except in the case of older *Sorex vagrans* and *S. monticolus* with worn teeth, whose identification uses a modified discriminant function based on skull measurements (Woodward 1993).

Sampling Design

We used a double-nested sampling design. Five individual pitfall cups were set in a 4 m radius plot to form a "trap circle." Three trap circles were placed 75 m apart in each replicate cutblock or stand. There is one set of three circles per 35-ha study block (Figure 1) except in 1993, when five circles were placed 30 m apart in a small cross-pattern. In 1992, before block layout, 21 sets of three circles were placed across the site. Two consecutive two-week trapping sessions were conducted between early August and early September in 1992, 1993, 1994 (pre-harvest), and 1995 (post-harvest).

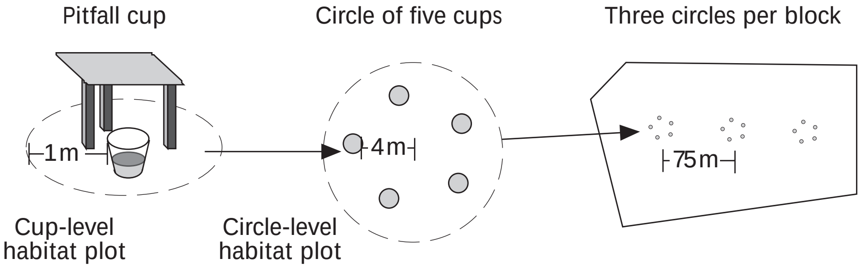

FIGURE 1 *Pitfall sampling design at Sicamous Creek. A 1 m radius habitat plot is established around each trap cup; five cups make up a trap circle, with a 5.6 m radius habitat plot; three circles are in each study block.*

Habitat Description

Plots to describe habitat features were established at two scales.

Cup-level plots We recorded the following in a 1 m radius plot centred on each trap cup:

- ground moisture (dry, mesic, or wet);
- surface soil composition (organic components, mineral soil, or both);
- percentage cover of moss and average depth;
- percentage cover of fine debris (litter) and average depth;
- grass cover (none, medium, or high);
- herb-layer cover (none, medium, or high) and the three most common herb-layer species;
- shrub-layer cover (none, medium, or high) and the three most common shrub species;
- distance to the nearest piece of downed wood greater than 7.5 cm diameter, its species, diameter, height above ground, and decay class (following Thomas 1979); and
- distance to the nearest tree, snag, or stump, greater than 7.5-cm dbh and its species, dbh, height, and profile class (following Thomas 1979).

Circle-level plots We recorded the following in a 5.6 m radius plot centred at the middle of each circle of five traps:

- an index of ground moisture, from 1 (xeric) to 5 (hydric);
- percentage cover of moss and average depth;
- percentage cover of fine debris and average depth;
- canopy cover;
- percentage cover of herb layer and the five most common herb-layer species;
- percentage cover of shrub layer and the five most common shrub-layer species;
- the species, dbh, height, and profile class (following Thomas 1979) of each tree greater than 7.5-cm dbh;
- the number of small conifers (< 7.5 cm, but over 30 cm tall) by species;
- the distance to any edges within 100 m; and
- the slope, aspect, and a description of topography.

We used the perimeter of the 5.6 m radius plot as an intercept transect for coarse woody debris, recording the following for each log greater than 7.5 cm in diameter at the point of interception: species, diameter, and height at the point of interception, and decay class (following Thomas 1979). An estimate of woody debris volumes per plot by decay class is derived from this information following Van Wagner (1968).

Sampling Across Edges

Additional trap circles were added to the patch-cut treatment blocks after harvesting to sample shrews and voles at the following distances from edges in each size of patch cut:

- 0.1-ha: 15 m into cutblock (centre), edge, and 15 m into leave strip (centre of leave strip);
- 1-ha: 50 and 25 m into cutblock, edge, 25 and 50 m into leave strip;
- 10-ha: 100, 50, and 25 m into cutblock, edge, 25, 50, and 100 m into forest.

Habitat plots were also conducted at these traps.

Analysis

For this summary, we calculated mean abundance and standard errors for adult and immature *Sorex cinereus* and *S. monticolus,* and for immature red-backed voles, heather voles, and bog-lemmings in the silvicultural systems treatments (i.e., partial cut, 0.1-ha patch-cut arrays, 1-ha patch-cut arrays, 10-ha clearcuts, and controls) using circles as subsamples of each block. The five traps in each circle and the two sessions each year are considered a single sample. We also calculated abundance of shrews and voles in four different "harvest types": uncut, partial cut, clearcut edge, and clearcut, again using circles within a treatment block as subsamples. This is a finer-scale analysis, based on the harvest type of each circle (i.e., a circle in the leave strip of a 1-ha array would be considered "uncut"; a nearby circle in the 1-ha patch cut would be considered "clearcut"). The significance of the differences was tested using nested analyses of variance. A complete analysis of sources of variation will be conducted when field data collection is complete.

RESULTS AND DISCUSSION

Species and Reproductive Status

A total of 1496 vertebrates of 11 species have been collected in four years. Two shrew species and three rodents comprise 96% of the sample: *Sorex cinereus* 46.9%, *S. monticolus* 20.3%, red-backed vole (*Clethrionomys gapperi*) 12.6%, heather vole (*Phenacomys intermdius*) 10.6%, and bog-lemming (*Synaptomys borealis*) 5.6%. *Sorex vagrans* (five individuals) and *S. palustris* (one individual) were the only other shrews caught.

Most animals we caught were immature: 92.1–95% of *Sorex cinereus* and 90.1–100% of *S. monticolus* in the four years, and virtually all of the rodents. We also saw abundant immature animals at other sites in the August collections, which we interpreted as primarily dispersing juveniles. Spring samples at other sites caught mainly adults, and late fall samples caught lower numbers of immatures, which may be post-dispersal juveniles. Dispersing animals may not provide a good index of micro-habitat or treatment use by shrews, but the short snow-free period at Sicamous Creek restricts our sampling in other seasons.

Annual Variation and Differences Between Treatments and Harvest Types

The five common species showed large annual variation in abundance in the uncut areas over the four years of sampling, with maximum: minimum ratios ranging from 3.5:1 for red-backed voles and 3.8:1 for *Sorex cinereus*, to 14.5:1 for bog-lemmings (Figure 2; uncut forest). These fluctuations were fairly consistent across blocks.

We captured enough individuals to test for post-harvest differences in abundance between harvest types (uncut, partial cut, clearcut edge, and clearcut) for five groups: mature *Sorex cinereus*, immature *S. cinereus*, immature *S. monticolus*, red-backed voles, and heather voles. None of these groups showed a significant difference between harvest types ($p > 0.23$ in all cases). However, two species showed distinct trends in numbers: immature *S. cinereus* were most abundant in uncut forest and edges, intermediate in partial cuts, and least abundant in clearcuts (Figure 2); red-backed voles were most abundant in uncut forest, intermediate in partial cuts, and least abundant in clearcuts and edges (Figure 2).

No significant differences were found between the overall treatments (Figure 3; $p > 0.17$ in all cases). This was expected because no obvious differences existed between the harvest types (i.e., uncut, partial cut, clearcut edge, and clearcut) that comprise the overall treatments. These results should be treated cautiously for several reasons:

- The post-treatment collections were made half a year after harvesting, and only one to two months after snowmelt following harvesting. Even species like shrews with high population turnover may not have responded to habitat changes so quickly—the adults caught in this sample were on the site during logging and the immatures were likely born under the snow soon after logging.

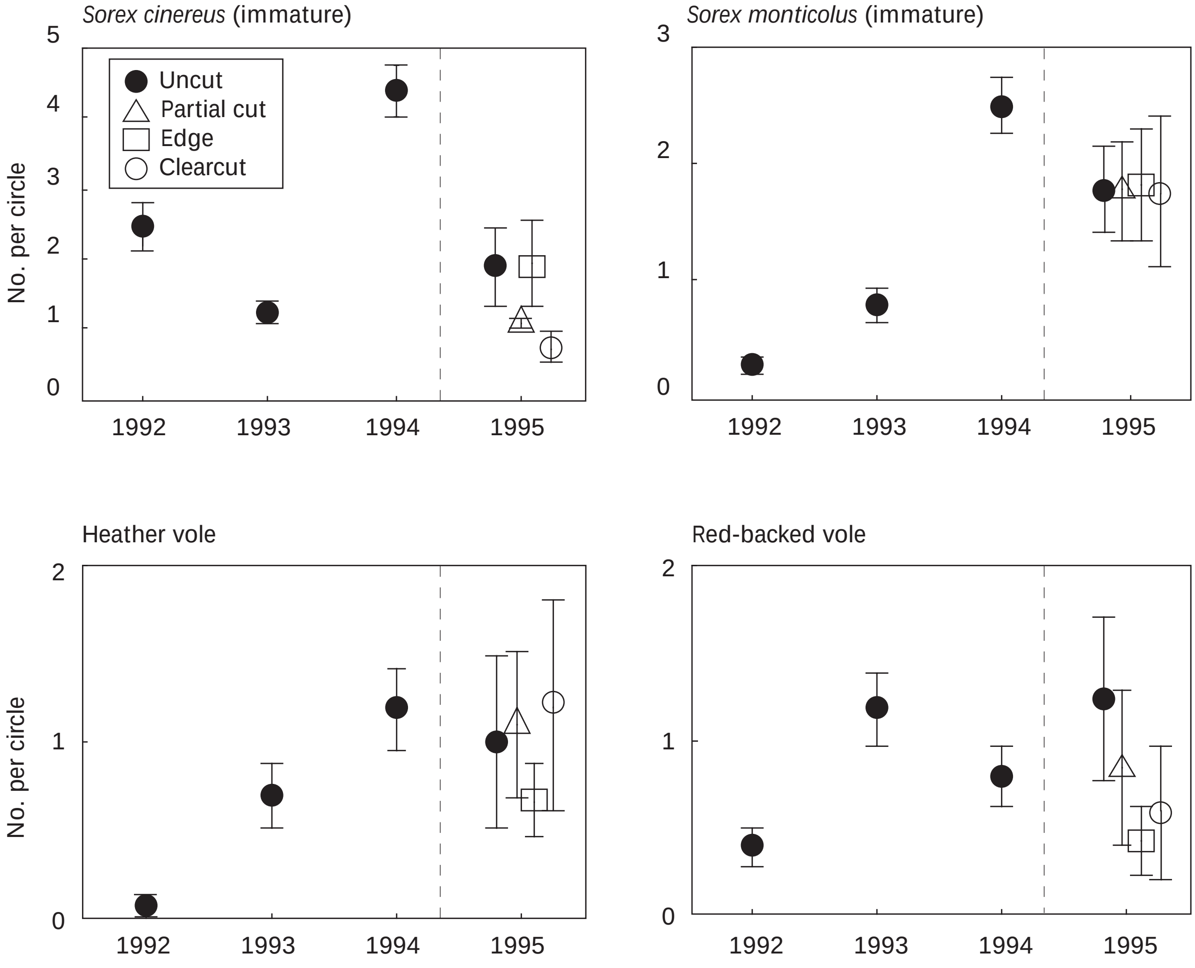

FIGURE 2 *Abundance of shrews and other small mammals in different harvest types at Sicamous Creek.*

- Some species, such as heather voles and *Sorex monticolus*, showed similar patterns of abundance in the different blocks in 1994 and 1995, indicating either that differences in habitat quality between blocks persisted after harvesting, or that the populations themselves had not yet changed.
- Dispersing juveniles caught at the end of summer may be poor indicators of good habitat for the species. Continued sampling, and sampling in other seasons will help reduce these limitations.

A complete data analysis, accounting for other sources of variation, will improve our ability to detect treatment effects despite the variable abundance of shrews.

Habitat Relationships

Habitat relationships of shrews and voles at the cup and circle level (indicated by simple step-wise regression) were either weak or variable between years, and because the abundance of several groups varied greatly from year to year, too few animals were caught in some years to estimate habitat relationships. Habitat relationship results will not be presented until additional post-harvest data are collected, the multi-scale habitat

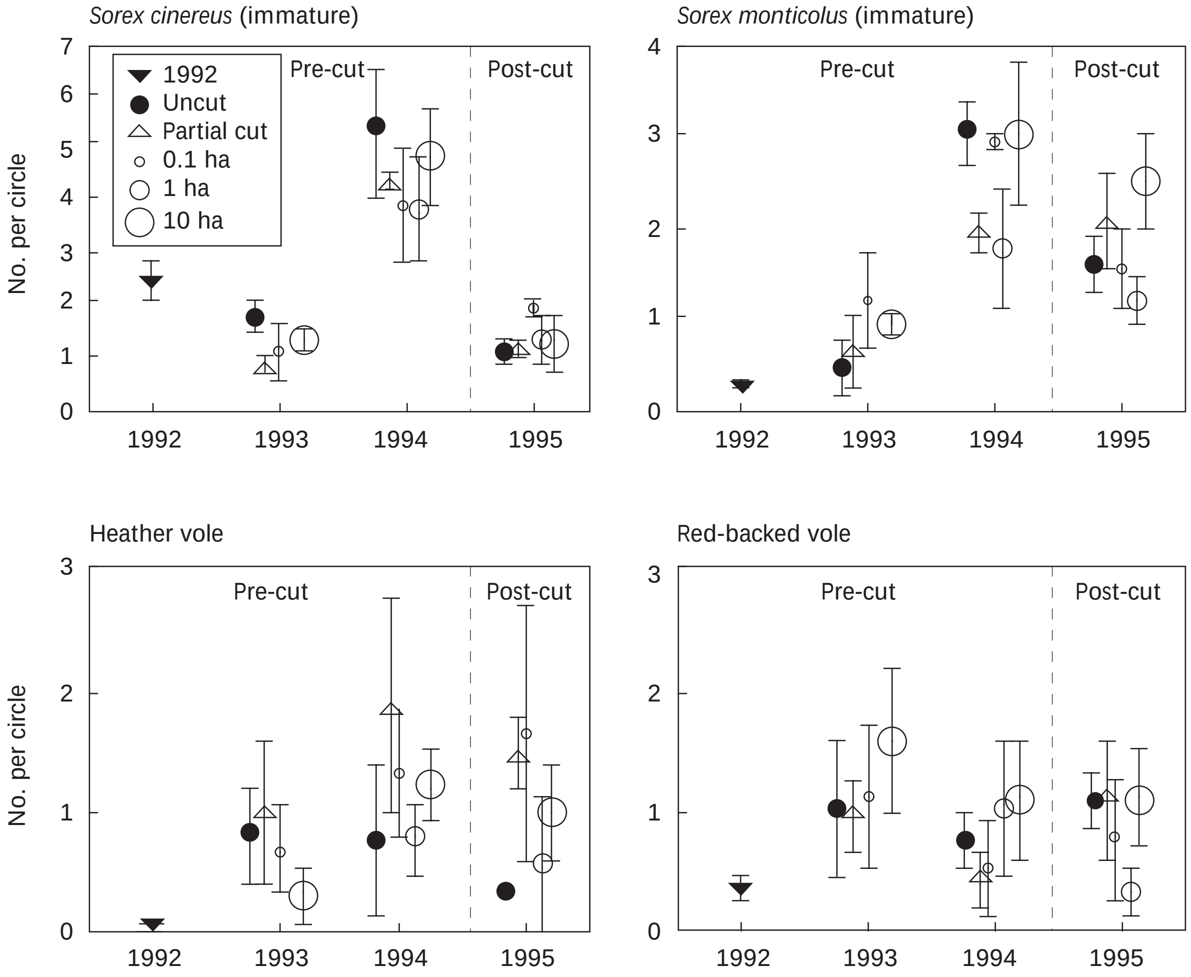

FIGURE 3 *Abundance of shrews and other small mammals in treatment blocks. No harvesting had occurred in the pre-treatment years.*

data are analyzed rigorously and fully, and the habitat relationships validated with independent data from our other sites.

Harvest Effects on Habitat Variables

Most habitat variables measured at the cup and circle level differed significantly between uncut forest, edge, partial-cut, and clear cut sites. Variables for canopy cover, surface moisture, moss cover, shrub cover, and number of live trees, snags, and small trees showed a similar pattern, with uncut sites showing higher values than edge sites, which showed higher values than partial-cut and clear cut sites ($p < 0.05$, anova for each). Forb cover showed a similar pattern, except that clear cuts had more cover than partial cuts. Variables for litter cover and litter depth showed the opposite pattern, with clear cut sites showing higher values than partial-cut sites, which showed higher values than edge and uncut sites. The total volume, average diameter, and profile class composition of coarse woody debris did not differ between the harvest types after harvesting (anova: $p = 0.65$ for total volume, $p = 0.51$ for average diameter; and Wilks' Lambda:

$p = 0.41$ for cwd profile class composition, $n = 45$ circles). The change in most variables represents the response expected with reduced or removed canopy, while the lack of change in coarse woody debris is not surprising because the harvesting occurred on a thick snowpack and included whole-tree removal, which resulted in little disturbance to existing coarse woody debris and little addition of new wood. The increase in litter reflects branches and needles lost from trees during skidding. Along with the other reasons cited above, the constant coarse woody debris levels may be another factor contributing to the lack of an obvious initial response of the small mammals to harvesting.

CONCLUSIONS

Shrews and rodents caught in pitfall traps at Sicamous Creek showed high levels of annual variation in three pre-treatment and one post-treatment years of sampling. Most animals that we collected in August sampling were immatures, which were probably dispersing and may not reflect habitat use by the species. Adults that we sampled after treatment would have been present on the site before harvesting, while immature animals would have been born under the snow before harvesting effects were obvious at ground level. For these reasons, it is premature to discuss treatment effects on shrews and other small mammals. At least one additional year of sampling, and sampling during other seasons is required. Similarly, habitat models will not be presented until we have completed additional post-treatment sampling, analyzed the relationships rigorously, and validated them with independent data.

Winter logging did not significantly change the volume, diameter, and profile class composition of coarse woody debris at our sampling sites. Because coarse woody debris is probably an important habitat feature for shrews and other small mammals, the harvesting may therefore not represent a large habitat change for these groups. However, canopy removal did change other forest floor habitat characteristics that may be important to shrews.

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Small Carnivores in Winter at Sicamous Creek: A Pilot Study

DAVE HUGGARD AND WALT KLENNER

ABSTRACT

A pilot study using winter animal track-intercept transects showed that marten were well distributed across the Sicamous Creek site before logging. Limited post-harvest surveys indicated that small carnivores were not using clearcuts in winter, but were using partial cuts and uncut forest at similar levels. The abundance of small weasels on the site declined with elevation. A precision analysis of the initial results indicates that a moderate surveying effort with animal track transects can yield estimates of track abundance with a standard error that is 20% of the mean. This is adequate to detect substantial treatment or edge effects.

INTRODUCTION

Small carnivore populations require an adequate abundance of their prey and special habitat features, especially snags or large coarse woody debris for denning and resting, and structures on the ground to allow subnivean access in winter (Corn and Raphael 1992). Marten are often considered an "old-growth species" (Buskirk and Powell 1994). Habitat features used by marten, and the small mammals they feed on, are often reduced or eliminated by clearcut logging (Buskirk and Ruggiero 1994). However, alternative harvesting systems, such as patch cuts or partial cuts, may benefit these animals (Lofroth 1993; Thompson and Harestad 1994). Harvesting effects on smaller weasels are not well known.

At Sicamous Creek, we are studying the effects of harvesting systems on small mammal populations and on habitat features used by carnivores. We are also developing a winter tracking program that will directly determine how small carnivores respond to harvesting systems, and how this response is related to measured changes in habitat features and prey populations. This work is important for forest management because of the role top carnivores play in ecosystems and the direct value of the species, particularly marten, as an economic and aboriginal cultural resource. To date, our study has documented the pre-treatment distribution of carnivores at the Sicamous Creek site, and we have conducted pilot post-harvesting surveys to help design future sampling.

We used surveyed transects, 250 m apart across the study site, as animal track-intercept transects in the winter. For any tracks crossing the transect, we recorded the exact location on the transect, the species of animal, and the behaviour indicated by the tracks (mainly, whether the animal appeared to be using the area, indicated by meandering or variable tracks, or whether it was travelling through, indicated by direct, regular tracks). For multiple track crossings of the same species in a short distance, we followed the tracks to determine whether one or more animals had made the tracks. Although more widely separated tracks do not necessarily represent more than one animal, they do represent more spatially and temporally separated records of an animal's use of an area and therefore offer independent indications of their use of habitat.

Time since last snowfall was recorded for each transect, and track abundance in each treatment block, harvesting system, or distance-from-edge category was indexed as the number of track crossings per km•day (kilometres of transect multiplied by the number of days or fractions of days since last snowfall). In the pilot study, most transects were surveyed one day after snowfall, so that 1 km•day is equivalent to 1 km of transect. Mean indices and standard errors were calculated using formulas for ratio data (Krebs 1989).

We also plan to follow individual marten tracks in the snow, recording the animal's behaviour and mapping its location relative to treatment blocks and cutblocks. We will establish simple habitat plots every 50 m and more complete 5.65 m radius plots every 200 m. Random habitat plots will also be established, and used to determine the selectivity of marten for different winter habitat features, particularly subnivean access points (Corn and Raphael 1992).

Analysis of Sampling Design

The results from the initial post-harvest transects were used to estimate the precision expected under different sampling designs. Two sampling variables were explored: (1) the number of replicate transects, and (2) the length of each replicate. Two sources of variation affect the number of tracks on a length of transect.

- Site-to-site variability in abundance of tracks: We used the observed mean number of tracks per km•day and the standard deviation (normal distribution) from the pilot transects to generate a distribution of the mean number of tracks expected on a given replicate segment.
- Sampling error in the number of tracks actually recorded along a certain segment: We modelled this error using a Poisson distribution based on the mean abundance expected on the segment. From the replicate segments, we calculated the standard error of the number of tracks per km•day. This process was repeated 100 times for each combination of segment length and number of replicate segments, to provide an average expected standard error for that sampling design.

The results of the simulations were used to graph the expected standard error of the mean number of tracks per km•day as a function of the number and length of replicate transect segments, which can be used to evaluate the expected precision of a proposed sampling design. We also plotted graphs of the expected standard error as a function of the total km•days of transects (length multiplied by number of replicates) for the different segment lengths. These figures can be used to determine the segment length that produces the smallest standard errors for a given distance of sampling.

RESULTS AND DISCUSSION

Pre-harvest track transects showed that marten were well distributed and abundant at the Sicamous Creek site in winter. Tracks of small weasels were less abundant and tended to be more common in the lower part of the study area, which is nearer to the large marshes. The same pattern was found in limited post-harvesting transects (Figure 1). Squirrel tracks were uncommon and patchy. Populations of voles were high throughout the site in the summers preceding the winter tracking, which may explain the abundance of marten.

