Vegetation Development after Clearcutting and Site Preparation in the SBS Zone
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by
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ABSTRACT

The changes in floristic composition and structure that occur after clearcutting and site preparation in four ecosystems in the Sub-Boreal Spruce Zone are outlined. Responses of species to burning and mechanical site