Marten tracks were not observed in the 10- or 1-ha cutblocks after harvesting, and weasels very rarely used these clearcuts. The 0.1-ha patch

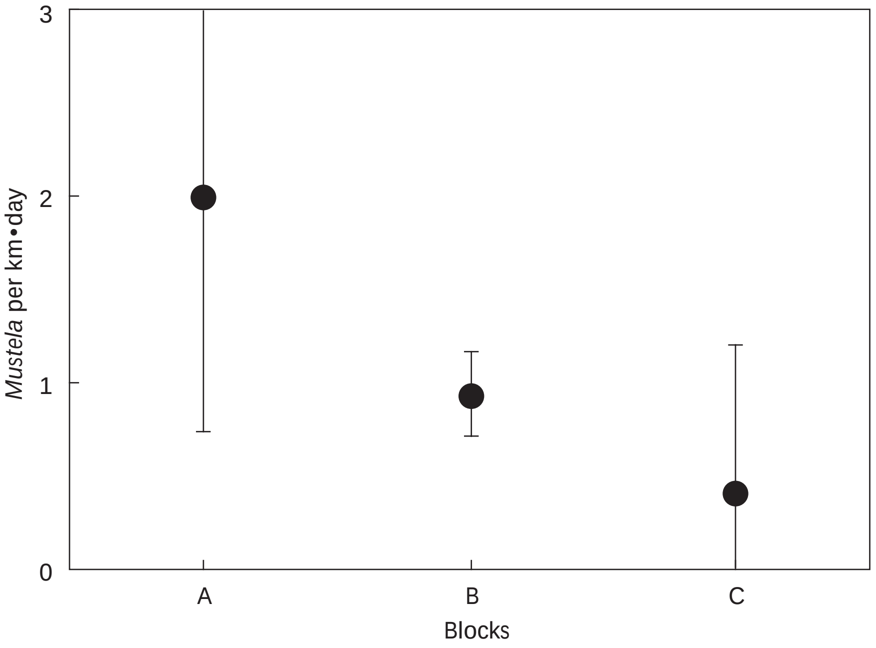

FIGURE 1 *Abundance of tracks of smaller weasels in lower (A), middle (B), and upper (C) study blocks. Error bars are two SE.*

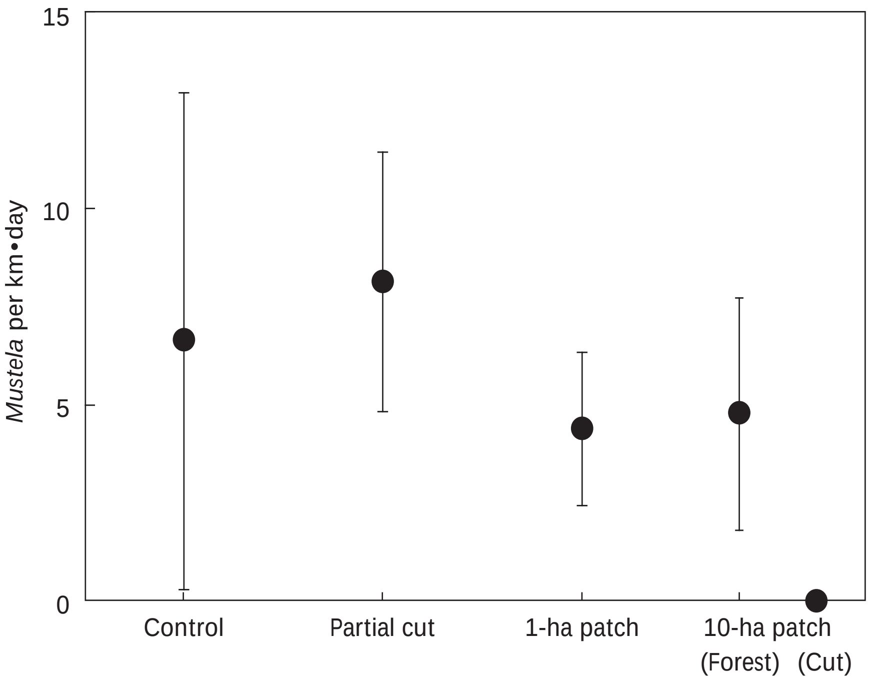

FIGURE 2 *Abundance of marten tracks in four treatment types (forested and cut areas of 10-ha blocks plotted separately). Error bars are two SE.*

cuts were not sampled adequately to assess their effects on carnivore use. The partial cuts and uncut forest adjacent to cutblocks had track abundances similar to the control areas, though our sampling was limited and there was considerable variability between the replicate control blocks (Figure 2).

Sampling Design

Figure 3 shows the relationship between the expected standard error (expressed as a percentage of the mean) for different combinations of segment lengths and numbers of replicate segments (based on marten tracks in the pilot transects). For example, with 10 segments of 0.4 km•day each, the standard error is expected to be about 25% of the mean. This is a fairly low sampling effort, indicating that reasonably precise estimates of track abundance should be possible in this study. Note that four segments of 1.0 km•day represents the same total length of transects, but should produce wider standard errors, almost 40% of the mean.

If travel distance between transects (overhead cost, Krebs 1989) is low or negligible, then the simulations suggest that more numerous, shorter transects produce smaller standard errors for a given total transect length, with an optimum of 200–400 m•day segments. However, when travel distances of 500 m between segments are used, then moderate numbers of moderate-length segments are more efficient, with an optimum of 800–2000 m•day segments. These longer travel distances between replicates allow the replicates to be further apart, which is desirable, and so replicate segments of 0.8–2 km•day seem optimal for future sampling.

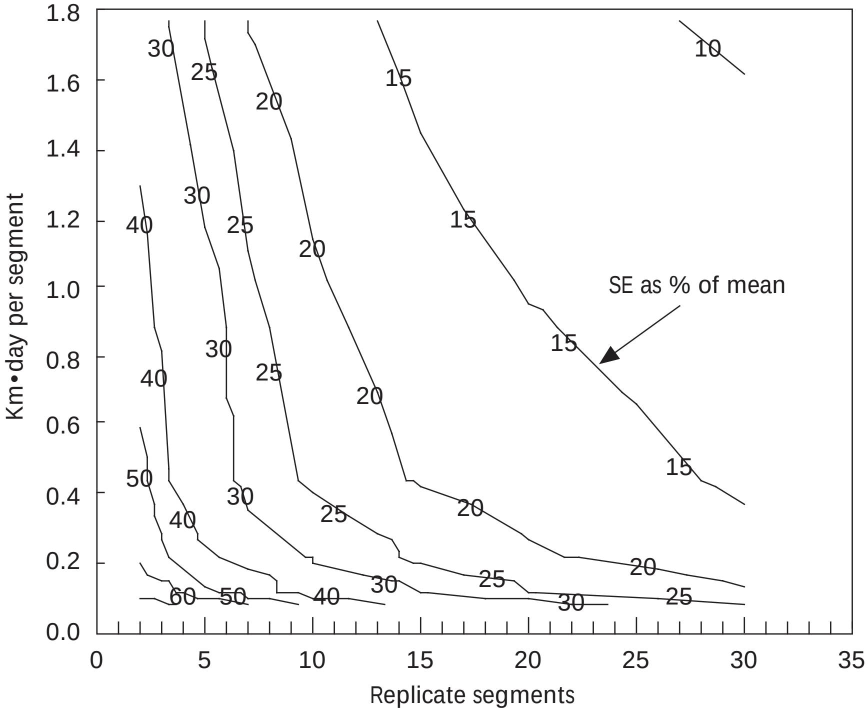

FIGURE 3 *Expected standard error of track abundance (expressed as % of mean), as a function of the length and number of replicate transect segments.*

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Spruce Grouse Distribution and Habitat Use at Sicamous Creek

DAVE HUGGARD

ABSTRACT

I used transect surveys and sign plots to measure the abundance, distribution, and habitat use of spruce grouse, and how these are affected by harvesting at the Sicamous Creek Silvicultural Systems research site. Spruce grouse at the Sicamous Creek site are patchily distributed. Transect surveys and grouse sign plots show that grouse are closely associated with knolls and ridges, and forest near marshes. Their association with marshes is predicted by a habitat model developed from the sign plots. The model predicts increased probability of grouse at sites with an increasing abundance of spruce, increasing canopy cover, decreasing tree diameter, and increasing density of "short trees" (trees which are shorter than the height expected from the height-to-diameter relationship). The reason for the association with knolls and ridges is unknown. Sign plots indicated that spruce grouse did not use recent clearcuts, and were 33% less abundant in partial cuts than in uncut forest. Grouse used forest 5 m from a cut edge much less than they used interior forest, but no obvious negative edge effects were found at distances greater than 5 m. All of these results are preliminary, as they are based on less than one year of post-harvesting surveys and plots.

INTRODUCTION

Spruce grouse in the southern interior of British Columbia are found only in high-elevation coniferous forests. In winter, spruce grouse feed exclusively on the foliage of conifers (Boag and Schroeder 1992), especially spruce, and roost in conifers throughout the year. Because of this close association with conifers, recent clearcuts do not provide suitable habitat for spruce grouse. However, the suitability of alternative silvicultural systems for these birds is not known.

Within coniferous forests, spruce grouse show preferences for certain habitat features, including areas with dense trees, high canopy cover, and dense understorey vegetation (Szuba and Bendell 1983). However, even in apparently uniform forests, spruce grouse distributions are patchy which suggests that additional habitat features may be important. Unknown habitat requirements can lead to two problems in managing forests to sustain spruce grouse populations.

1. Silvicultural operations alter many aspects of forest structure, some of which are probably important for spruce grouse. Unless we know the habitat features required by grouse, and how these are affected by different silvicultural activities, forest management may unknowingly eliminate suitable habitat.
2. If we do not know the habitat features required by grouse, their patchy distribution creates a large source of unexplained variance. This variance makes it difficult to measure the effects of silvicultural treatments or responses to landscape features such as edges.

An objective of this study is to develop reliable habitat models for spruce grouse. Such models will allow us to predict the effects of habitat change due to forest management, and to explain part of the variation in grouse abundance which will enable a more precise estimate of treatment and landscape effects.

The effects of forest fragmentation and other landscape-level changes on grouse are poorly known. Fritz (1979) showed that spruce grouse disappear from smaller isolated patches of forest. The range of spruce grouse has retracted in eastern North America with forest loss and fragmentation (Boag and Schroeder 1992). However, these results are not very applicable in western forests, where harvesting does not generally produce isolated patches of forest. In Scandinavia, intensive forest management has caused changes in the population density (Klaus 1991), breeding biology (Storch 1991), and home ranges (Ims et al. 1993) of grouse in old forests. One species, the capercaillie (*Tetrao urogallus*), is declining significantly because of forest fragmentation (Wegge et al. 1992). The most noticeable initial effect of dispersed cutblocks on landscapes in British Columbia is the creation of "high-contrast" edges and the loss of interior forest, particularly when many small patch cuts are added to a contiguous forest. The effect of these edges on spruce grouse is the main landscape "fragmentation" effect being addressed in this study.

METHODS

Three methods are used to assess spruce grouse distribution, habitat use, and response to edges:

1. spring transect surveys, for the overall distribution of birds, especially breeding males;
2. plots for grouse sign and habitat features; and
3. transects for grouse sign parallel to cut edges.

Spring surveys and sign plots were conducted in 1994 (pre-harvest) and 1995 (post-harvest). Edge transects were conducted after harvesting, in 1995, only. This work is continuing in 1996.

Spring Transect Surveys

A grid of transects covering the study site at 250-m intervals (north/south and east/west lines) were surveyed at least twice in the spring before snowmelt and twice immediately after snowmelt. Surveys began in the morning at first light. The location of any birds seen or heard (usually displaying males), tracks on the snow when present, and droppings were all recorded. Repeated surveys by different observers almost always find birds in the same locations as the first surveys, suggesting that the surveys provide a reasonably complete picture of grouse distribution on the site at this time of year.

Plots For Grouse Sign and Habitat Attributes

I established 5.65 m radius plots (0.01 ha) in which I recorded the following variables: plot location, distance to cut edge, distance to marsh edge, topography type, slope, aspect, canopy cover, shrub cover and main species, forb cover and main species, ground cover type, site moisture, number and species of small trees (< 7.5 cm diameter), and diameter at breast height, height, and decay class (following Thomas [1979]) of each tree greater than 7.5-cm dbh. I also counted the number of grouse droppings on the plot, though the analysis was based primarily on presence versus absence of droppings. Plot recording took place in the spring just after snowmelt and data represent over winter use by the grouse.

Two sampling schemes were used to locate plots:

1. Systematic: plots were evaluated every 200 m along surveyed transects that were 250 m apart. Different locations were sampled in different years by choosing different starting points.
2. Grouse-centred plots: 13 plots were evaluated in areas where spruce grouse were seen repeatedly. One was a site where grouse were frequently observed (centre), four were 25 m away from this centre (in the cardinal compass directions), four were 50 m from the centre, and four were 100 m away. The grouse-centred plots indicated habitat use in areas where I know that grouse are present.

Transects For Grouse Sign Parallel to Cut Edges

I established 2 m wide strip transects in the forest parallel to cutblock edges at the following distances from edges of the different-sized cutblocks:

- 0.1-ha blocks: 5 and 10 m into the forest;
- 1.0-ha blocks: 5, 10, 20, and 50 m into the forest; and
- 10-ha blocks: 5, 10, 20, 50, and 100 m into the forest.

For the 0.1 and 1.0-ha patch-cut arrays, these transects were established in the leave strips between blocks. The furthest distance from the edge is near the middle of the leave strip. On each transect I recorded the number of grouse droppings and number of groups of droppings within 1 m of the transect line. These transects were also conducted just after snowmelt to represent over winter use.

RESULTS AND DISCUSSION

Distribution of Grouse

Spruce grouse are the only grouse species observed at the Sicamous site. Pre-harvest surveys in 1994 showed that spruce grouse were patchily distributed (Figure 1a). The same distribution was seen in 1995 after the silvicultural treatments took place, although extended transect lines revealed a further cluster of spruce grouse above the study area to the south (Figure 1b). Initial survey results from 1996 indicate a similar pattern. Such a heterogeneous distribution could make identification of treatment effects difficult, particularly if I could not identify factors explaining this patchiness.

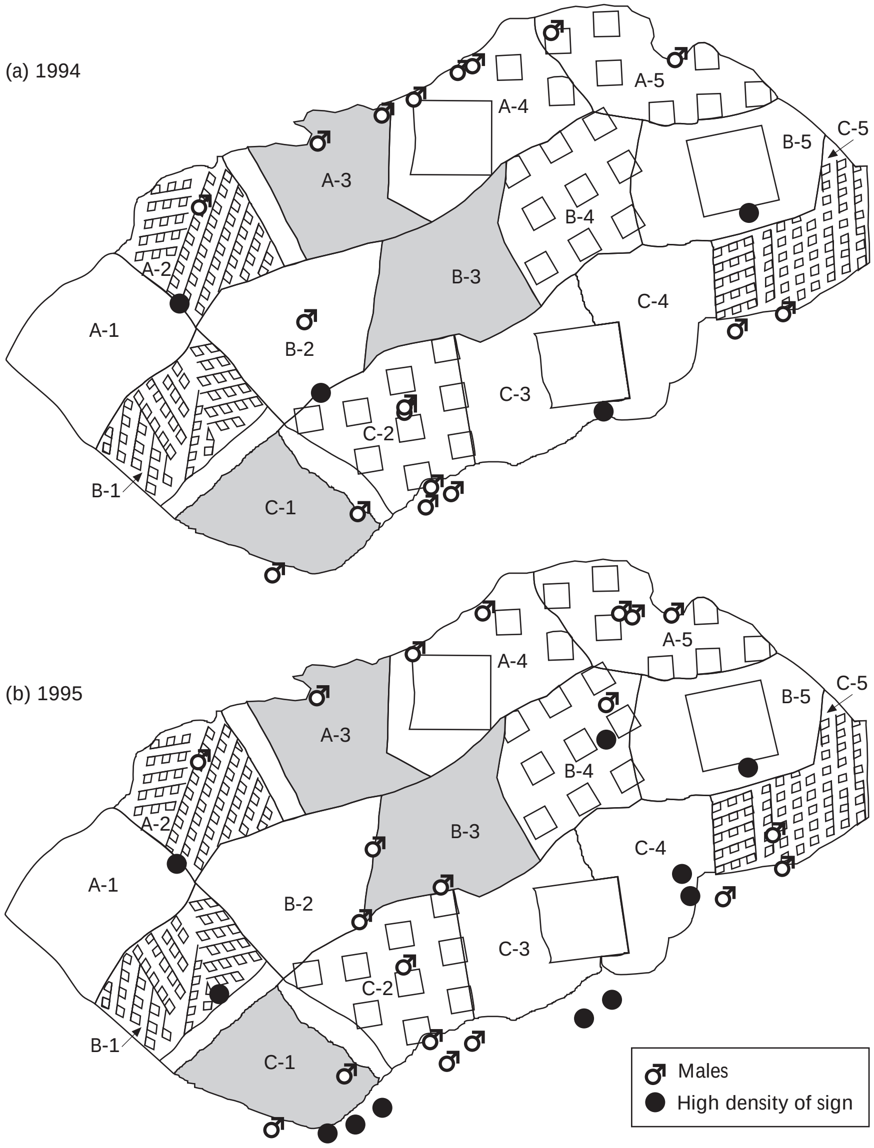

FIGURE 1 *Locations of male grouse and areas with high densities of sign on transect surveys.*

In some cases, spruce grouse were seen in exactly the same location in the pre- and post-treatment years. Although I have not banded any birds, these undoubtedly represent the same individuals. Spruce grouse are known to live several years and show high site fidelity (Boag and Schroeder 1992). Therefore, the results presented here, which are based on a single year after harvest, should not be overinterpreted.

Habitat Use From Plots

Systematically located plots only contained grouse droppings 13.4% of the time in 1994 and 10.5% of the time in 1995. The low levels of grouse sign in these plots weakens the analysis of habitat use by grouse; analysis of the data from systematic plots is therefore not included here. However, these plots do provide information about the overall habitat characteristics available to grouse at the study site. They also confirm the completeness of the transect surveys because high densities of droppings were only found in systematic plots where grouse had already been detected on transects.

Grouse-centred plots had higher overall levels of grouse occurrence (46.6% in 1994 and 38.7% in 1995) and were a more efficient way to indicate habitat preferences.

Topography Differences

Percentage occurrence of grouse sign in the grouse-centred plots was far higher on knolls (small hills that are noticeably higher than the surroundings in at least a 270° arc) than in the smooth or rolling slopes or gullies which are the most common topography type (Scheffé test $p < 0.001$; Figure 2). Ridges or benches had an intermediate occurrence, lower than knolls, but higher than other topography types. Topography type is the most obvious factor to differentiate between used and unused sites.

Regression Habitat Model of Grouse Occurrence

I used step-wise regression of grouse presence/absence in plots versus 17 initial habitat variables to compare the used and unused plots in other topography types (i.e., not knolls or ridges/benches). The step-wise

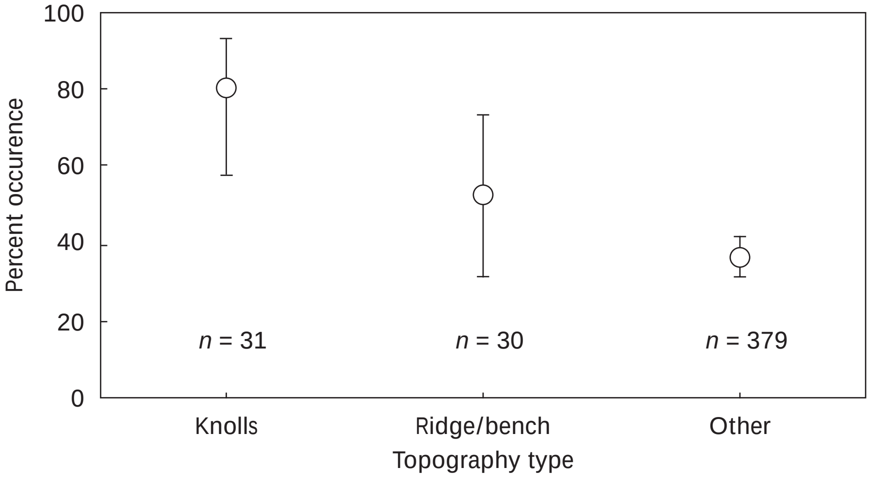

FIGURE 2 *Occurrence of grouse sign by topography type in grouse-centred plots.*

regression produced equations that contained the same variables and similar coefficients for the two study years. [A multiple logistic regression would be more appropriate here, but has not yet been conducted.] I interpret the dependent variable in these linear regressions as the predicted probability of a grouse being present on the plot, given its habitat characteristics, although the linear equation could theoretically produce nonsensical probability values greater than one or less than zero. The best-fit equation for the two years combined was:

$$P(\text{grouse}) = 0.437 - 0.020 \cdot LIVEDBH + 0.085 \cdot \log_e(CANOPY) + 0.121 \cdot \log_e(LIVESPRUCE) + 0.077 \cdot \log_e(SHORT),$$

where: P(grouse) = the probability of grouse sign on the plot;
LIVEDBH = the average dbh of live trees (in cm);
CANOPY = the percent canopy cover;
LIVESPRUCE = the number of live spruce trees on the 0.01-ha plot; and
SHORT = the number of "short" trees on the 0.01-ha plot.

In this multivariate equation, $r^2 = 0.166$, $r^2_{\text{adj.}} = 0.158$, and $p < 0.0001$. The probability of grouse sign increases with canopy cover, the number of live spruce and the number of short trees, and decreases with the average size of the live trees. These variables probably indicate the dependence of grouse on spruce, and their need for cover in the relatively open forests of the Engelmann Spruce – Subalpine Fir (ESSF) biogeoclimatic zone. The preference for smaller and shorter trees may be because they are "stressed," producing forage with lower levels of secondary chemicals toxic to grouse (Mueller 1993). Other habitat variables were unimportant in influencing the probability that grouse occurred on these plots.

Does Observed Habitat Preference Explain the Patchy Distribution of Grouse?

I used the information from these plots to ascertain whether the patchy distribution of grouse on the site was caused by their selection of certain habitat features. Alternative possibilities were that grouse aggregate independently of habitat (e.g., loose flocking for predator avoidance or loose "lekking" for breeding), or that unmeasured features were responsible for the patchiness (e.g., the location of predators).

Approximately 40% of the known grouse, or areas with high densities of droppings, are on knolls or ridges. Of the remaining 60% of birds, or high-density sites found on the transect surveys, almost all are in forest very near to marshes. A strong relationship existed between distance from marsh and the probability of grouse presence on a plot (Figure 3). The polynomial regression curve suggests greatest probabilities of grouse presence 15–30 m from a marsh, with lower numbers further away, and also lower numbers immediately adjacent to the marsh edge.

I then used the multiple regression model above to determine whether this observed distribution is caused by grouse responding to measured habitat changes near the marsh, or is a direct response to the marsh itself (perhaps reflecting different growing conditions for forage trees, distributions of predators, winter microclimatic conditions, etc.). For this analysis I calculated the residual probability of grouse presence after the habitat

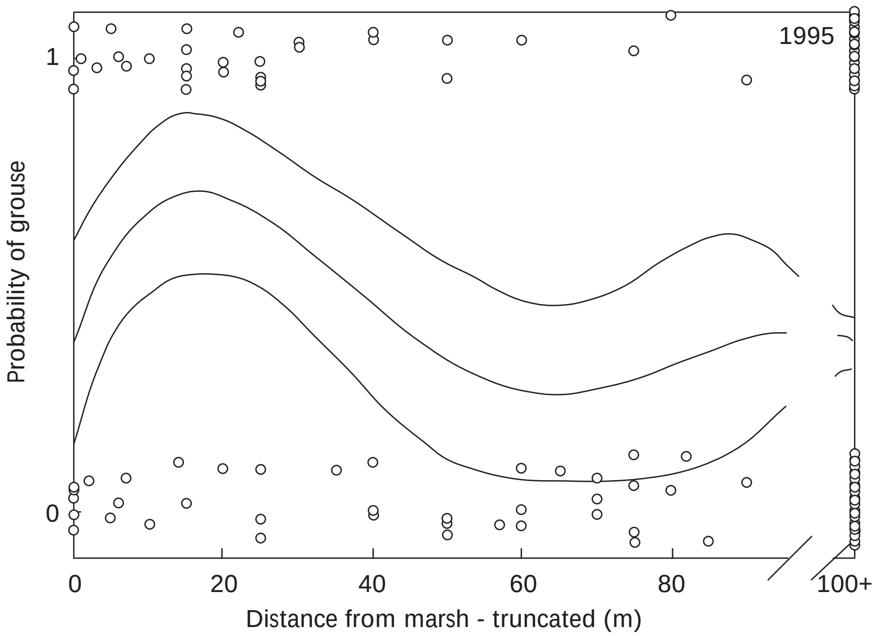

FIGURE 3 *Relationship between the probability of grouse occurring on a plot and distance to a marsh. Each point is a plot, with (= 1) or without (= 0) grouse (some scatter added to separate points).*

relationships had been accounted for (the observed presence/absence minus the probability of presence from the equation above). If habitat change alone were responsible for the observed pattern, I would expect the residuals to show no relationship with distance from marsh. If the marsh itself influenced the distribution, I would expect a relationship to remain after habitat differences had been accounted for. The residuals showed almost no relationship with distance from marsh (Figure 4), indicating that habitat differences near the marsh were mainly responsible for the higher occurrence of grouse in these areas.

In summary, the patchy distribution of grouse appears related to birds responding to either topographic features (≈40% of the birds) or to habitat features associated with marsh edges (≈55% of the birds).

Initial Treatment Effects

Grouse were never detected on transect surveys or plots in clearcuts after harvesting (though they were in some of these areas before harvesting). Occurrence of grouse on grouse-centred plots after harvesting was 33.3% lower in partially cut blocks than in uncut or the leave strips in patch-cut arrays, though the difference is not statistically significant ($\chi^2 = 1.62$, 1 *df*, $p = 0.13$). This is (by chance) precisely the reduction expected if the only effect of removing 33% of trees by partial cutting was to remove 33% of the habitat, such as foraging-trees, used by grouse.

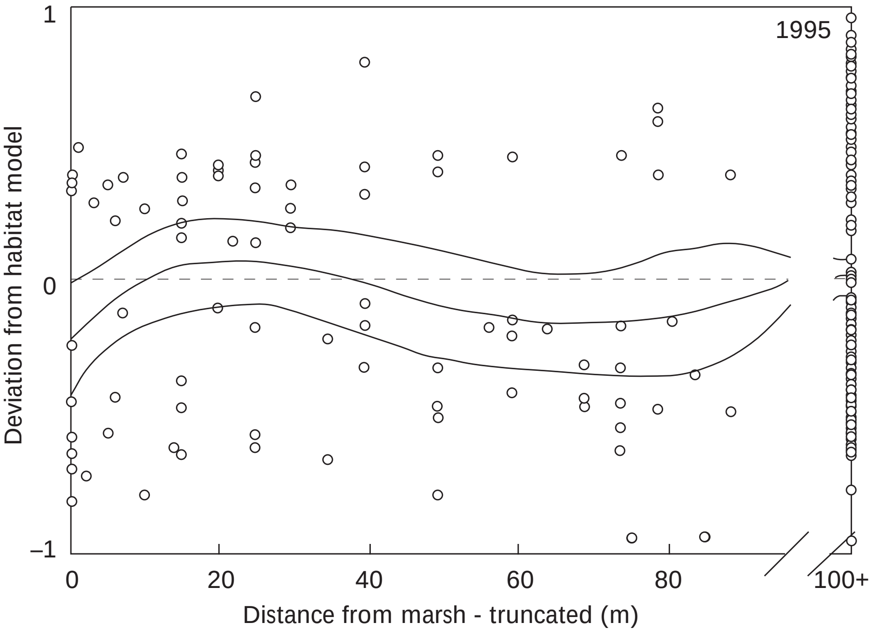

FIGURE 4 *Relationship between the probability of grouse occurring on a plot and distance to a marsh after the habitat differences have been accounted for.*

Edge Effects

Transects for grouse sign parallel to 0.1-, 1-, and 10-ha cutblocks showed far lower numbers of droppings 5 m from the edge compared to 10 m or further (Figure 5). In the 1-ha blocks, numbers appeared to increase slightly through to the 50-m transect, while the 10-ha block, with fewer replicates, showed no consistent pattern after 5 m. In all cases, the mean values have wide confidence intervals because the droppings were aggregated on some transects. However, using median numbers and percentage occurrence showed similar patterns. Because the 5-m effect was seen on all sides of cutblocks and ground conditions were similar on all transects, I think this reduction in numbers reflects that grouse use the areas immediately adjacent to cutblocks less, rather than a bias in my ability to detect grouse droppings near edges.

CONCLUSIONS

My initial work on spruce grouse and this preliminary analysis suggest that grouse have certain distinct habitat preferences, which create a patchy distribution in the ESSF study area. Grouse are associated with knolls or ridges, and features associated with forest near marshes, including higher

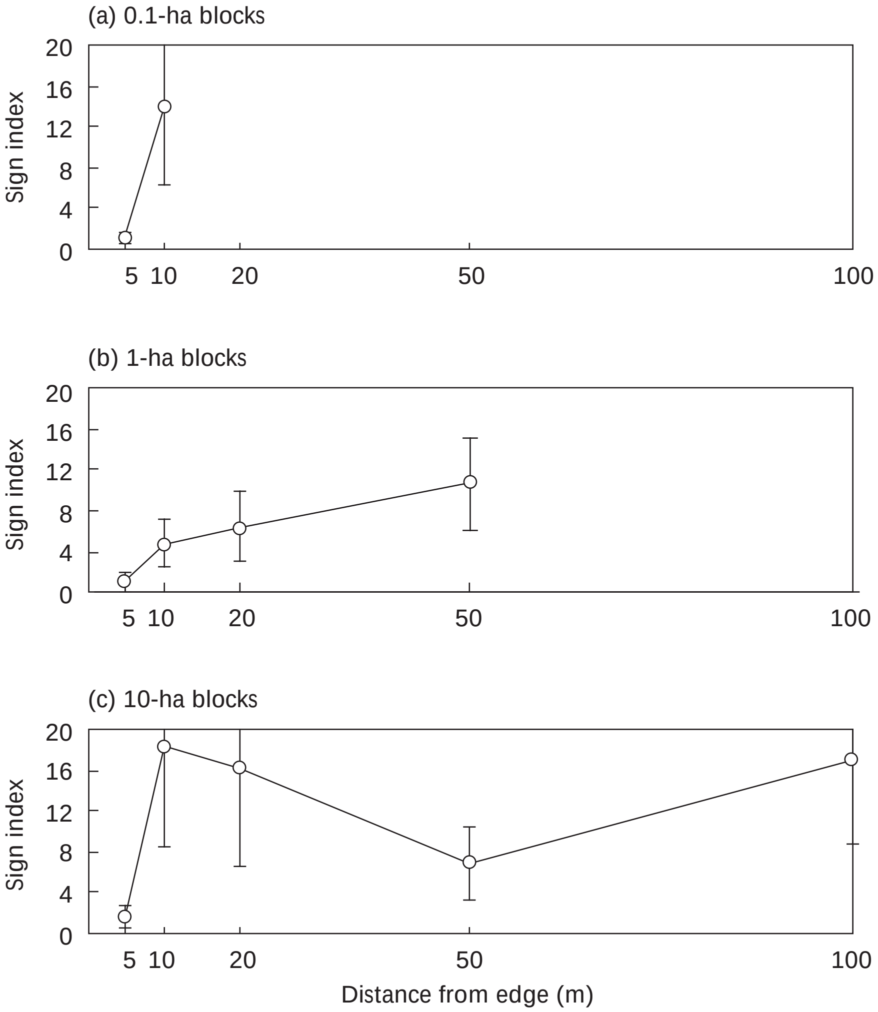

FIGURE 5 *Edge effects on the abundance of grouse sign in cutblocks of three sizes.*

densities of spruce, more canopy cover, small diameter trees, and trees that are shorter than expected given their diameter. If confirmed by subsequent work, some special management of these areas, which form a small percentage of the forested area, may be warranted to maintain healthy grouse populations. For the ongoing spruce grouse research, this knowledge will help us to account for spatial heterogeneity which will enable a more precise measurement of the effects of silvicultural treatments and the features they produce, such as edges.

The initial edge transects showed that spruce grouse use the forest 5 m from a cut edge far less than further into the forest. No obvious reduction in use occurs at 10 m or more. Although small, this edge effect (if confirmed) could be important in the arrays of 0.1-ha cuts, where 30% of the remaining forest is within 5 m of an edge when 33% has been cut. This would mean that over 60% of the forest would be negatively affected by the first rotation with this harvest system.

Most importantly, all of these results represent the first year of post-treatment data—the sign plots and edge transects reflect use by grouse in the winter when logging occurred. Individual birds seem to be in the same place they were before logging, which suggests that most responses to logging may not have occurred yet. Also, the decline in grouse occurrence in the partial cuts may be an initial response to disturbance by logging activities, rather than indicating decreased habitat quality.

We will continue spring surveys in 1996 and 1997 to monitor longer-term changes in spruce grouse abundance and distribution following logging. Sign plots will be conducted where new grouse are found, and additional edge transects will be evaluated. An additional set of sign surveys will focus on knolls in different treatments and edge positions, because the higher occurrence of grouse sign in these sites will allow a more precise estimate of any treatment or edge effects. Radio-telemetry is planned to provide a second index of habitat use and to estimate home ranges of the spruce grouse.

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Three-toed Woodpecker Nesting and Foraging at Sicamous Creek

WALT KLENNER AND DAVE HUGGARD

ABSTRACT

We used transect surveys, nest searches, foraging observations, and radio-telemetry to study the nesting and foraging habitat of resident cavity-nesting birds at Sicamous Creek. Three-toed woodpeckers are the only common woodpecker species, with six to seven pairs using the site in 1994 (before harvesting) and 1995 (after harvesting). Nests are usually in subalpine fir of 30–35-cm dbh that have recently died, and are closely associated with forest edges. Nesting sites appear abundant, but may be seriously reduced by snag removal adjacent to cutblocks. Males and females show similar foraging preferences in summer and winter, choosing recently dead subalpine fir snags, especially the largest stems. Preferred foraging habitat will therefore also be substantially reduced by snag removal, although three-toed woodpeckers may be able to switch to live spruce for foraging.

INTRODUCTION

Woodpeckers play a key role in forest ecosystems, providing nest holes for other cavity-nesting birds, and roosts for a wide range of birds and mammals (e.g., Bull et al. 1986). Woodpecker foraging activity also creates feeding opportunities for several other species (Miller and Nero 1983) and may limit the abundance of several forest "pest" insects (Holmes 1990). These relationships between woodpeckers and many other species have led to woodpeckers being called "keystone species" (Daily et al. 1993), whose loss would seriously disrupt forest ecosystems.

Woodpeckers and other cavity-nesting birds are among the wildlife species that are likely to be negatively affected by traditional forest management. Woodpeckers rely on several forest attributes that will probably be reduced in managed forests, including snags, coarse woody debris, diseased trees, and trees with insect infestations. Alternative silvicultural systems, such as partial cuts or patch cuts, may provide better habitat for woodpeckers than traditional clearcuts, but the increased edge and associated removal of snags may also eliminate important habitat features in these systems.

Experience in other areas with a longer history of forest use than British Columbia shows that woodpecker distribution and abundance is reduced as a consequence of forest management. In Sweden's intensively managed forests, one woodpecker species is almost extirpated (white-backed woodpecker; Aulen 1988) and several other species are declining seriously (Angelstam and Mikusinski 1994). The same situation is reported from Finland (Tiainen 1985). In the southeastern United States, extensive logging of older forests has caused the extinction of one species (ivory-billed woodpecker) and has endangered another (red-cockaded woodpecker; Walters 1991), leading to expensive restoration projects. In the Pacific Northwest, special management areas are being established for pileated woodpeckers, which are considered vulnerable in intensively managed landscapes (Bull and Holthausen 1993). Forest management based on a knowledge of the habitat and landscape requirements of woodpeckers will hopefully avoid a similar situation in British Columbia.

Numerous studies have described the characteristics of nesting trees used by woodpeckers. However, few have studied high-elevation forests. Furthermore, there is great variability among the areas that have been studied (Bull et al. 1986; Goggans et al. 1988), which suggests that local studies are needed to guide management in the British Columbia Interior. More importantly, several studies have found that nesting sites do not limit the size of woodpecker populations, and have emphasized the importance of foraging habitat (Walankiewicz 1991; Welsh and Capen 1992). However, the habitat used by cavity nesters for foraging is not well documented, particularly in high-elevation forests. Previous studies have focused on summer foraging, even though winter is a critical time for survival of cavity nesters in other areas (Nilsson 1987). Finally, landscape-level changes caused by harvesting, such as the creation of edge and thin leave strips, and interspersion of young- and old-forest stands, have unknown effects on the nesting and foraging habitat of cavity nesters.

This study addresses three gaps in our knowledge of cavity-nester ecology:

1. nesting habitat in high-elevation forests;
2. foraging habitat in summer and winter; and,
3. effects of alternative harvesting systems with the associated snag removal and landscape-level changes.

The study examines all resident cavity nesters, but this summary presents results only for three-toed woodpeckers, the common breeding woodpecker at the Sicamous Creek site.

METHODS

Transect Surveys

We conducted transect surveys to index the abundance and distribution of woodpeckers in the study area, to find birds for foraging observations, and to locate nests. Transects 125 m apart across the study area were surveyed throughout the year in the mornings. Observers covered 100 m in 10 minutes, recording the location and distance where each woodpecker was detected, and finding the bird to identify species and sex. We then observed the bird's foraging activity or attempted to follow it to its nest. Transect surveys and foraging observations were conducted in winter 1993–94, summer 1994 (pre-harvest), summer 1995, and winter 1995–96 (post-harvest).

Foraging Observations

The foraging activity of woodpeckers was observed as they were found during transect surveys, or when they were located using radio-telemetry. Each tree or piece of downed wood used for foraging was an "observation"; sequential observations for an individual bird form a "series." For each observation, we recorded: the location, the species and sex of the bird, the individual if the bird was banded, the distance to nearest harvested edge, the bird's behaviour (following Remsen and Robinson 1990), the position of the bird in the tree and its foraging substrate, the duration of the foraging, and the distance from the previous foraging tree, if known. For each foraging tree, we recorded the species, diameter at breast height (dbh), height, and profile class (following Thomas 1979). When birds were observed foraging on downed wood, we recorded the species, diameter, height above ground, and decay class (following Thomas 1979). We also established a 5.6 m radius "foraging plot" centred on each foraging tree, in which we recorded the species, dbh, and height and profile class of all trees greater than 7.5-cm dbh.

Available Habitat

To compare the characteristics of the foraging trees and plots to those available to the woodpeckers, we established habitat availability plots within 100 m of each foraging tree. This distance encompasses the area that is readily available to a foraging woodpecker, since woodpeckers typically fly at least this far between foraging bouts (observations during this study). A point was chosen within 100 m by following a randomly chosen compass direction for a random distance between 0 and 100 m. This produced a distribution of points that were more heavily weighted towards locations near the foraging tree than further away (an inverse-square distribution). We used two types of availability plots:

1. "Random" plots were centred on any random tree greater than 7.5-cm dbh. To select a random centre-tree, we chose the *n*th tree in a strip transect 20 m wide, where *n* is a random number between 1 and 20 (C.J. Krebs, Zoology Department, University of British Columbia, pers. comm., 1994).
2. "Matched" plots were centred on a tree that was the same species and profile class as the foraging tree, and within 10-cm dbh. The same randomization procedure is used, but only matched trees were counted.

The same information was collected on the 5.6 m radius random and matched plots as was collected on the foraging plots.

The availability of distances from edge was indexed by measuring the distance from 180 systematically located points to the nearest edge on 1994 and 1995 maps of the site. This procedure will be done with a Geographic Information System (GIS) when digital maps of the study site are available.

Radio-telemetry

Radio-telemetry was used to:

1. increase the number of foraging observations and reduce possible biases associated with incidental foraging observations;
2. allow comparison of the foraging habitat of individual birds; and
3. to estimate the spatial use and home ranges of individuals.

Birds were captured at roost or nest trees and fitted with a small, tail-mounted radio-transmitter weighing approximately 1.5 g (Model BD-2G, Holohil Systems, Carp, Ontario). Radio-tagged birds were located at different times each day, and a series of foraging observations where made.

Analysis of Selectivity

We analyzed the selectivity of woodpeckers for tree types and plot characteristics at several scales. We calculated the bird's selective preference for three tree characteristics: tree species, dbh class, and profile class at three scales.

1. Within-plot: the tree used for foraging was compared to the trees available within the surrounding 5.6 m radius plot. The proportion of trees in each plot with each characteristic was calculated. These proportions represent the expected use if woodpeckers were using the trees in the plot randomly. This expected use was then compared to observed use.
2. Local area: the trees used in an area were compared to the available trees in a 100 m radius. These available trees were indicated by the random plots associated with each foraging plot.
3. Overall: the trees used by all woodpeckers were compared to all the other trees in random plots. Ivlev's index was used to indicate selectivity for these categorical data (Krebs 1989), with confidence intervals generated using a bootstrap procedure (Krebs 1989). In the initial analysis presented here, we treated each observation as independent, even though they should be regarded as subsamples of each series of observations. This simplification tends to falsely narrow the confidence intervals if observations within a series are similar to each other (Hejl et al. 1990)

For plot characteristics, such as stem density, mean dbh, basal area, and snag density, we calculated selectivity at two scales.

1. Local area: foraging-plot characteristics were compared to the characteristics of the random and the matched plots (separately) within 100 m.
2. Overall selectivity: foraging-plot characteristics were compared to all the random or matched plots. The plot characteristics are continuous variables, and *t*-tests, paired *t*-tests, or non-parametric comparisons were used, as appropriate.

Woodpecker Numbers, Movements, and Ranges

Three-toed woodpeckers were the only woodpecker species breeding on the site, though hairy woodpeckers, flickers, and sapsuckers visited in the late summer. Transect surveys, nest locations, and radio-telemetry indicated six or seven pairs of three-toed woodpeckers using the site in 1994 (pre-treatment) and in 1995 (post-treatment).

Four woodpeckers were banded and fitted with radio-tags in 1994. Two of these were recaptured, along with three additional birds, and fitted with radio-tags in 1995. Inadequate numbers of independent observations, and several occasions when radio-tagged birds could not be located, prevented us from analyzing home ranges for these birds. However, the "high-use" areas of three birds from each year are shown in Figure 1. The radios revealed extensive movements outside these areas by some birds, including two individuals that often foraged in an Interior Cedar-Hemlock zone forest approximately 2 km away, and two that used areas near large marshes adjacent to the study site.

Nest Trees and Locations

We found five three-toed woodpecker nests in 1994 and seven in 1995 (Figure 1). We suspect that two nests on the site were missed in 1994 and one in 1995. All nests were in subalpine fir trees. Nine of 12 nests were in trees between 30- and 35-cm dbh; two were in 24-cm dbh trees, and one in a 38-cm dbh tree. Ten of 12 trees were profile class 3 (recently dead, with complete bark cover), while two were class 4 (older snags, with partial bark loss). Four of the snags were broken off more than halfway down the bole.

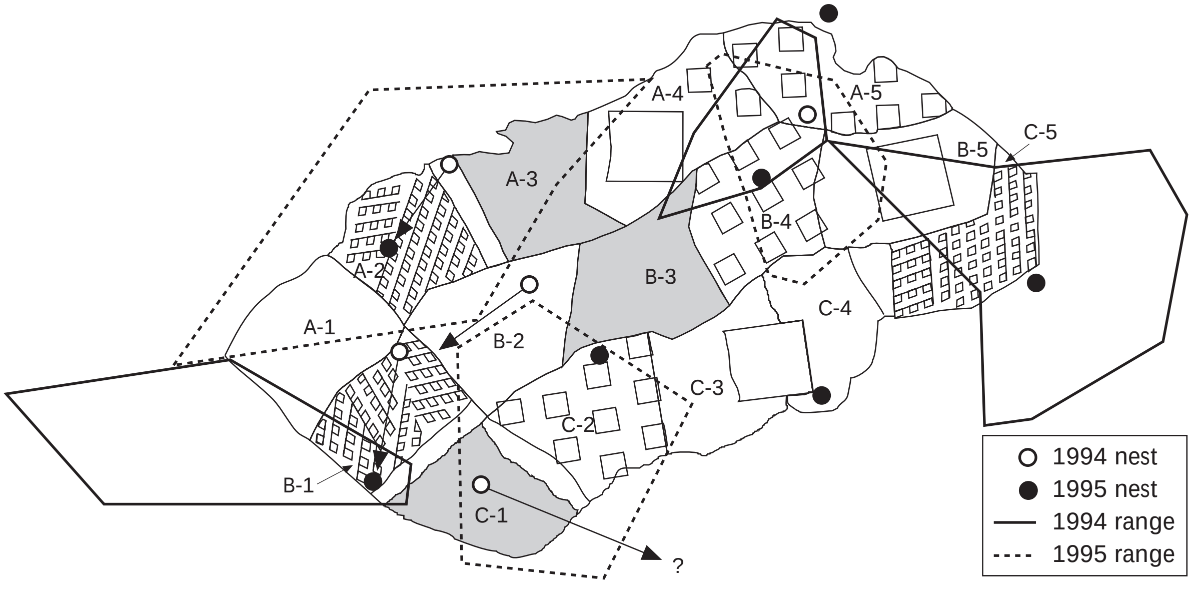

FIGURE 1 *Three-toed woodpecker nests at Sicamous Creek in 1994 and 1995, and high-use ranges for three radio-tagged birds in each year. Arrows indicate known or probable movements of banded birds' nests between years (1 cm = 400 m).*

All three-toed woodpecker nests that we found were within 50 m of a harvested edge (roads in 1994, roads or cutblocks in 1995). Our nest searches were done along transects and should not be biased towards roads or other edge areas, and we are confident that we found all but three nests on the site in the two years. This proximity to edge was far greater than would be expected by chance because a high percentage of the site was further from any edge in both years (Figure 2). The orientation of the edges adjacent to nests did not differ from random (circular $r = 0.255$, $p > 0.25$; Figure 3).

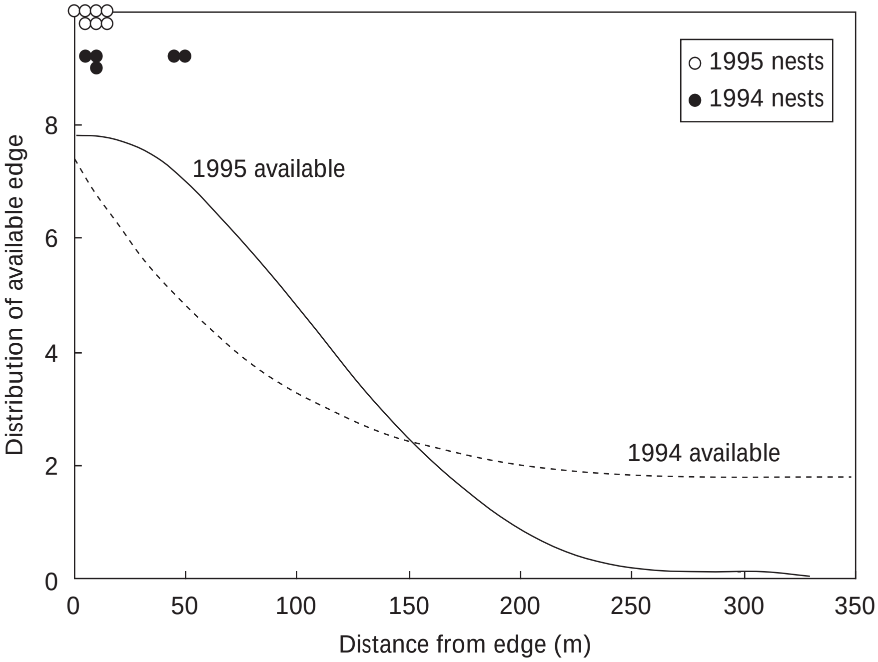

FIGURE 2 *Distances of 1994 and 1995 nests from harvested edges and the overall (available) distribution of distance from edge in the study site (1994: roads only).*

Although a restricted range of tree types is used for nesting, our random plots indicate that 8.2% of trees greater than 7.5-cm dbh on the site would be of suitable species, dbh, and profile class. Harvesting increases the amount of edge, which seems the preferred position for nests. After harvesting, 15% of the area was within 20 m of an edge. This would mean that 1.2% of trees on the site would be acceptable nesting sites for woodpeckers. This represents about 5000 acceptable sites, implying no nest tree limitation for the seven pairs at Sicamous Creek, although additional criteria are undoubtedly used in nest choice. However, nesting habitat could become a concern in managed forests. Preferred nest sites appear to be snags within 20 m of an edge, and the snag removal required within a tree height of cutblocks could therefore remove all preferred nest sites.

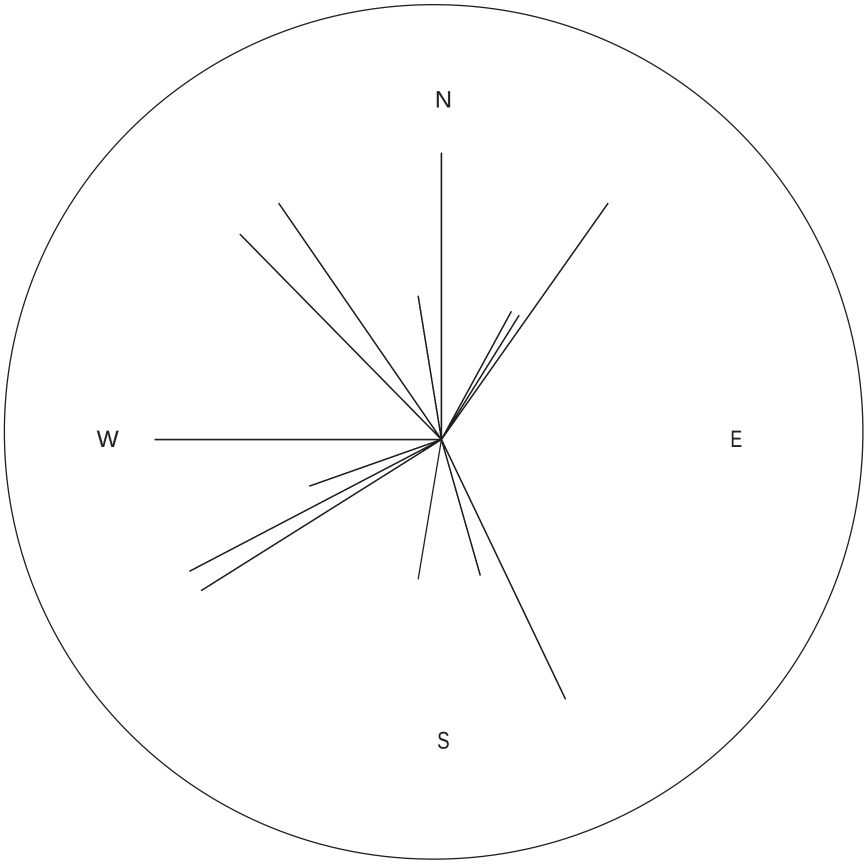

figure 3 *Orientation of edges adjacent to nests. Each line indicates one nest's edge; short lines used for each edge when nest is adjacent to two edges.*

Foraging Observations and Selectivity

We recorded 327 foraging observations in the summer before harvesting (76 series), 284 observations in the winter before harvesting (71 series), 439 observations in the summer after harvesting (61 series), and 77 observations in the winter after harvesting (16 series). Selectivity has been analyzed at the within-plot and overall scales for tree characteristics, and the overall scale for plot characteristics.

Selectivity for tree characteristics (species, dbh, and profile) was almost identical between male and female three-toed woodpeckers, and was very similar in summer and winter and in pre- and post-harvest years. Selectivity at the within-plot and overall scale was also very similar because the composition of the foraging plots did not differ much from random plots and the forest composition is fairly uniform across the site. Only within-plot selectivity is presented here.

The woodpeckers showed no obvious selectivity for either of the available tree species, subalpine fir or spruce. They did show increasing preference for larger dbh classes and relative avoidance of smaller trees in all seasons (Figure 4). Profile class 3 (recently dead) trees were always preferred, while live trees and old snags were generally avoided (Figure 5).

The few live trees that were used for foraging were usually spruce; foraging on subalpine fir was predominantly on recently dead (class 3)

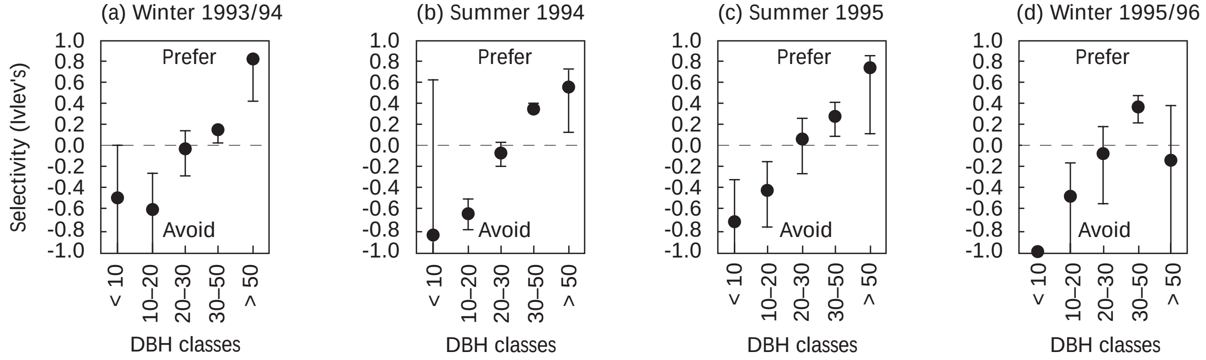

FIGURE 4 *Selectivity for dbh class: within-plot selectivity by females; overall selectivity and selectivity by males was almost identical.*

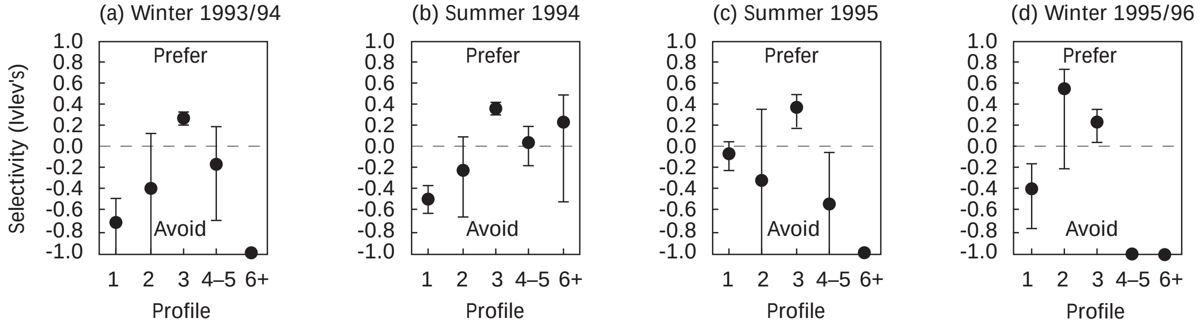

FIGURE 5 *Selectivity for profile classes: within-plot selectivity for females; overall selectivity and selectivity by males is almost identical.*

snags. Three-toed woodpeckers are usually thought to specialize at foraging by flaking bark off spruce boles. The unexpectedly high use of recently dead subalpine fir by woodpeckers at the Sicamous site may result from the abundance of these snags, and the prevalence of bark beetles (*Dryocetes* spp.) and secondary wood-boring beetle larvae in recent subalpine fir snags (L. Harder, Simon Fraser University, pers. comm., 1995). Snag removal adjacent to cutblocks would eliminate this foraging habitat. However, three-toed woodpeckers may persist in areas without snags by switching their foraging to live spruce. The snag-removal areas at Sicamous Creek will allow us to test this possibility.

The characteristics of foraging plots used by males and females also did not differ substantially. Foraging plots had higher densities and basal areas of snags than random plots, but similar levels to the matched plots. This reflects the use of recent subalpine fir snags by woodpeckers, and the clumped distribution of these snags. Any differences in foraging-plot characteristics therefore seem to result from the preference for recent subalpine fir snags, not any selectivity for the plot characteristics themselves. In other words, foraging choices seem based primarily on the tree itself, not the surrounding 0.01-ha area.

CONCLUSIONS

Three-toed woodpeckers are the only woodpeckers commonly breeding at Sicamous Creek with six to seven pairs in the study area. This implies a territory of roughly 1 km² per pair. Radio-telemetry indicated some birds travelled much further to forage in other forest types. These ranges and movements are much larger than typical harvest reserves, meaning that most woodpeckers in managed forests will be affected to some degree by harvesting.

Woodpecker nests are located in recently dead subalpine fir snags of moderate size which are near forest edges. Suitable nesting locations appear abundant at Sicamous; however, snag removal around the edges of cutblocks could seriously reduce the availability of preferred nesting habitat.

Foraging woodpeckers select recently dead snags, particularly subalpine fir, with increasing preference for larger-sized stems. Removing snags, reducing "pest" insects associated with recent snags, and shortening rotations to reduce the number of trees dying may all substantially decrease the abundance of this preferred foraging habitat. However, three-toed woodpeckers also use live spruce and may switch to this tree type if snags are eliminated.

Transect surveys, foraging observations, and nest searches are continuing at Sicamous Creek. This work will document longer-term changes in woodpecker abundance or distribution following harvest, determine whether harvested edges affect foraging habitat, and measure the response of woodpeckers to the snag-removal areas. More intensive radio-telemetry will delineate home ranges and determine whether removal of forest and snags within a home range leads to increased home range size.

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Effects of Harvest Treatments on the Abundance of Small Mammals at Sicamous Creek

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ABSTRACT

Mark-recapture live-trapping was used to examine the effects of four harvesting treatments on small mammal populations at the Sicamous Creek research site in 1994 and 1995. Population size of red-backed voles and deer mice declined from 1994 to 1995 on almost all of the treatment units, but long-tailed vole populations did not display this trend. Similarly, populations of both red-backed voles and deer mice did not show consistent trends on most of the harvesting treatments, but appear to be declining on the 10-ha opening treatments. Long-tailed vole population size was highly variable in relation to the harvesting treatments in both 1994 (before harvest) and 1995 (after harvest). Similar variability in trends both between and within species was noted for survival and body weight. These results suggest that either the brief one-year monitoring period after harvest is not adequate to clearly identify demographic trends, or the effects of the harvesting treatments on small mammals are highly variable. Future analyses will focus on evaluating trends in the second and third years after harvesting and on assessing the relationship of small mammals to habitat attributes within treatments.

INTRODUCTION

Maintaining biodiversity has become an important land management objective in British Columbia (Fenger et al. 1993). Small mammals are one of the biodiversity indicator groups that are being monitored in relation to the harvesting treatments at Sicamous Creek. These species play an important role in forest ecosystems, consuming vegetation and seeds (Banfield 1974), affecting forest regeneration (R. Parish, B.C. Ministry of Forests, pers. comm., 1995), and dispersing seeds and spores (Maser et al. 1981). Forest management practices that change the abundance of grass, forb, and shrub cover, as well as structural features of the habitat which provide nest sites and cover (e.g., downed wood), will likely lead to changes in small mammal species composition and abundance. Because of their importance to avian and terrestrial carnivores (Banfield 1974; Ehrlich et al. 1988) and the role they may play in the population dynamics of defoliating insects (Torgersen et al. 1990), shifts in the diversity or abundance of small mammals will likely have important consequences for terrestrial food webs.

Although the methods and sampling procedures for small mammals are well documented (Galindo-Leal 1990; McComb et al. 1991; Sullivan 1992), determining population trends in relation to habitat changes usually requires repeated sampling. This is because many species of small mammals show considerable variation in diversity and abundance from year to year (Sullivan 1977; Asher and Thomas 1985). In addition to these annual fluctuations, the documented effects of forest management practices on mice and voles are highly variable. Several studies have demonstrated an increase in deer mice (*Peromyscus maniculatus*) and a decline in red-backed vole (*Clethrionomys gapperi*) populations in response to both clear-cutting (Gashwiler 1984; Scrivner and Smith 1984) or partial-cutting (Medin 1986). Others noted little response by either species to harvesting (Sullivan 1979a; Medin and Booth 1989), or an increase in deer mouse populations alone (Sullivan and Krebs 1981; Walters 1989). In a series of studies in the Pacific Northwest, Aubry et al. (1991), Corn and Bury (1991), and West (1991) all found that species distribution or abundance was not consistently related to forest age or specific habitat structures, with the exception of known habitat specialists such as the shrew mole (*Neurotrichus gibbsii*) or red tree vole (*Arborimus longicaudus*). In contrast, Carey and Johnson (1995) found a higher biomass of mice and voles in old forests, and clear relationships to shrubs, herbs, and downed wood.

Much of this confusion may stem from the use of study areas where previous site history is unknown, or of different sampling methodologies. Galindo-Leal (1990) and McComb et al. (1991) noted the differences in capture success between live-trapping and snap (kill) trapping. Snap trapping also does not provide information on long-term population structure and survival within the areas being sampled. These important shortcomings limit the usefulness of snap-trapping data for certain applications, and do not permit a closer analysis of the relationships between population density and habitat suitability (van Horne 1983) or the determination of population sources and sinks (Pulliam 1988). Although several studies have examined the relationships between habitat structure and small mammal populations, the avoidance of habitat edges (Walters 1989), the use of forested corridors for travel by mice and voles (Yahner 1982; Diffendorfer et al. 1995), and habitat fragmentation (Loman 1991) have received little attention.

A shift in species composition or an increase in the abundance of some species of small mammals may not be a desirable change. Deer mice and chipmunks (*Tamias* spp.) have been identified as important nest predators of songbirds (Wilcove 1985; Angelstam 1986; Rudnicky and Hunter 1993). Also, mice and voles can seriously reduce the efficiency or increase the costs of seeding and planting operations (Sullivan 1979a, b; Sullivan et al. 1990). For example, H. Merler (B.C. Ministry of Forests, pers. comm., 1995) has observed significant damage by voles to the root systems of mature subalpine fir at the Sicamous Creek site. Habitat changes that favour species such as chipmunks or deer mice (nest predators) or voles that damage seedlings may compromise the overall objective of maintaining biodiversity, and increase the costs of reforestation.

The experimental design of the Sicamous Creek trials is a one-way analysis of variance (ANOVA) with five treatments and three replicates of each treatment. In each harvesting treatment except the control areas, approximately 35% of the timber volume was removed using the following opening sizes: 0.01 ha (uniform partial cut), 0.1 ha (small patch cut), 1 ha (large patch cut), and 10 ha (clearcut). To monitor small mammal populations, a trapping grid was established on each of the treatment units (Figure 1). With the exception of the 1-ha treatment sites, all trapping grids were established in late 1993 on a representative site within each treatment. Trapping grids were established on the 1-ha treatments in early 1995. Each live-trapping grid consists of 49 traps in a 7 × 7 array of stations at 15-m intervals. On the patch-cut treatments, the trapping grids were positioned to represent the habitat types following harvest. For example, on the 0.1-ha openings, the trapping grid extends across several openings, with about 60% of the sampling area in forest. Fifteen grids are being used to sample small mammal responses to the harvesting treatments, three grids were established to assess responses to the edges and leave strips created during harvesting, and 12 grids are being used to assess the role of downed wood on small mammal populations (V. Craig et al., this proceedings, page 243).

One Longworth-style live trap is placed at each station. Traps are baited with a mixture of oats, sunflower seed, and a slice of apple. The nest chamber of each trap is filled with coarse brown cotton for insulation. Live-trapping is conducted on each grid at three-week intervals

FIGURE 1 *Schematic diagram of the Sicamous Creek research site with locations of trapping grids (dark stippled squares) on control (A-1, B-2, C-4), uniform partial cut (A-3, B-3, C-1), 0.1-ha patch-cut (A-2, B-1, C-5), 1-ha patch-cut (A-5, B-4, C-2) and 10-ha clearcut treatments (A-4, B-5, C-3).*

during the snow-free period from mid-June to late September. During each trapping session, traps are set for two consecutive overnight periods (i.e., approximately one hour before sunset until three hours after sunrise) and are locked open during the day. Captured animals are permanently identified with unique numeric ear tags, and the species, weight, sex, and reproductive condition are recorded before the animal is released at the point of capture. Between trapping sessions, traps are locked open to allow animals to enter and maintain familiarity with the traps. When capture success exceeds 80% of the 49 traps on a sampling grid on both days of trapping session, an additional 25 traps are added at alternating stations to prevent trap saturation.

Data were analyzed with Jolly-Seber (Seber 1982) mark-recapture programs (available from Dr. C.J. Krebs, University of British Columbia, Vancouver, B.C.). These programs were used to estimate population size, survival, and average body weight for the three species captured most frequently at the Sicamous Creek site: red-backed voles, deer mice, and long-tailed voles (*Microtus longicaudus*). For each trapping grid, the Jolly-Seber estimates for each trapping session were averaged for that year, and these seasonal averages were used to represent that treatment. Sexes were combined for the analyses, and a repeated measures ANOVA (Steele and Torrie 1980; SYSTAT 1992) was used to examine treatment differences in population size, survival, and body weight before and after harvest. The Type 1 error level was set at 0.05.

RESULTS AND DISCUSSION

The number of animals captured in 1994 (before harvest) and 1995 (after harvest) is outlined in Table 1. By far the most abundant species at the Sicamous Creek site was the red-backed vole (1974 individuals), with moderate numbers of deer mice (331 individuals), and long-tailed voles (328 individuals). Other small mammals captured at the Sicamous site include

TABLE 1 *Number of individual red-backed voles, deer mice, and long-tailed voles captured by treatment before (1994) and after harvest (1995)*

Species	**Year**	**Control**	**Partial-cut**	**0.1 ha**	**1 ha**	**10 ha**	**Total**
Red-backed vole	1994	200	233	230	n/a	239	902
	1995	208	286	273	194	111	1072
Deer mouse	1994	71	64	80	n/a	64	279
	1995	17	5	3	14	13	52
Long-tailed vole	1994	44	34	53	n/a	35	166
	1995	62	11	23	12	54	162
Total		602	633	662	220	516	2633

shrews (*Sorex* spp., 240 individuals), meadow voles (*Microtus pennsylvanicus*, 119 individuals), heather voles (*Phenacomys intermedius*, 115 individuals), yellow-pine chipmunks (*Tamias amoeneus*, 76 individuals), and Northern bog lemmings (*Synaptomys borealis*, 8 individuals). Many animals were captured repeatedly, yielding a total of 3262 and 3663 captures in 1994 and 1995, respectively. The following results represent a preliminary analysis only. A minimum of three years of monitoring after harvest will be necessary to establish reliable estimates of population trends. Other parameters such as home range size will also need to be evaluated.

Population Size

From 1994 to 1995, the population size of red-backed voles decreased across all treatments ($p = 0.025$, Figure 2), and showed a tendency towards a year × treatment effect ($p = 0.088$). The strong year effect reflects an overall decrease in red-backed vole populations at the Sicamous Creek site regardless of treatment, while the year × treatment trend reflects a decrease in red-backed vole numbers on the 10-ha openings, but little change elsewhere. The high variability in red-backed vole numbers in the partial-cut treatments in both 1994 and 1995 indicates large differences in population size across the three replicates. Deer mice also showed a strong decline in population size from 1994 to 1995 ($p < 0.001$), and a tendency for lower populations on the partial-cut, 0.1-ha, and 10-ha treatments than on the control and 1-ha opening treatments ($p = 0.058$). In contrast, long-tailed voles showed neither year × year ($p = 0.730$) nor year × treatment differences ($p = 0.557$). These results illustrate that the species being monitored show different responses to the harvesting treatments, and that the short monitoring period after harvest may not be adequate to establish reliable trends.

Survival

The survival of red-backed voles in the summer of 1994 was similar to the 1994–95 over winter period ($p = 0.824$, Figure 3) across treatments, but there was a significant year × treatment effect ($p = 0.045$). Over winter survival on the control areas was higher than during summer, while on the partial-cut areas, summer survival was higher than during the 1994–1995 over winter period. Deer mice did not show any significant differences either by year ($p = 0.772$) or year × treatment ($p = 0.358$). Long-tailed voles showed higher survival in the 1994–1995 over winter

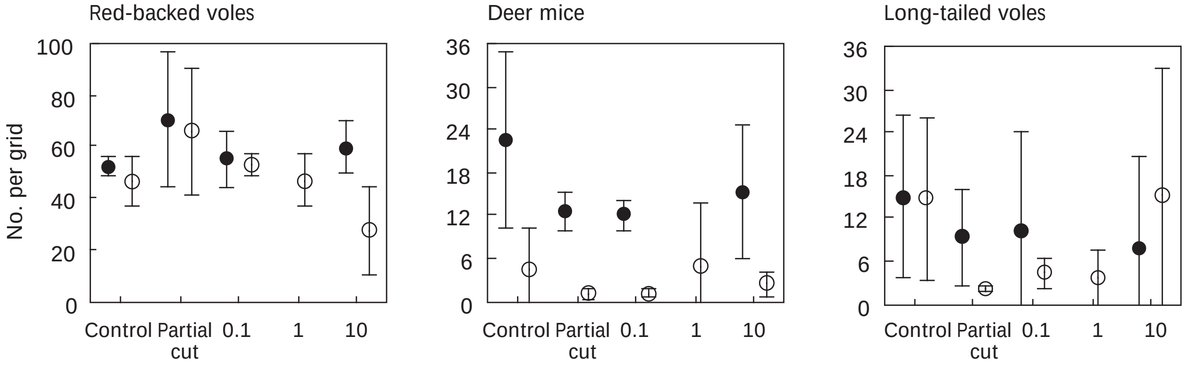

FIGURE 2 *Estimated mean number (+ 2 SE) of small mammals captured per grid before (1994 ●) and after harvest (1995 ○) using Jolly-Seber population estimates.*

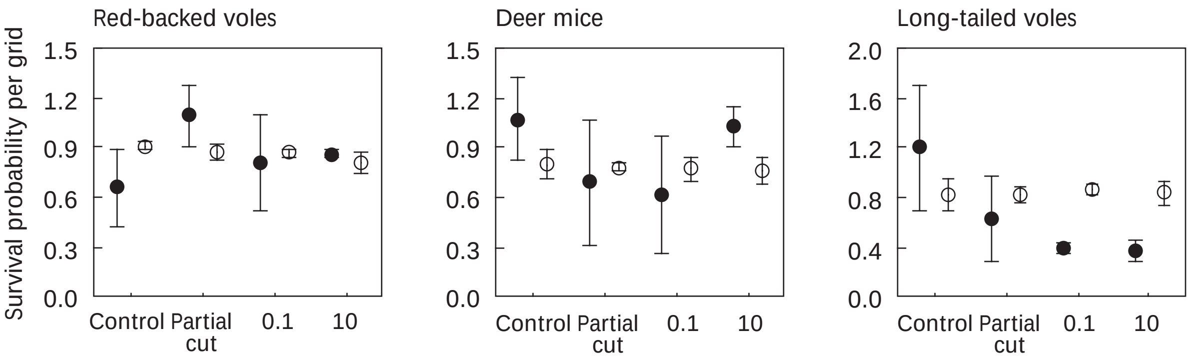

FIGURE 3 *Estimated mean (+ 2 SE) four-week probability of survival for small mammals in summer 1994 (●) and overwinter 1994/95 (○) using Jolly-Seber survival estimates.*

period ($p = 0.040$) than during the 1994 summer, but no year × treatment differences. As with the estimates of population size, the survival estimates do not show consistent patterns across the range of opening sizes at the Sicamous Creek site for either red-backed voles, deer mice, or long-tailed voles.

Body Weight

Body weights of red-backed voles tended to decrease from 1994 to 1995 ($p = 0.129$, Figure 4), and also showed a weak trend for lower body weights on the partial-cut, 0.1-ha, and 10-ha openings ($p = 0.146$). Deer mice did not display this trend (year effect: $p = 0.928$; year × treatment effect: $p = 0.350$), but long-tailed voles showed both an overall decline in body weights in 1995 ($p = 0.011$) and a tendency for reduced body weights on all treatments except the control.

CONCLUSIONS

The preliminary analyses presented above indicate high variability in response by several species of small mammals to the changes in forest

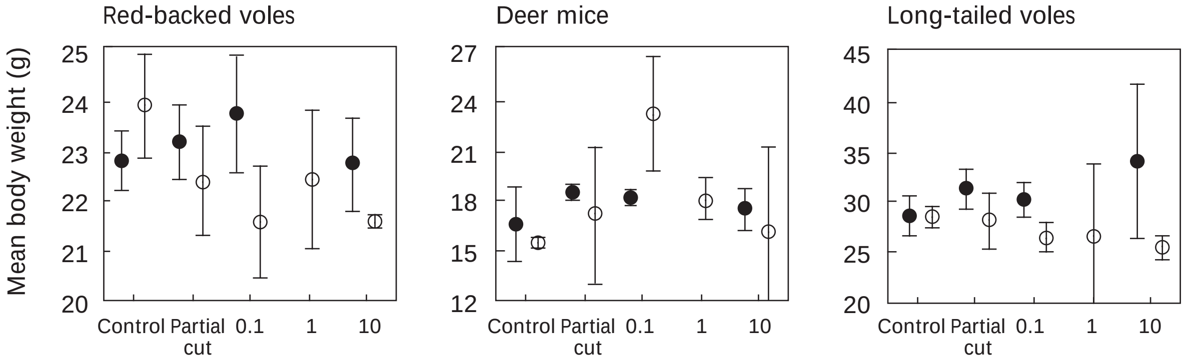

FIGURE 4 *Mean (+ 2 SE) body weight of small mammals captured per grid before (1994 ●) and after harvest (1995 ○).*

structure arising from several different types of harvesting entries. The decline in population size of red-backed voles and deer mice, and the increase in long-tailed voles on the 10-ha clearcut areas will need to be monitored for at least two more years to evaluate population trends as the biomass and vigour of grasses, forbs, and shrubs increase. Consistent trends in population size, body mass, and survival both within and across species in the different harvesting treatments may become apparent over the next two years of monitoring, or show continued high variability. The present lack of a clear pattern may reflect ecological reality, or be the result of habitat variability and/or the influence of surrounding forest edge habitats.

Future analyses will incorporate a minimum of two additional years of monitoring, and will assess the role of specific habitat structures (e.g., grass, forb, and shrub cover, mosses and lichens, etc.) and the role of adjacent forest edges in influencing capture success. Several habitat structures will be quantified in 1996 using 2 and 5.6 m radius plots around a sample of 25 trapping stations on each small mammal trapping grid. Some of the habitat structures that will be measured include: (1) an index of ground moisture; (2) soil composition (mineral/organic); (3) percentage of area covered by terrestrial mosses and lichens, and average depth; (4) grass cover (none, medium, or high); (5) herb cover (none, medium, or high), and three dominant shrub species; and (6) the distance to forested edges or openings.

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Relationships Between Small Mammals and Downed Wood at Sicamous Creek

VANESSA CRAIG, WALT KLENNER, AND TOM SULLIVAN

ABSTRACT

We monitored small mammal populations on uncut (control) blocks and 10-ha clearcut treatment blocks during the summer and fall of 1994 (pre-treatment) and 1995 (post-harvest) as part of an ongoing study on the relationship between small mammals and downed wood. The study site was harvested during the winter of 1994–95. We manipulated the amount of downed wood on six 1.7-ha areas on the clearcut blocks in 1995 (July–October) to provide a replicated range in the amount of downed wood on the site. The data collected to date show no clear relationships between change in small mammal population size with forest type or amount of downed wood. However, the four-week probability of survival of red-backed voles over the winter of 1994–95 was lower on clearcut areas than forested areas. Because of the wide variation in small mammal population size, the short time since harvesting, and the lack of a full post-treatment year of data collection on areas with downed wood manipulations, these data are preliminary. Future years of data collection will provide us with more complete information on the initial and longer-term relationships between these treatments and small mammal populations.

INTRODUCTION

Maintaining biological diversity in forested ecosystems is increasingly important and has received more emphasis with the introduction of the Forest Practices Code and its associated biodiversity guidelines. Research has provided information on the effects of forestry on large mammal species; however, little information is available about the effects on small mammals or other components of biodiversity.

Mice and voles have an important role in the forest ecosystem. Some small mammal species such as the red-backed vole (*Clethrionomys gapperi*) and deer mouse (*Peromyscus maniculatus*) distribute mycorrhizal fungi, which is essential to seedling survival, throughout plantations. A study which investigated repeated Douglas-fir plantation failures within an area in southern Oregon suggested that the plantations failed because the mycorrhizal fungi usually associated with the planted seedlings were lacking (Amaranthus and Perry 1989). Small mammals constitute the majority of the diet of many species (e.g., marten, coyotes, owls and other raptors) and therefore are important in the maintenance of local biodiversity and are integral to the normal functioning of the ecosystem. However, small mammals can also have deleterious effects. Voles can decimate plantations of seedlings by girdling the stems or clipping the leaders. In addition, conifer seed predation by deer mice has discouraged the use of a reforestation technique (seeding) that may prove advantageous in some areas (Sullivan and Sullivan 1982). On the Sicamous Creek Silvicultural Systems research project site, small mammal damage to subalpine fir tree roots was identified as a major damaging agent which may be related to tree mortality (Merler, this proceedings, page 41).

The basic requirements for all animals are food, water, and cover. The importance of cover to small mammals while travelling and foraging is well established (e.g., Anderson 1986; Merkens et al. 1991; Harestad and Shackleton 1990) and is presumed to reduce the risk of predation by their many terrestrial and avian predators. Small mammals will avoid or show reduced use of areas with inadequate cover (Anderson 1986; Harestad and Shackleton 1990; Craig 1995). Studies have shown that vegetation and downed wood are important habitat components, providing food and cover (Morris 1979; Kaufman et al. 1983; Hayes and Cross 1987; Barnum et al. 1992; Planz and Kirkland 1992; Carter 1993; Amaranthus et al. 1994; Tallmon and Mills 1994; Craig 1995). Downed wood also provides nesting and foraging sites (Thomas 1979; Maser et al. 1981; Maser and Trappe 1984).

Research on small mammals has concentrated on determining habitat features associated with small mammal capture sites, but few studies have investigated the use of a particular habitat feature in detail. To maintain populations of small mammals in managed stands, current management practice advocates leaving adequate amounts of downed wood on the sites. This approach requires testing because:

- information on the amount of downed wood required by small mammals is inadequate, and
- management on harvested sites has long-term effects on site characteristics through second-rotation harvests.

With this study, we are testing the assumption that downed wood is important to provide cover and nest sites for small mammals, and that small mammal populations depend on specific levels of downed wood for their presence.

In this paper we discuss some general results from the first two years of our study. Full analysis of treatment effects will require multiple years of post-treatment data, complete analyses of population demography, such as sex ratio and growth rates, and inclusion of microhabitat data, which is not yet complete (post-treatment microhabitat data collection around trap stations will be completed in 1996).

To determine the influence of downed wood on small mammal population size and community diversity, we are studying small mammals on forested and clearcut areas with varying amounts ("low," "medium," or "high") of downed wood. The experimental design is a 2 × 3 factorial (forest and clearcut, three levels of downed wood, with three replicates of each treatment and control).

In 1993, three small-mammal trapping grids (100 × 100 m) were placed in each of the three, 10-ha clearcuts (nine grids), and in each of the forested control blocks (nine grids). Each trapping grid consists of a 7 × 7 m trapping station array with stations 15 m apart. A livetrap is placed at each station and filled with coarse brown cotton, oats, and apple. Each grid is trapped for a two-day period every three weeks during the snow-free period (June/July–Sept./Oct.), resulting in four trapping sessions in both 1994 and 1995. Animals captured are identified to species, and their sex, weight, and breeding condition recorded. Each animal is marked with an individually numbered ear tag, and released at its point of capture.

Pre-treatment data were collected in 1994. The site was logged during the winter of 1994–95, and downed wood manipulated during August to October 1995 on clearcut areas. Downed wood on the forested grids was manipulated in July–August 1996.

Amounts and characteristics (diameter, decay class) of downed wood were measured on each treatment area in 1994 (pre-treatment) and 1995 (post-harvest), and were measured again in 1996 (post downed-wood manipulation on clearcut areas), and will also be measured in 1997 (post-downed wood manipulation on forested areas). Volume of downed wood on the site was estimated following Van Wagner (1968).

Downed Wood Manipulations

Within each block, treatments designated as "low," "medium," or "high" were randomly assigned to each of the three grids in each block. The treatment areas extend outside each grid, providing a 15-m treatment buffer. The "low" treatment had all wood greater than 6 cm in diameter removed (the excavator was to destroy all logs of decay class 4 or 5), the "medium" site had the normal amount of wood left on site, and the "high" had extra wood (taken from the surrounding area) scattered over the site.

Manipulations began during harvesting of the study areas in the winter of 1994–95. The skidder operator moved as much wood as possible off the "low" downed-wood sites, and left extra downed wood (in the form of downed snags that normally would be removed from the site) on the "high" treatment sites. During site preparation (mounding, August–October 1995), the excavator operator removed as much downed wood as possible from "low" sites, and redistributed piles of downed wood on "high" sites more evenly across the grid. A Worker Development Crew from the Salmon Arm Forest District moved all remaining downed wood greater than 6 cm in diameter on “low” sites off the grids (September–October 1995).

Similar types of manipulations occurred in the forested grids during the summer of 1996. All the manipulations in the forested areas were completed by hand.

Habitat Structure

Habitat structure was assessed pre- (1994) and post-harvest (1995). Habitat characteristics (including vegetation, surface cover, downed wood, and trees) were measured around each trap station within 2 m radius plots. More extensive post-treatment measurements were in 5.6 m radius plots around trap stations in 1996. These measurements include more detailed information on vegetation, surface cover, and downed wood, as well as other measures. More detailed descriptions of the type of data collected are provided in Huggard and Klenner (this proceedings, page 200). Habitat data will be analyzed in conjunction with the trapping data when assessments are complete.

Analyses

Estimates of mean volume (±2 standard errors of the mean [SE]) of downed wood both pre-harvest and post-harvest (mid-treatment) are presented to provide an indication of the variability of downed wood present.

Jolly-Seber mark-recapture population analysis (Jolly 1965) using capture data from trapping grids, provides estimates of the population size and the four-week probability of survival estimates of each species of small mammal captured. Mean four-week survival probability estimates (±2 SE) are presented for summer 1994 and winter 1994–95. Jolly-Seber population growth rates, generated for each trapping session, were analyzed using a randomization program designed for use with time-series data (W. Hochachka, University of British Columbia, 1993). Population growth rates were used as a demographic estimate instead of population size because they are less autocorrelated over time and the results are therefore easier to interpret. The program conducts 5000 randomizations of the data to determine whether the before treatment/after treatment changes in dynamics within a grid are affected by differences among treatments.

All significance levels (α's) were adjusted to account for experiment-wise error.

RESULTS AND DISCUSSION

Downed Wood Manipulations

The pre-harvest amount of downed wood on the study grids was generally similar across treatment areas (Figure 1). The pre-harvest mean volume on uncut–high (future “high” forested treatment) blocks was more than two standard errors lower than the clearcut–low blocks, but it did not differ from the other forested blocks.

Data collected on downed-wood volume in July 1995 (early downed-wood manipulations) illustrate the progress in manipulating the level of downed wood during harvesting. The mean volume of downed wood on the clearcut areas (clearcut–medium represents the "norm") is similar to the pre-harvest volumes (Figure 2). In addition, the characteristics of the downed wood on site (i.e., diameter and decay class profiles) are also similar (Huggard and Klenner, this proceedings, page 200). Future data collection will provide information on how the mean length of pieces varies across treatments. This is a potentially important characteristic of downed wood because small mammals use lengths of wood as travel corridors (Barnum et al. 1992; Craig 1995). The amount of downed wood on the clearcut treatments was further manipulated after these measurements

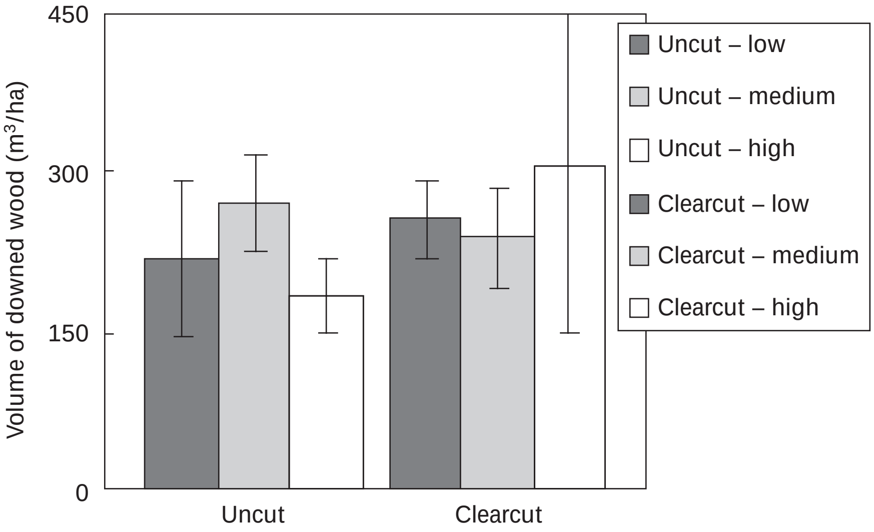

FIGURE 1 *Mean volume of downed wood (± 2 SE) on study areas before harvest (1994).*

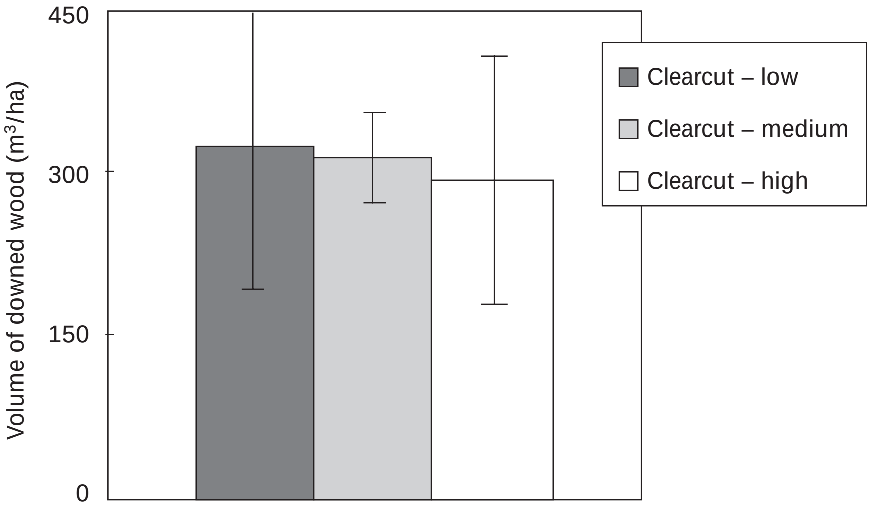

FIGURE 2 *Mean volume of downed wood (± 2 SE) on clearcut treatment areas in July 1995 after harvest and with early downed-wood manipulations.*

were taken—on “low” treatment areas by machine and worker development crews (completed in October 1995) and on “high” treatment areas by machine (completed in August 1995). Movement of downed wood on the low treatment sites was quite successful and resulted in a large decrease in the amount of wood on the site (pers. obs.). Measurements in 1996 provided post-manipulation data on volume and composition (diameter, decay class, length) of downed wood present across treatments.

Small Mammal Populations

The deer mouse, the long-tailed vole (*Microtus longicaudus*), and the red-backed vole were captured frequently enough in 1994 and 1995 to conduct meaningful mark-recapture analyses. Captures of meadow voles (*Microtus pennsylvanicus*), heather voles (*Phenacomys intermedius*), and chipmunks (*Eutamias amoenus*) were infrequent. Analyses of potential treatment effects on these populations will require more complicated analyses and are not discussed here.

Figures 3–5 illustrate the estimated population sizes for each species over time. The last trapping session (1994) was cut short by snow, thus some of the values do not have error bars, as trapping occurred only in one of the three replicates during that session.

Deer mouse populations decreased substantially, and fairly uniformly, between 1994 and 1995. Red-backed vole populations were slightly smaller in 1995, while long-tailed vole populations were relatively constant between years. The estimated population size of all three species fluctuated over time, reflecting natural processes such as birth, death, immigration, and emigration. None of the species showed statistically different responses in population growth rate between the representative uncut (uncut–medium, the “norm”) and clearcut areas (clearcut–medium, the “norm”) ($p > 0.05$, $\alpha = 0.05$). However, red-backed vole populations were lower on the three clearcut treatment areas than on the forested areas (Figure 5).

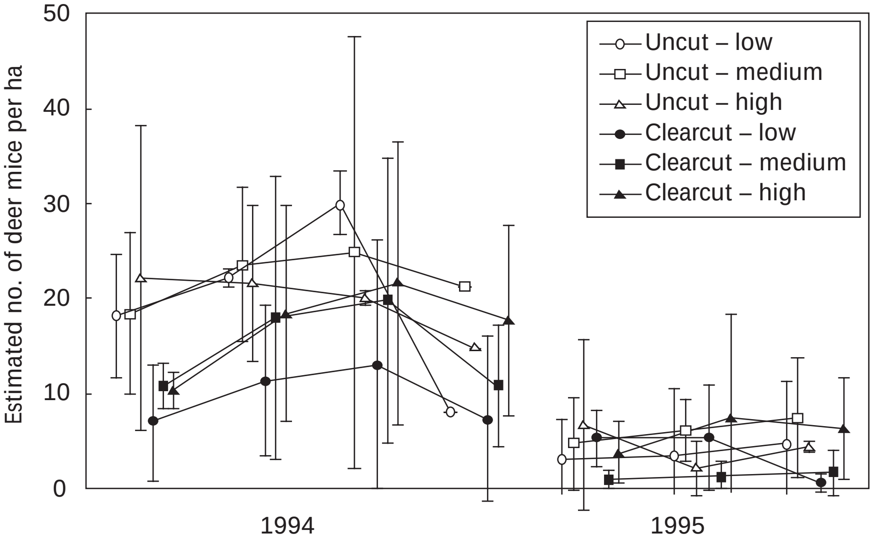

FIGURE 3 *Mean population estimates (± 2 SE) of deer mice on treatment areas before (July–October 1994) and after (July–September 1995) harvest.*

No significant relationship between population size and downed-wood treatment on clearcuts was evident ($p > 0.05$, $\alpha = 0.0167$). Deer mice populations did show significant differences between the “low” downed-wood treatment and the “medium” and “high” treatments ($p < 0.005$, $\alpha = 0.0167$). However, this more likely reflects a pre-harvest treatment effect (1994) because the deer mouse population was much lower on the pre-

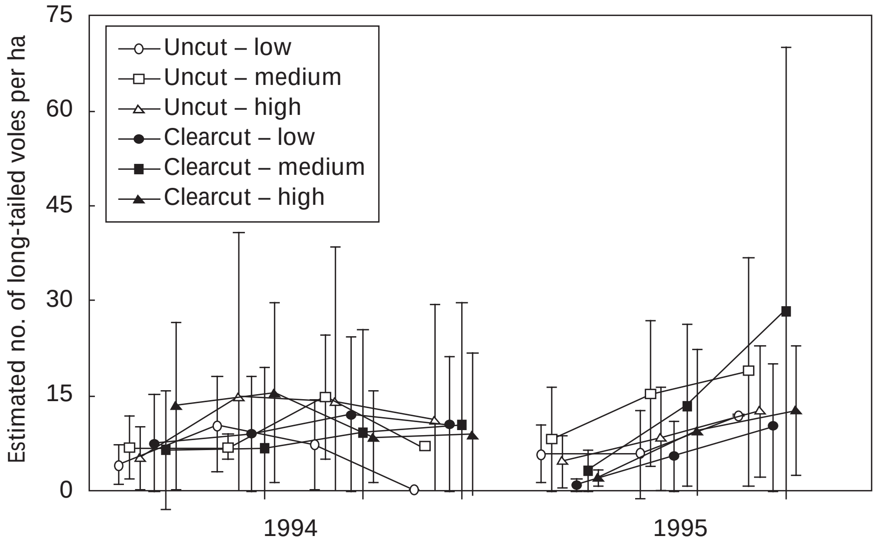

FIGURE 4 *Mean population estimates (± 2 SE) of long-tailed voles on treatment areas before (July–October 1994) and after (July–September 1995) harvest.*

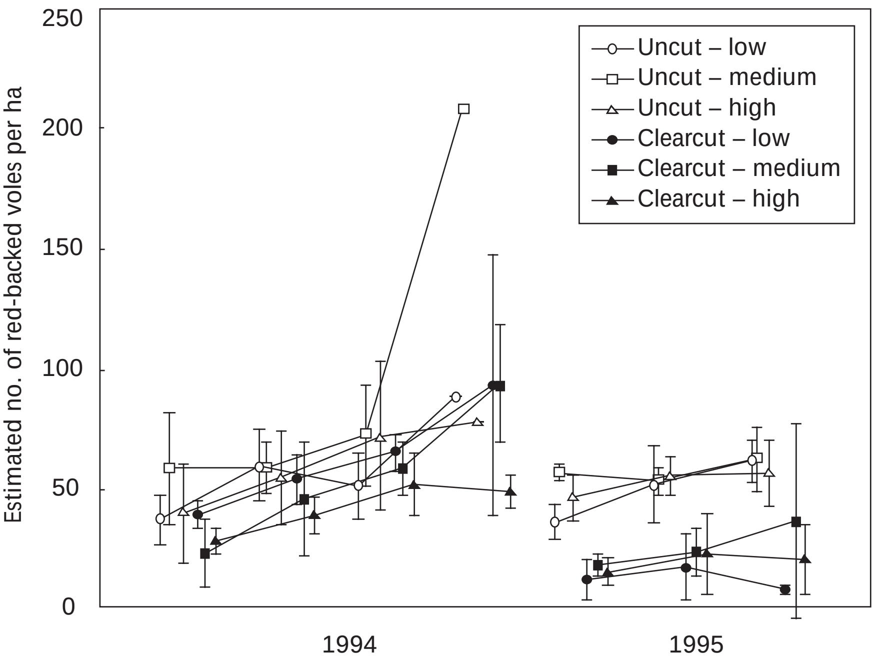

FIGURE 5 *Mean population estimates (± 2 SE) of red-backed voles on treatment areas before (July–October 1994) and after (July–September 1995) harvest.*

harvest clearcut–low treatment blocks than any other blocks (Figure 3).

The variability of small mammal population sizes makes it difficult to identify treatment effects without multiple years of data. In addition, because the downed-wood manipulations were not completed until October 1995 (the last trapping session for the season was in September), the data presented are not strictly post-treatment data. Additional data will permit us to more confidently assess treatment effects.

Survival

One of the potentially most important indicators of treatment effects is survival rates (which reflect both death of animals as well as length of stay on an area) of individuals over time. Because of the nature of mark-recapture analyses (requiring extended data), only four-week survival estimates for summer 1994 and winter 1994–95 are available (Figures 6–8).

Summer and winter survival probabilities are within two standard errors of each other for all species; however, survival estimates for long-tailed voles tended to be higher overall during the winter than summer (Figure 8). Red-backed voles appeared to respond differently immediately after harvesting; the survival estimates for all of the cut grids are more than two standard errors lower than the control grids during the winter. This may reflect either an increased mortality rate on the clearcut grids (including increased mortality during harvesting), or an emigration of animals unwilling to live on a disturbed area. Future over-winter survival data will clarify the cause and whether the result is biologically meaningful.

No differences between downed-wood treatments are evident within the first winter. However, during the winter of 1994–95, downed-wood levels did not vary significantly (Figure 2). Over-winter survival data for 1995–96 will provide a better indication of the importance of downed wood for these animals.

CONCLUSIONS

Initial results suggest that deer mice and long-tailed voles were not affected adversely by harvesting within the first year. Red-backed voles, however, show a slightly reduced probability of survival on clearcut areas when compared with forested areas, which may reflect increased mortality or emigration.

No differences in response of the three species were noted among the downed-wood treatments. This result may reflect the fact that manipulations of downed wood were not completed until after the trapping season had ended. The data presented are preliminary. Because of the variable nature of small mammal populations, the short time since harvesting, and the lack of a full post-treatment year of data collection, additional years of monitoring small mammal populations are required to confidently assess short term as well as potential long-term treatment effects.

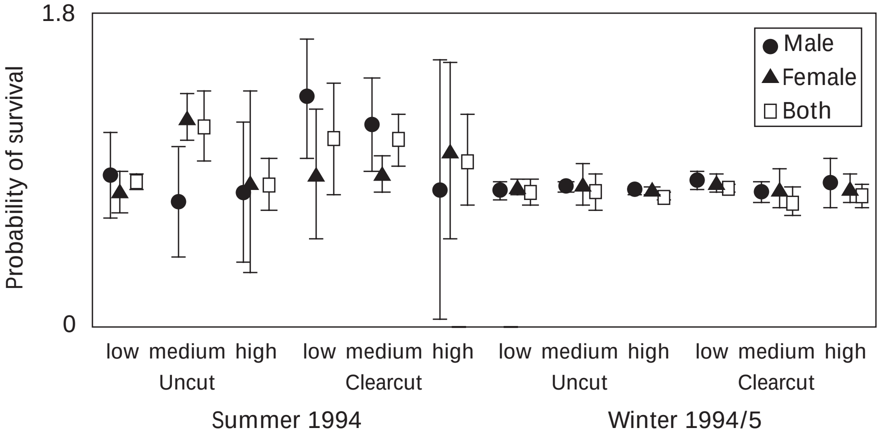

FIGURE 6 *Mean four-week survival probability estimates (± 2 SE) of deer mice during the summer of 1994 and the winter of 1994–95.*

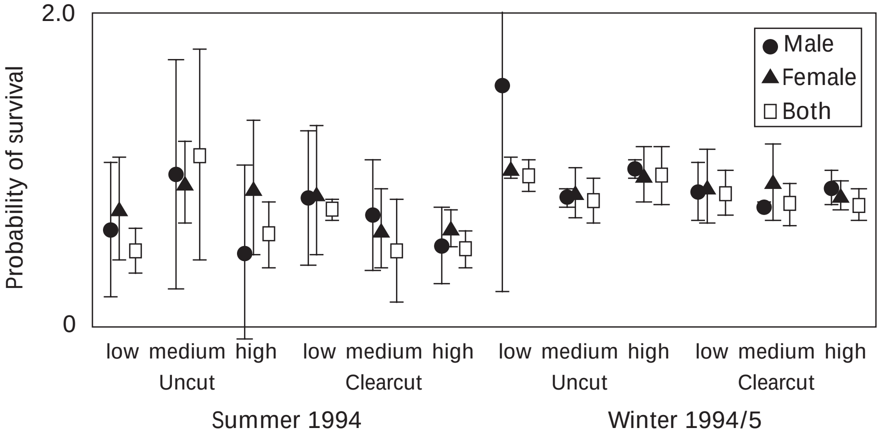

FIGURE 7 *Mean four-week survival probability estimates (± 2 SE) of long-tailed voles during the summer of 1994 and the winter of 1994–95.*

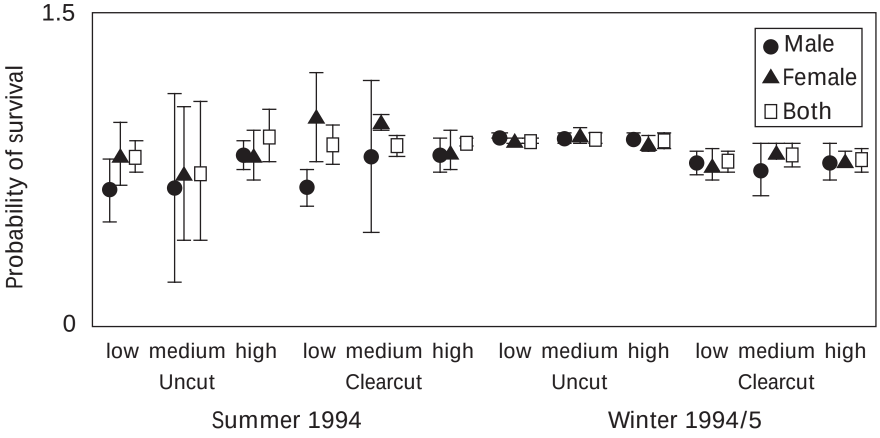

FIGURE 8 *Mean four-week survival probability estimates (± 2 SE) of red-backed voles during the summer of 1994 and the winter of 1994–95.*

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The Effects of Alternative Silvicultural Practices on the Songbird Communities at Sicamous Creek and in other Engelmann Spruce – Subalpine Fir Forests

THOMAS DICKINSON AND ERNEST LEUPIN

ABSTRACT

The community of songbirds breeding in the experimental forest at Sicamous Creek has been studied since 1992. Early morning song censuses were used to collect information on the abundance and diversity of breeding birds in the original forest (1992–93), following the construction of logging roads (1994), and after the completion of experimental harvesting treatments (1995). The song census techniques used to produce these inventories have proven to be an accurate way to detect the species richness and abundance of individuals in this forest.

Breeding bird diversity in the original forest was relatively low (18 species) and similar to the communities that occur in remnants of original forest in other high-elevation forests. A small number of "core" species (approximately 10 species that were relatively abundant and for which successful breeding was confirmed) dominated the original community at Sicamous Creek. Following the development of roads and the harvest of timber from the stand, all of the core species remained in the remnant forest patches. However, the relative densities of at least two of the core species changed significantly. In addition, the spatial distribution of individuals of several species has changed. These changes may reflect the loss of particular features of the habitat required by these species, or the realignment of competitive interactions in the original community. Future work will continue to monitor change in the community and test explanations of why the changes are occurring.

INTRODUCTION

General Background

Each spring, songbirds returning from the neotropics join with resident species to form a breeding assemblage in temperate forest ecosystems (Hagan and Johnston 1992; Finch and Strangel 1993). The community of songbirds breeding in a forest has become a commonly used indicator of the effects of disturbance, especially development disturbance, on the habitats found there (Temple and Wiens 1989; Maurer 1993). This attention to songbirds in forests has come about for several reasons. First, the forest resources that provide food, nest sites, song perches, and roosting sites are precisely those altered by development activities (Blake and Karr 1987; Haila et al. 1989; Robbins et al. 1989). Second, many songbird species that breed in temperate forest ecosystems are neotropical migrants and recent changes in their populations have focused attention on the effects of activities on their breeding ground (Terborgh 1989; Sherry and Holmes 1992). Finally, bird numbers are relatively easy to monitor because species-specific songs are used to advertise a male songbird's territories to potential mates and rivals. Monitoring songbird populations has proven to be an effective way to detect both large- and smaller-scale changes in ecosystems (Askins et al. 1987; Hunter 1990; Freemark and Collins 1992; Hagan et al. 1996). Nevertheless, because of the remoteness and relative inaccessibility of high-elevation forest systems, little is known about the make-up and dynamics of its songbird communities.

Objectives

We have been collaborating with other researchers in the Kamloops Forest Region since 1991 to study the ecology of high-elevation Engelmann Spruce – Subalpine Fir (ESSF) forest ecosystems. Our general objectives in this research are to:

- describe the composition of the songbird community in high-elevation forests and to document the typical range of year-to-year variation in community structure in both managed and unmanaged stands;
- determine the manner in which various songbird species use the resources in ESSF forest ecosystems and some of the specific habitat attributes linked to their successful breeding;
- measure the effect that different silvicultural systems have on breeding songbirds, both in the harvested stands and in the fragments of original forest which remain after harvesting; and
- follow the recovery of the songbird community back to its original condition after harvesting.

We present some general findings from our studies of songbirds that breed in high-elevation ecosystems in the Kamloops Forest Region and particularly, the preliminary results of our studies of the effects of the experimental harvesting at Sicamous Creek.

The Results of a Retrospective Study

In 1991, we used song censuses to study the effects of different silvicultural practices in the Blomley Creek watershed located approximately 5 km southeast of East Barriere Lake in the Kamloops Forest District (for details see Dickinson and Flood 1992). Twelve stands, all of approximately 50 ha, were studied. These stands included three that were composed of the original forest, seven that were selectively harvested over the previous 20 years (one harvested in 1969, three in 1981, and three in 1986), and two that were clear-cut (one in 1977 and one in 1986). Like the forests at Sicamous Creek, the high-elevation forests in this watershed are classified as being within the ESSFwc2 biogeoclimatic subzone (Lloyd et al. 1990).

The results of this study are illustrated in Figure 1, which shows the habitat associations of some of the most common species recorded in censuses. Several key points emerged from this retrospective study. First, relatively few species seemed to be only associated with remnants of

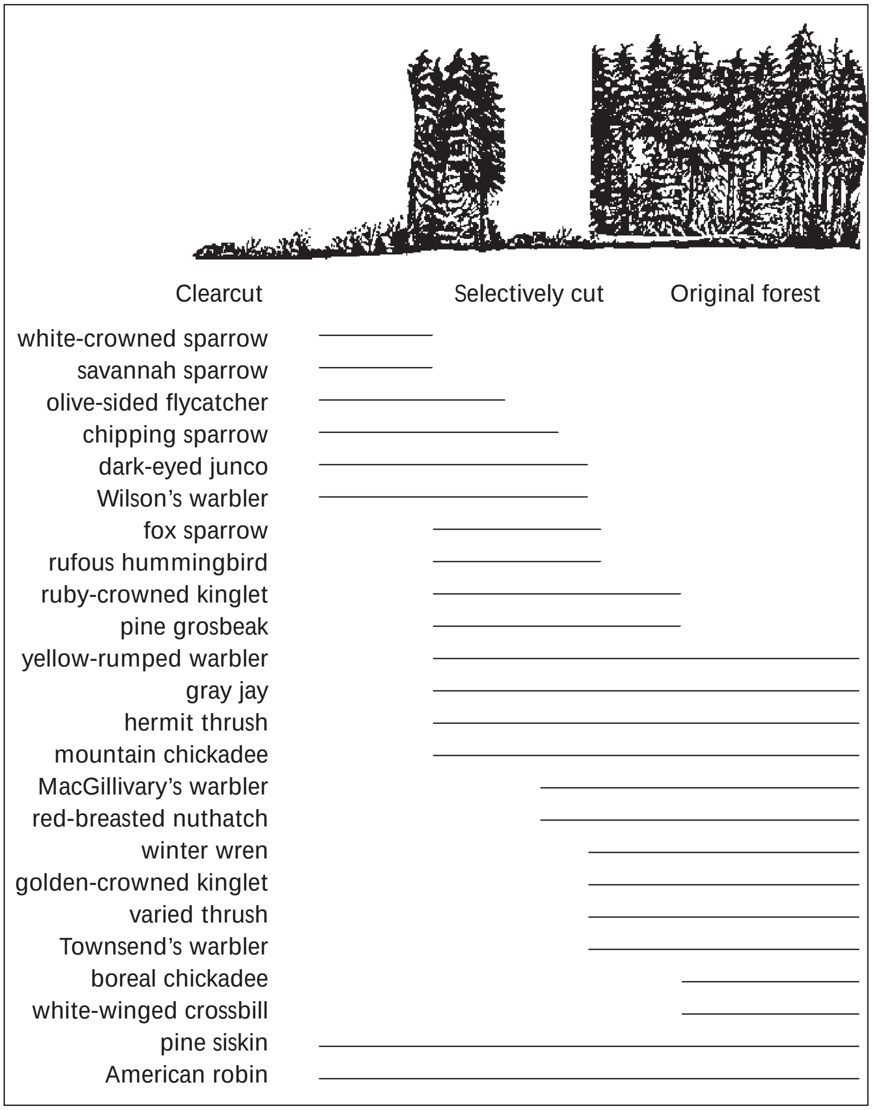

FIGURE 1 *Habitat associations of some common bird species at Blomley Creek.*

original forest. Some species, such as varied thrush (see Appendix 1 for scientific names of songbirds), have been described by others (e.g., Bryant 1995; Sheick et al. 1995) as species that require the attributes provided by forest interior habitats. Second, several species (e.g., white-crowned sparrows) clearly appear to be associated with habitats created as a result of clearcut logging; again, other workers have noted similar associations for these species in different forest ecosystems (e.g., Hutto et al. 1993). Finally, most species breeding in this forest showed a range of tolerance for different silvicultural practices. Some, such as hermit thrush, could be found in most selectively harvested stands as well as in the original forest. Others, especially winter wrens, were abundant in the original forest, but would also occupy some harvested stands after those stands had regenerated to a certain extent.

The results of this pilot research pointed out the need for a controlled study of how harvesting activities affect the songbird habitats in ESSF stands. The experimental study proposed for the Sicamous Creek research site provided just such an opportunity. At Sicamous, we could study the effect of experimentally producing different opening sizes on the songbird community. To a limited degree, our retrospective study could tell us what we could expect as a result of manipulating the size of openings.

METHODS

Study Site

Since the spring of 1992 we have focused our efforts on studying the songbird communities in ESSF forests in the Sicamous Creek Research Forest. As described elsewhere in this volume (see Vyse, this proceedings, page 4), this study area is located approximately 15 km south of the town of Sicamous, B.C. The original stand in the experimental area was composed of a more-or-less continuous mature forest on a slope with a northern aspect. In the winter of 1994, four different harvesting treatments were applied to 30-ha experimental treatment blocks within the stand, in a replicated randomized block design. Our study has concentrated on censusing the songbirds breeding in approximately 500-ha of this forest.

In 1992, nine parallel transect lines separated by 250 m were established in the original forest at Sicamous (Figure 2, Lines A–I). On each line, permanent stations were located every 250 m producing a grid of census points. In 1993, this grid of points was enlarged by adding two additional transects (A and J). The length of the existing transects was increased by 750 m to cover the entire proposed experimental area. In 1993, another transect was also added along the Mara Mountain Lookout trail in an area that was not slated for harvesting.

Monitoring Songbirds

We have used unlimited-radius point counts to monitor the abundance and diversity of breeding birds in our study areas (Reynolds et al. 1980; Ralph et al. 1995b). To address some of the potential shortcomings of this technique, we have limited potential sources of error and verified the precision of the method. A discussion of this technique and the results of some of our analyses are contained in Appendix 2. These point counts consisted of an observer and a recorder noting the identity of all of the individuals that sang in a 15-minute period. In addition to the identities of the singing birds, their positions and movements were plotted on a Cartesian plane to guard against counting the same individual more than once.

One transect was censused per day and all song censuses were conducted between 05:00 and 08:30 (PDT). All of the transects were censused at least twice during each breeding season (i.e., between June 10 and July 20) to detect any seasonal changes in the community. The timing of the first census was determined by the disappearance of the snowpack and the arrival of the latest of the migrant songbird species.

Data Analyses

Indices of the richness, diversity, evenness, and dominance of the species in the community were calculated using standard equations (Wiens 1989).

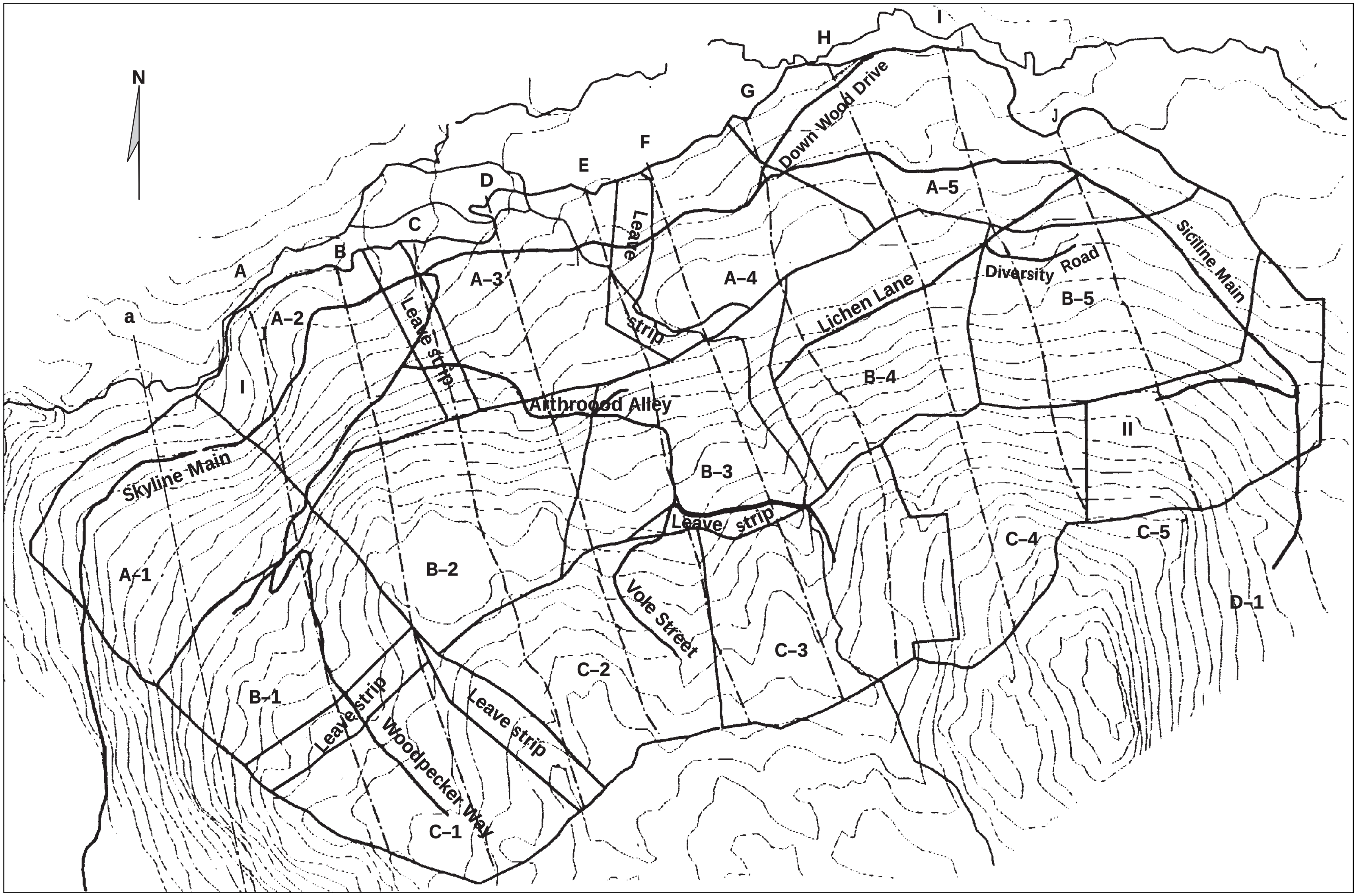

FIGURE 2. *Sicamous Creek research project layout and census line locations (an additional line lies off the map to the east).*

Difference in these measures between years was tested using a Wilcoxon test (Zar 1984).

Because the songs of different songbird species are audible at different distances in forests, an Effective Detection Distance (EDD) was calculated for each species using a standard technique (Wolf et al. 1995; and see Appendix 2). These detection distances were then used to calculate the actual area sampled at each census point for each species. For some analyses, densities (e.g., numbers per 40 ha) were determined for each species using these calculated, species-specific detection areas. In other analyses, the actual number of individuals counted within these detection areas was determined at each census point. One-way analyses of variance (Zar 1984) were conducted to examine the changes in the densities and abundance across the different years of the study.

We tested whether the Sicamous study site represented a homogeneous array of habitats during three different periods:

1. before road construction and harvesting (1992–93);
2. after the main access roads had been constructed (1994); and
3. after the experimental harvesting had taken place (1995).

The uniformity of each species' distribution in the original forest was determined using two-way analyses of variance (Zar 1984) with transect lines representing treatments and the seven census points on each line representing blocks. The dispersion of individuals in the forest was determined by comparing the mean and variances of each species' count at each census point.

RESULTS

Community Composition

Table 1 illustrates the changes in four measures of community composition across the four years of the study. Between 1992 and 1993, the number of species detected by our censuses increased from 14 to 18 species; 10 species (hereafter referred to as "core" species) made up more than 90% of the

TABLE 1 *Year-to-year changes in the songbird community statistics at Sicamous*

Statistic	Original forest 1992	Original forest 1993	Road/pre-harvest 1994	Post-harvest 1995
Species richness (S)	14.0	18.0	18.0	18.0
Species diversity ($H' = \Sigma\, p_i \log p_i$)	0.998	1.006	1.020	1.055
Species evenness ($J' = H'/H\max$)	0.874	0.805	0.812	0.841
Dominance ($\lambda = \Sigma\, p_i^2$)	0.117	0.113	0.110	0.095

community. This increase is partly explained by the addition of several habitats—including a variety of natural openings—when points were added to the sampling grid in 1993. Since then the species richness has stabilized at 18 species.

The three other community statistics reported in Table 1 (each of which takes into account the relative abundance of individuals as well as the number of species) all changed over the four years of the study. Table 2 indicates that the source of these changes lies in decreases in the relative abundance of several common species from 1993 to 1995, and increases in the abundance of several less common species over that same time interval. Among the most obvious changes were declines in the relative abundance of varied thrush and golden-crowned kinglets and increases in the relative abundance of hermit thrush and American

TABLE 2 *Changes in the relative abundance of songbird species in the Sicamous Forest*

Species	**1992**	**1993**	**1994**	**1995**
Winter wren (*Troglodytes troglodytes*)	.178	.118	.153	.145
Varied thrush (*Ixoreus naevius*)	.175	.167	.164	.101
Golden-crowned kinglet (*Regulus satrapa*)	.129	.130	.094	.094
Yellow-rumped warbler (*Dendroica coronata*)	.103	.091	.097	.099
Red-breasted nuthatch (*Sitta canadensis*)	.092	.093	.098	.095
Dark-eyed junco (*Junco hyemalis*)	.089	.143	.138	.132
Gray jay (*Perisoreus canadensis*)	.060	.041	.048	.033
Boreal chickadee (*Parus hudsonicus*)	0.60	.031	.029	.028
Hermit thrush (*Catharus guttatus*)	.050	.112	.091	.154
Pine grosbeak (*Pinicola enucleator*)	.030	.039	.040	.043
Wilson's warbler (*Wilsonia pusilla*)	.020	.012	.008	.029
Swainson's thrush (*Catharus ustulatus*)	.010	.002	.001	.000
Brown creeper (*Certhia americana*)	.003	.010	.008	.008
Fox sparrow (*Passerella iliaca*)	.003	.000	.000	.000
American robin (*Turdus migratorius*)	.000	.002	.001	.010
Lincoln's sparrow (*Melospiza lincolnii*)	.000	.002	.000	.008
Ruby-crowned kinglet (*Regulus calendula*)	.000	.002	.000	.001
Olive-sided flycatcher (*Contopus borealis*)	.000	.002	.002	.000
Townsend's warbler (*Dendroica townsendi*)	.000	.002	.000	.007
Chipping sparrow (*Spizella passerina*)	.000	.000	.000	.003
Mountain chickadee (*Parus gambeli*)	.000	.000	.020	.001
White-crowned sparrow (*Zonotrichia leucophrys*)	.000	.000	.000	.001

robins. During this period, access roads were constructed (1994) and harvesting activities were completed (1995). Nevertheless, the differences in the rank order of the top 10 species (which make up the core of the community) were not significantly large (Wilcoxon test, $p > 0.10$).

Changes in Species Densities and Distributions

Of the 10 species that made up the core of the community at Sicamous, six were abundant enough to calculate reasonable estimates of their density at each census point. These data permitted an evaluation of the spatial dispersion of individuals through the stand and the changes in density during disturbances associated with harvesting. Table 3 shows the results of two-way analyses of variance performed to determine whether systematic differences existed in the densities of these species along transect lines or among the rows of census points aligned up the hill (i.e., from the northwest to the southeast border of the area). No differences were found for any of the six species during either of the pre-harvest periods. These results suggest that the original stand provided a homogeneous array of habitats for the birds. Following road construction and harvesting activities, some parts of the study area had higher densities of hermit thrush than did others. The distributions of dark-eyed juncos and yellow-rumped warblers also became significantly more heterogeneous in 1995, after opening sizes had been manipulated.

Table 4 examines the changes in the actual densities of core species through the course of the study. The density of several of these species remained remarkably similar throughout the four years of monitoring; dark-eyed juncos, yellow-rumped warbler, and winter wren had similar densities before and after development activities. The densities of golden-crowned kinglets and varied thrush declined over the course of the four

TABLE 3 *The results of two-way analyses of variance to examine the homogenity of distribution for six core species; the tabulated data are the* F*-value and the probability* p *(in parentheses).*

Species	**Original forest 1992**	**Original forest 1993**	**Roads/pre-harvest 1994**	**Post-harvest 1995**
Dark-eyed junco	0.23 (> 0.90)	0.63 (> 0.70)	0.33 (> 0.90)	2.90 (< 0.01)
Varied thrush	0.33 (> 0.80)	1.56 (> 0.40)	1.86 (> 0.10)	1.29 (> 0.25)
Hermit thrush	0.95 (> 0.44)	1.36 (> 0.80)	2.56 (< 0.02)	1.96 (< 0.10; > 0.50)
Yellow-rumped warbler	1.17 (> 0.30)	0.39 (>0.85)	0.71 (>0.60)	2.08 (< 0.10: > 0.05)
Golden-crowned kinglet	1.23 (> 0.30)	1.37 (> 0.20)	1.21 (> 0.30)	0.17 (> 0.95)
Winter wren	1.24 (> 0.30)	1.25 (> 0.25)	2 (<0.10; > 0.05)	0.69 (> 0.65)

TABLE 4 *Changes in the density of six key songbird species at Sicamous Creek since 1992. The tabulated data are the mean number of individuals (and variance) per census point.*

Species	**Original forest 1992**	**Original forest 1993**	**Roads/pre-harvest 1994**	**Post-harvest 1995**
Dark-eyed junco	0.80 (0.36)	1.00 (0.53)	.084 (0.43)	0.82 (0.71)
Varied thrush[a]	1.44 (0.72)	1.44 (0.52)	1.18 (0.52)	0.70 (0.52)
Hermit thrush[a]	0.24 (0.22)	0.91 (0.72)	0.57 (0.43)	1.20 (1.12)
Yellow-rumped warbler	0.74 (0.54)	0.63 (0.39)	0.67 (0.32)	0.67 (0.60)
Golden-crowned kinglet	0.86 (0.40)	0.84 (0.40)	0.77 (0.39)	0.68 (0.60)
Winter wren	1.24 (0.33)	1.01 (0.46)	1.34 (0.57)	1.25 (0.54)

[a] One-way analysis of variance, $p < 0.001$.

years, while the densities of hermit thrush increased. In the case of varied and hermit thrush, the changes were significantly large.

The ratio of the mean number of individuals in a sample to its variance is one way to determine spatial patterns of dispersion. This measure, which is apparent in Table 4, shows that in the original forest individuals of each of the six core species were more-or-less uniformly distributed throughout the stand (i.e., the means were greater than the variances). Following harvesting, the dispersion of each of these species changed; for several species the dispersion pattern is much *less* uniform (i.e., the differences between the means and variances becomes smaller).

DISCUSSION

Although our findings must still be viewed as preliminary, they have already provided several insights into the nature of the community of songbirds that breeds in this forest type.

Bird Communities in High-elevation Forests

While few in number, the species that occupied the original forest at Sicamous were representative of those found in other high-elevation forests in the Kamloops Forest Region and in much of western North America (Hutto et al. 1993; Bryant 1995; Scheick et al. 1995). This community included species that appear to rely on interior forest habitats, as well as those that apparently adapt to a range of disturbances associated with timber harvesting.

Much work remains to be done to characterize the specific forest attributes required by these species. We have collected data on the nesting sites, foraging locations, and song posts used by the species that inhabited the original forest. For example, the varied thrush is one species that appears to rely on an interior forest habitat; we have only recorded them nesting in suppressed subalpine fir (*Abies lasiocarpa*), which are abundant in the understorey of mature stands (see also Cannings et al. 1987). Our goal is to identify similar features of the forest for each of the core species and then relate changes in the abundance of birds to changes in the resource base and predation rates brought about by harvesting (Small and Hunter 1988). As collaborators in this project continue to provide us with more data on the structure of this forest, we will be better able to identify what it was about the habitats in this forest that determined the distribution of the species we found there.

Density and Distribution of Core Species Before Harvesting

Fortunately, we had several seasons of pre-harvest inventory to provide a baseline for comparing the effects of creating different opening sizes in this forest. Most studies have had to rely on retrospective analyses of communities that may not have been equivalent before harvesting (Crawford et al. 1981; Scheick et al. 1995). The relative consistency in the abundance of different species at Sicamous before harvesting suggests that the populations of breeding songbirds were comparably stable. In addition, the uniform dispersion of birds in the original stand suggests that those breeding opportunities were relatively homogeneous before the experiment. These insights give us a much greater confidence that post-harvesting changes actually reflect the effects of the different experimental treatments.

Initial Responses to Manipulations of Opening Size

Few studies have been able to document the early development of forest fragmentation effects on songbirds (e.g., Hagan et al. 1996). Our results suggest that the initial response of the community of songbirds breeding at Sicamous was not dramatic. Most of the original species have remained in the remnants of the original forest and in selectively harvested stands. Although the densities of several species have changed following harvesting, the initial response was smaller than we might have expected from our retrospective study. In most instances, it appears that the species accommodated the disturbance of the stand. This may be because adjacent patches of forest absorbed the displaced territory-owners who returned from migration to their former habitats (Darveau et al. 1995).

What will determine the long-term effects of manipulating opening sizes in the stand? Research in other forest types is beginning to indicate that the pairing success and the nesting productivity of individuals will dictate whether they persist in disturbed forest remnants (Hagan et al. 1996). Our future work at Sicamous will examine this aspect of community dynamics. By examining the productivity of core species, we may be able to determine whether the changes we observe arise from lost resources or different competitive relationships among similar species (e.g., varied thrush and hermit thrush).

Is the Sicamous Study Site a Stand or a Landscape?

The experiment at Sicamous represents an opportunity to examine the processes that shape songbird communities at two different scales. Because the sizes of the treatment blocks are relatively large, we will be able to determine how the change in resources available in a particular place affects the species of birds that breed there. These are stand-level questions. As such, they may allow us to comment on which silviculture prescriptions are “best” for a particular species of songbird at a particular site (cf., Martin 1992). In addition, because we can compare the community of songbirds which occupy the remnants of original forest with what was there before, we may be able to identify the indirect effects of harvesting on an entire development area. This also addresses a stand-level question, but at a larger scale than that which is often studied. The spatial scale of this study will allow us to examine the spill-over effects of harvesting on remnants of the original forests and the interaction of different harvesting effects in forest fragments (Crawford et al. 1981; Freemark and Collins 1992). Both of these are important factors influencing landscapes.

To address questions at a very large scale—a true landscape scale—the effects of harvesting at Sicamous must be compared with other equivalent sites. Many of the same harvesting practices have been implemented elsewhere in the Kamloops Forest Region to accommodate the habitat needs of mountain caribou (*Rangifer tarandus*). Another of our future goals will be to compare our findings from Sicamous with data collected in these other ESSF forests.

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APPENDIX 1. Common and scientific names of songbird species

Common name	Scientific name
Olive-sided flycatcher	*Contopus borealis*
Gray jay	*Perisoreus canadensis*
Mountain chickadee	*Parus gambeli*
Boreal chickadee	*Parus hudsonicus*
Brown creeper	*Certhia americana*
Red-breasted nuthatch	*Sitta canadensis*
Winter wren	*Troglodytes troglodytes*
American robin	*Turdus migratorius*
Varied thrush	*Ixoreus naevius*
Hermit thrush	*Catharus guttatus*
Swainson’s thrush	*Catharus ustulatus*
Golden-crowned kinglet	*Regulus satrapa*
Ruby-crowned kinglet	*Regulus calendula*
Yellow-rumped warbler	*Dendroica coronata*
Wilson’s warbler	*Wilsonia pusilla*
Townsend’s warbler	*Dendroica townsendi*
Orange-crowned warbler	*Vermivora celata*
McGillivray’s warbler	*Oporornis tolmiei*
Savannah sparrow	*Passerculus sandwichensis*
Chipping sparrow	*Spizella passerina*
Fox sparrow	*Passerella iliaca*
White-crowned sparrow	*Zonotrichia leucophrys*
Dark-eyed junco	*Junco hyemalis*
Lincoln’s sparrow	*Melospiza lincolnii*
White-winged crossbill	*Loxia leucoptera*
Pine siskin	*Carduelis pinus*
Pine grosbeak	*Pinicola enucleator*

There have been numerous approaches to the problem of monitoring the abundance and diversity of birds in different habitats (Ralph and Scott 1981; Ralph et al. 1995a). Indeed, no one standard methodology that is suitable for all situations. The reality is that the "best" monitoring method for any research problem will represent a compromise between the economy of collection effort and the precision and accuracy of the estimates produced. It thus becomes the responsibility of each investigator to determine what, precisely, is being estimated by the methods employed (Johnston 1995). In this appendix, we analyze the effectiveness of the census methods we have used to study the bird community breeding in high-elevation forests.

As described in the main body of this paper, our methods have involved establishing census stations along transect lines, in which census points are separated by at least 250 m. Morning song censuses are conducted between 05:00 (the approximate time of sunrise in June and July) and 08:30 (when a noticeable decline in the activity of all species occurs). A typical census begins with a two-minute silent period after an observer and recorder arrive at a station. During the subsequent 15 minutes, the identity and location of all singing individuals are then noted on a map that includes a set of Cartesian co-ordinates. Records are kept both of the one-minute interval in which a bird was first detected and any movements during the census that could result in an individual being counted once. By many standards, these censuses are long; we conducted long censuses to determine their accuracy in detecting individuals and species.

Figure A2.1 illustrates the rate at which new individuals were added to censuses in the 1993 breeding season at Sicamous Creek. (A similar curve describes the rate at which individuals are added to censuses.) As several authors have noted (e.g., Dawson et al. 1995; Savard and Hooper 1995), both the number of individuals and the species accumulate during these censuses and begin to reach an asymptote at about 12 minutes. Perhaps more revealing than the proportion of species detected by the censuses is the variation that exists among censuses with respect to this measure. As illustrated in this figure, the among-census variation decreases substantially only after 10 minutes of monitoring. We have continued to conduct 15-minute censuses in this study to reduce the point-to-point variation in estimates as much as possible.

Another important problem in censusing whole communities of birds is that different species' songs are audible at different distance in forest habitats (Reynolds et al. 1980; Wolf et al. 1995). One strategy to alleviate this problem is to fix a radius within which most species' songs are detectable and count all of the birds that occur within that area. These are referred to as "fixed radius censuses." An alternative technique, referred to as an "unlimited distance census," records the actual position of the singers, as well as their identities. Subsequent analysis determines how likely it would be to detect an individual of a particular species in concentric 10-m rings around the census point. By convention, when two successive rings record fewer birds than the previous ones, the effective

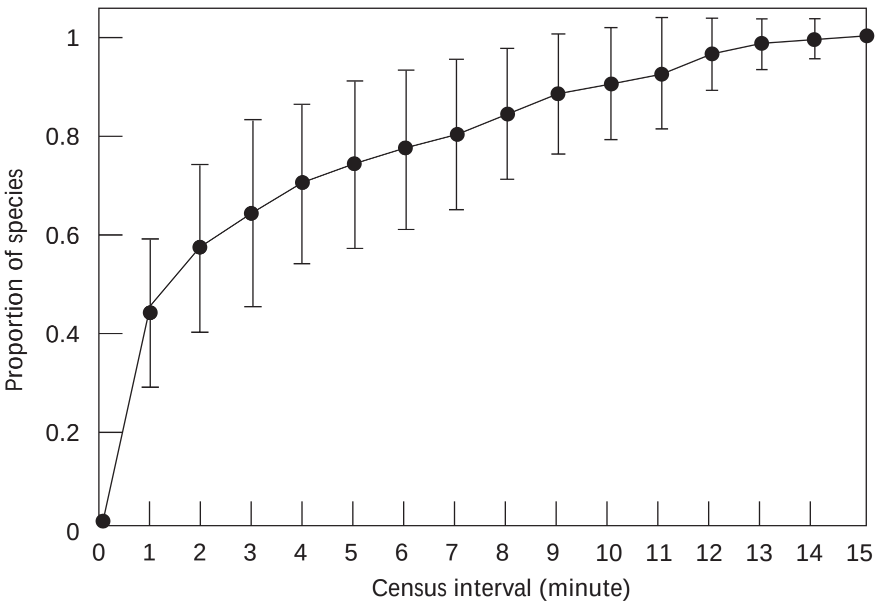

FIGURE A2.1 *The average rate of accumulation for each minute of 15-minute censuses. Error bars show the standard deviations among censuses for each one-minute interval.*

detection distance (EDD) has been reached. The EDD can then be used to determine which area was actually sampled for a particular species and how many birds occurred within that area.

Table A2.1 illustrates the effect of treating data on the abundance of the six core species in essf forests as if they were collected using these two sampling strategies. In three cases—winter wren, hermit thrush, and dark-eyed juncos—it makes little difference if 100 m, 50 m, or unlimited distance plots had been used. For one species, golden-crowned kinglets, fixing a radius of a circle at 100 m systematically misses birds, likely because their songs do not transmit far. For other species, such as varied thrush, 50 m radius plots seem to underestimate densities. With these species, some individuals seem inhibited from singing by the presence of an observer. This is indicated by lower average density measures and higher variances.

To limit extraneous sources of error in our density estimates, we have chosen to use unlimited-distance censuses as our method for censusing the birds in essf forests. Others have also argued in favour of this method (e.g., Thompson and Schwalbach 1995) and recently it has become recommended as a standard for monitoring programs (Ralph et al. 1995b).

Several authors have questioned the precision of density estimates based on song censuses (Verner 1985). We examined this question by comparing estimates based on our unlimited distance censuses with those produced by mapping the territories of three of the core species in the community. To map the territories of these birds, we played songs back to them as we moved around a large (>1 ha) previously marked 20 × 20-m

TABLE A2.1 *Estimated density (mean no. per 40 ha, +/- SE) for six songbird species at Sicamous*

Species	**Effective detection distance (m)**	**Estimated density**		
		Unlimited distance plots	**Fixed 40 m radius plots**	**Fixed 100 m radius plots**
Dark-eyed junco	90	14.51 (1.13)	12.87 (2.59)	13.57 (0.97)
Golden-crowned kinglet	40	57.52 (5.45)	48.69 (3.37)	14.13 (0.88)
Hermit thrush	80	12.68 (1.51)	12.31 (2.56)	12.03 (1.11)
Varied thrush	100	14.83 (0.98)	9.51 (2.17)	14.83 (0.98)
Yellow-rumped warbler	80	11.81 (1.32)	13.99 (2.62)	7.96 (0.86)
Winter wren	80	13.99 (1.30)	13.99 (2.40)	12.31 (0.86)

grid. By noting the points on the grid where the territory-owner ceased responding to playback, we determined the size of the area he defended.

Table A2.2 shows the estimated densities of winter wrens, golden-crowned kinglets, and hermit thrushes based on song censuses and on territory mapping. Although the number of mapped territories is relatively small, there is a good correspondence between the two estimates; none of the differences are significantly large (*t*-tests; all *p* values >0.05). The variability in both methods of estimating densities is, however, quite large. Thus, we concur with the conclusions of other workers (e.g., Hamel 1984; DeSante 1986) that the estimates of densities produced by song censuses are best used as relative measures, rather than absolute quantities.

TABLE A2.2 *A comparison of densities (mean no. per 40 ha) using song censuses and territory mapping (standard error shown in parentheses)*

Species	**Density**	
	Based on territory mapping	**Based on song censuses**
Winter wren (*n* = 4)	19.94 (5.90)	13.99 (1.30)
Golden-crowned kinglet (*n* = 4)	48.21 (6.52)	57.52 (5.43)
Hermit thrush (*n* = 3)	9.7 (6.20)	12.68 (1.51)

In conclusion, our long-duration, unlimited distance, song censuses appear to produce relatively accurate and fairly precise estimates of the abundance and diversity of birds breeding in essf forests. One reason why the technique may work so well in these forests is that the songbird community is rather small and it is unlikely that observers make identification errors. Another source of consistency in this study has been the small number of field workers that have been responsible for making observations; more than three-quarters of the censuses conducted in this project have been carried out by one individual as the observer.

The Use of Artificial Substrates to Measure Effects of Timber Harvesting and Road Construction on Stream Invertebrates in the Sicamous Creek Watershed: Preliminary Report

BRIAN HEISE

ABSTRACT

The effects of clearcut logging and road construction on stream invertebrate communities were investigated in three creeks in the Sicamous Creek watershed of British Columbia. Artificial substrate wire baskets containing gravel were placed in clearcut, road-silt impacted, and control streams, and retrieved after a colonization period. Preliminary analysis of a set of baskets from a clearcut and control stream showed that the harvested stream had significantly fewer total organisms, stoneflies, and flatworms than the control stream. The invertebrate community from both sites was dominated by Diptera, mostly in the family Chironomidae. The results are discussed in terms of potential implications for stream riparian management in British Columbia.

INTRODUCTION

Streams are important components of forested ecosystems. They provide high-quality water for domestic use, irrigation, recreation, and fisheries. British Columbia's Forest Practices Code sets out strict guidelines for harvesting procedures aimed at preserving biodiversity and providing sustainability of these forest resources.

One component of these guidelines is the establishment of riparian management areas around forested streams (British Columbia Ministry of Forests 1995; Stevens et al. 1995). The riparian guidelines for streams on the coast differ from those for the interior of the province. There is a need for studies on the interactions between forest operations and streams in the interior of the province to confirm the effectiveness of the interior guidelines. To date the most complete research on the effects of forest harvesting has taken place in the Coastal Western Hemlock biogeoclimatic zone, notably in the Carnation Creek Watershed Experiment (see Hartman and Scrivener 1990). One of the areas requiring research is the wet, high-altitude interior of British Columbia, where streams may react to forest harvesting differently because of differences in logging methods, climate, runoff timing, soil types, and forest cover. As well, more research is needed on the effects of harvesting practices on small (< 1 m wide) headwater streams. Forest harvesting around these streams is not as tightly controlled as for larger streams, partly because of reduced fisheries values. However, headwater streams provide fine particulate organic matter (FPOM) and cool water to downstream, larger rivers (Vannote et al. 1980), and these larger rivers may contain salmonids. Information on the effects of harvesting practices on these streams will be helpful to select the best management options for interior forests.

The removal of the forest canopy by clearcutting can cause dramatic changes to the physical environment of streams. Clearcutting reduces the amount of coarse woody debris entering the stream, resulting in fewer debris dams and greater movement of sediments (Bisson et al. 1987; Hartman et al. 1987; Webster et al. 1992; Davies and Nelson 1994). In many cases, this sediment is in the form of silt, which can smother the natural stream substrate (Beschta 1978; Culp and Davies 1983; Webster et al. 1983; Everest et al. 1987; Vuori and Joensuu 1996). The loss of a riparian canopy results in an increase in light penetration to the stream, and an increase in stream temperatures (Brown and Krygier 1970; Webster et al. 1983; Noel et al. 1986; Beschta et al. 1987; Davies and Nelson 1994).

In addition to the physical changes to streams, clearcutting also causes changes in the invertebrate food supply, functional feeding groups, and community structure. The loss of the riparian canopy causes a major shift in the type of food available to macroinvertebrates in these streams. Invertebrates in forested headwater streams depend heavily on the leaves of those forests as a source of energy (Minshall 1967). This allochthonous food supply is reduced after clearcutting, as less leaf litter (coarse particulate organic matter: CPOM) reaches the stream (Culp and Davies 1983; Webster et al. 1983; Webster et al. 1990; Stout et al. 1993). The abundance of algae increases dramatically in clearcut streams with the lack of forest cover and increased light (Murphy and Hall 1981; Murphy et al. 1981; Webster et al. 1983; Noel et al. 1986; Wallace and Gurtz 1986; Davies and Nelson 1994). However, this pattern may not develop if the stream is phosphorus limited (Culp and Davies 1983). If it does develop, it may be only seasonal in nature (Reed et al. 1994).

After clearcutting, a shift in food resources from leaf litter to algae should produce a change in the dominant functional feeding groups of invertebrates from shredders (organisms that feed on the dead leaves and colonized micro-organisms) to scrapers (organisms that scrape algae from hard surfaces). This predicted pattern has been verified for some streams (Webster et al. 1983; Wallace and Gurtz 1986; Reed et al. 1994). In other studies there was no significant difference between shredder proportions in logged and unlogged streams (Hawkins et al. 1982; Culp and Davies 1983). In one study undertaken in a hardwood forest, shredder production actually increased after clearcutting because of the presence of leaves of early successional species, which were faster for the shredders to process (Stout et al. 1993).

There have been conflicting reports on the effects of clearcutting on the abundance and diversity of macroinvertebrate communities. Densities of invertebrates in clearcut streams have been both lower (Culp and Davies 1983; Davies and Nelson 1994; Vuori and Joensuu 1996), and higher (Murphy et al. 1981; Silsbee and Larson 1983; Noel et al. 1986) than in control streams. In two studies where the abundance of invertebrates was higher in the clearcut stream, invertebrate diversity was lower or the same in these sites (Newbold et al. 1980; Carlson et al. 1990).

My study on the effects of logging practices on stream invertebrate community structure in a forest in British Columbia's interior is part of the Sicamous Creek Silvicultural Systems Project. The research site is a high-elevation forest of Engelmann spruce and subalpine fir in the ESSFwc2 biogeoclimatic subzone (Lloyd et al. 1990). Details regarding project design and the various research projects being undertaken can be found in Vyse (see this proceedings, page 4). This study of stream invertebrates is the only research being done on the aquatic components of the forest ecosystem at Sicamous Creek. This research integrates well with other studies of the birds and mammals that use the riparian area of these streams.

The two main objectives of my study are:

1. to document the types and life histories of invertebrates living in these ESSF streams, as little is known of their basic biology and distribution; and
2. to compare the aquatic invertebrate communities in clearcut, road-silt impacted, and control streams running through the Sicamous Creek research area.

Stream communities were compared by placing artificial substrate samplers in all creeks, and retrieving the baskets after a colonization period.

The purpose of this paper is to describe the research methods, report some preliminary results, and discuss research needs.

METHODS

Study Site

The Sicamous Creek Silvicultural Systems Project site is located north of Mount. Mara, near the town of Sicamous, B.C. For a detailed description of the project site see Vyse (see this proceedings, page 7). The individual creeks monitored are all first- or second-order headwater tributaries of the north fork of Sicamous Creek.

Three streams running through the project area were selected for study (Figure 1). Two of the streams (D Creek and E' Creek) ran through clearcuts. A third creek (E Creek) had an upstream section which was unlogged and that acted as a control and a downstream section which ran beside a new logging road and that was sampled for silt effects. All three streams were within 900 m of one another, in treatment areas C2 and

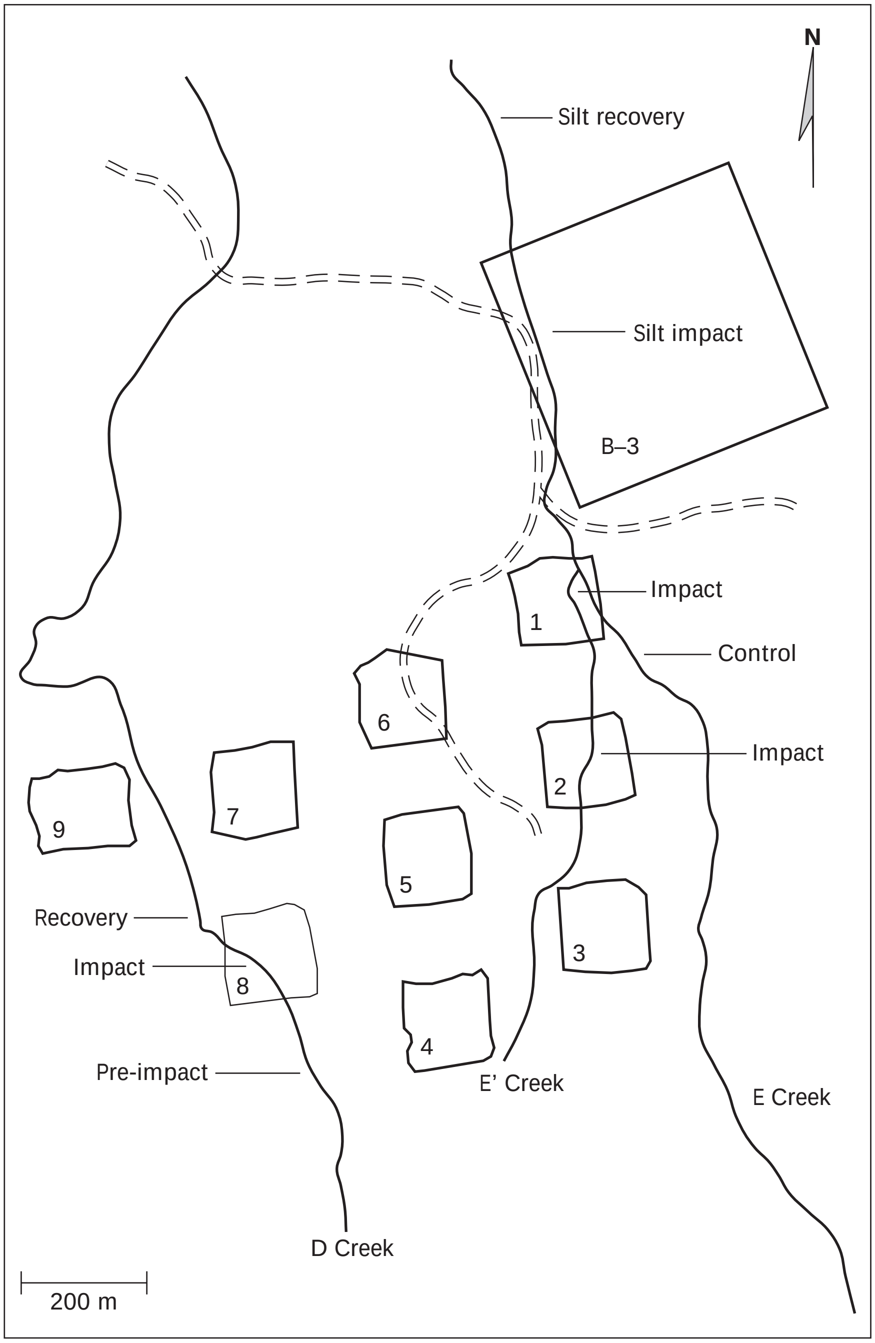

FIGURE 1 *Map of the study site showing the three study creeks and the location of treatments. The direction of flow for all streams is from south to north. Squares marked 1 through 9 represent 1-ha cutblocks. The square labelled "B-3" is a 10-ha selective cut. The dashed lines represent gravel logging roads.*

B3. In all streams invertebrate sampling was done using artificial substrate wire baskets, filled with gravel (these samplers are described in detail later). In D Creek, the baskets were placed in three sections. These sections were a pre-impact zone upstream of cutblock 8, an impact zone in cutblock 8, and a recovery zone downstream of cutblock 8. In E' Creek, baskets were placed in cutblocks 1 and 2. Upper E Creek served as a control stream. Baskets were placed in it adjacent to the cutblocks of E' Creek. Baskets were also placed in an area of E Creek below and beside a logging road. This stretch of stream appeared to be heavily effected by sand and silt which had eroded from the road, and so was considered a silt impact section (Figure 1). Note that this section was also in a selectively cut block. Baskets were placed in a section of this stream downstream of the silty area to examine recovery of the invertebrates from the effects of silt.

Twenty baskets were placed in each section in all creeks. Within each section, baskets were placed in a stretch less than 80 m in length. Baskets were placed in the stream in September 1995 (the first-year post-harvest) and retrieved in the first week of July 1996. This retrieval date was the earliest date when all of the streams were free of snow. Retrieved baskets were placed in sealed plastic containers with 70% ethanol. A new set of baskets was placed in each stream section following basket retrieval to monitor changes in the stream communities over time. This monitoring will continue for four years.

In addition to the artificial substrate samples, qualitative kick samples of invertebrates were taken each month in representative stream sections. Organisms were preserved in Kahle's solution. The kick samples allow for analysis of invertebrate life histories, the collection of relatively undamaged specimens for identification, and provide an indication of the selectivity of the baskets for certain taxa. During these monthly sampling trips, adult insects were collected using sweep-netting of riparian vegetation and were preserved in Kahle's solution. This helped with the identification because species keys are available only for the adult stage of many aquatic insects.

Artificial Samplers

Preliminary sampling using conventional Surber and Hess samplers indicated that these methods would not effectively sample the invertebrates in these streams, and would not allow unbiased comparisons between streams. To compare the aquatic invertebrate communities in streams running through harvested areas with those in non-harvested areas, artificial substrate samplers were placed in the streams. These artificial substrates were baskets made of 1-cm mesh hardware cloth, filled with 2 cm diameter gravel (Figure 2). Each basket measured 12.5 × 12.5 × 5 cm and was labelled with numbered Dymo© and flagging tape. Baskets were buried with the top of the basket flush with the natural substrate. Baskets were placed in pairs in locations of similar depth and velocity.

Analysis of Invertebrate Communities

In the laboratory, invertebrates were separated from the substrate material by eye under a magnifying lens for coarse material and under the low power of a dissecting microscope for fine material. The invertebrate communities found on the baskets will be analyzed for density, total diversity, EPT (Ephemeroptera + Plecoptera + Trichoptera) taxa richness

FIGURE 2 *Photograph of the artificial substrate baskets used in the study.*

(Wallace et al. 1996), indicator species, functional feeding groups, and indices of stream health. Life history information will also be analyzed.

For this preliminary paper, I compared the number and types of organisms that had colonized nine baskets in each of block 1 (E' Creek) and the control stream (E Creek). Means were compared using the Mann-Whitney U-test. This procedure does not require the assumption of normality, which is difficult to obtain with small sample sizes.

RESULTS

The baskets retrieved from the harvested stream contained significantly fewer organisms than those from the control stream (Figure 3, $p = 0.0118$). The communities in both streams were dominated by larval dipterans (mostly Chironomidae) (Figure 4). The clearcut stream had significantly fewer stoneflies ($p = 0.0003$), flatworms ($p = 0.0052$), and fewer diptera, though this latter difference was of marginal significance ($p = 0.0576$).

DISCUSSION

The finding of significantly lower abundance of organisms in the clearcut stream (E' Creek) is not surprising, given that logging activity in this stream modified the habitat so drastically. E' Creek was heavily affected by the addition of slash (tree branches, needles, and bark), such that no riffle-run-pool sequence was present. Instead, the creek flowed through

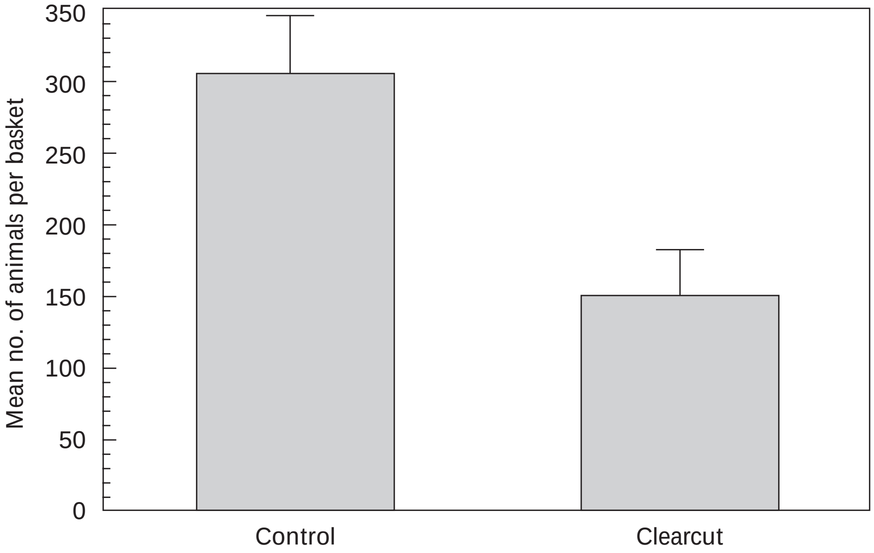

FIGURE 3 *Mean number of invertebrates in baskets recovered from control (E Creek) and clearcut block 1 (E' Creek) areas of the study. Error bars represent 1 SE.*

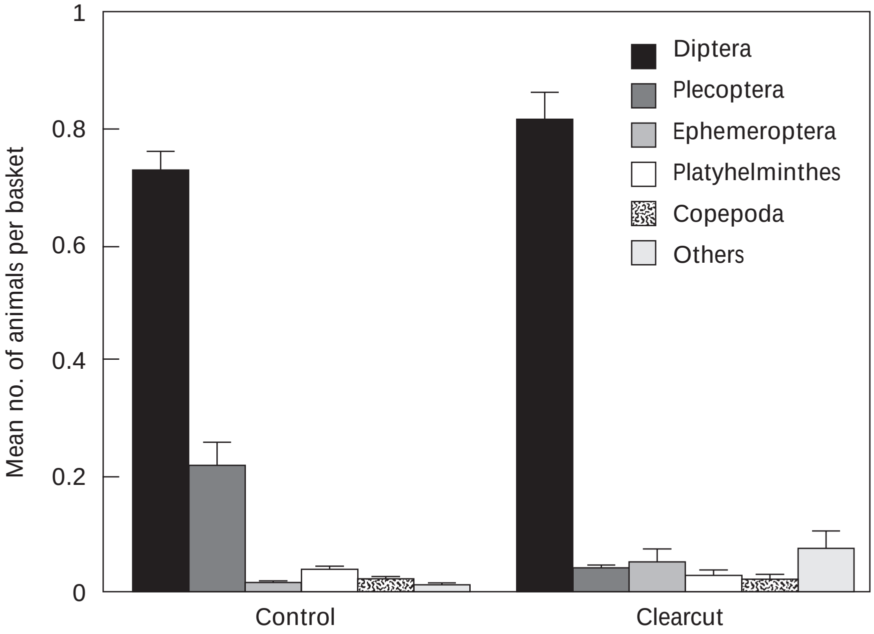

FIGURE 4 *Mean number of major invertebrate groups in baskets recovered from control (E Creek) and clearcut block 1 (E' Creek) areas of the study. Error bars represent 1 SE.*

the cutblock in one continuous pool, often disappearing from sight under the slash. The substrate was dominated by fine (< 1 mm diameter) organic material, and needles and wood chips. The natural gravel/rock substrate was occasionally observed where the organic layer had been eroded away. In contrast, E Creek had discernible riffles, runs, and pools, with the substrate varying accordingly. In E Creek, the mineral substrate was never completely covered by fine organic matter.

An interesting preliminary result of this study is the lower abundance of stoneflies in the clearcut stream compared to the control. Other studies have found both a decrease (Webster et al. 1983; Davies and Nelson 1994; Vuori and Joensuu 1996) and an increase (Newbold et al. 1980; Stout et al. 1993) in the number of stoneflies in harvested streams. Most of the stoneflies at Sicamous appear to be shredders, and so should have benefited from the increased levels of coarse particulate organic matter, in the form of slash, available in the clearcut. However, the clearcut stream also contained abundant silt, and probably lower oxygen levels, due to the breakdown of the organic material present. The number of stoneflies in E' Creek may therefore represent a trade-off between having a rich source of coarse particulate organic matter for feeding and having high silt, and possibly low oxygen conditions, in the clearcut stream.

Various sampling techniques have been used to quantify invertebrate numbers. Although artificial substrate samplers are commonly used in environmental impact studies (Buikema and Voshell 1993), my use of artificial substrate baskets to quantify the effects of logging activities on stream invertebrates is unusual. Most researchers have used Surber or Hess samplers, but these samplers are limited because they are specific for certain substrates. I used artificial substrates because they reduce variability by providing a standard sampling unit (Rosenberg and Resh 1982). This readily allows different streams to be compared. One possible disadvantage of this method is that the baskets may be selected for or against by particular taxa (Buikema and Voshell 1993). Silsbee and Larson (1983) examined logging effects using wire basket artificial substrates and also took Surber samples for comparison to the baskets. While these authors found that basket and Surber samples were different in species composition and invertebrate density, the two methods detected the same pattern of disturbance. The consistent use of artificial substrate baskets in this study at Sicamous Creek will allow direct comparisons between the various treatments, but may limit comparisons with other studies. The monthly kick samples, though only qualitative, should indicate biases in the taxa collected by the basket samplers.

The literature on logging effects includes examples of cases in which stream invertebrate populations both increased and decreased (Murphy and Hall 1981; Culp and Davies 1983; Silsbee and Larson 1983; Noel et al. 1986; Davies and Nelson 1994; Vuori and Joensuu 1996). It is difficult to make direct comparisons with these studies, as the studies varied in factors such as forest type (e.g. hardwood vs. softwood), biogeoclimatic zone, and sampling methods (Surber, single stone, artificial substrates). As well, study design varied in that some studies compared pre- and post-impact effects on a few streams, while others examined only post-impact effects on several streams. Amongst the latter group, the length of time between harvesting and the study varied greatly, and in some cases was as great as 25 years. The two basic study designs are similar in that they both monitor community changes following a single disturbance (in this case, logging) and so could be considered "pulse" disturbance studies (Bender et al. 1984). Reed et al. (1994) suggest that species abundance techniques, which are commonly used to detect logging effects, may be the most effective method of detecting "pulse" disturbances in streams.

The amount of time elapsed between harvesting and stream sampling is critical, as streams change over time, and the rate of change varies for different systems. Streams can take from months to years to recover from the effects of silt and sand (Webster et al. 1983; Yount and Niemi 1990). The dominant food source and invertebrate functional feeding groups can also change with regrowth of the riparian canopy. For example, Wallace and Gurtz (1986) found that both primary and secondary production of the scraper *Baetis* initially increased with logging, but declined four and five years later. Studies do not usually run long enough to monitor complete recovery of logged streams, as this process is estimated to take from 25 to 300 years (Hedin et al. 1988; Webster et al. 1992).

Research into the effects of clearcut logging on stream communities is needed in specific regions, and management decisions made regarding harvesting and riparian preservation should reflect these regional differences. The question of what constitutes "regional" remains to be determined. More research is needed to determine if one set of standards will suffice for the "interior" of the province versus the "coast," or if guidelines should be done on a site-series basis (e.g., is a wetter ESSF subzone stream affected in the same manner as a drier ESSF subzone stream, given equivalent harvesting guidelines?).

Clinnick (1985) suggests that the width of buffer strips to reduce sedimentation into streams depends on the local conditions of soil type and slope, and should extend up into the headwater streams within a watershed. Any effects of clearcutting found in this study of streams in the Sicamous Creek watershed may represent a worst-case scenario, as no buffer strips were left beside the streams, and slash was allowed to fill the stream channels. This research will continue for several years to assess recovery of the streams from the effects of logging. The results of this research will hopefully provide useful information for forest managers to aid in harvesting decisions.

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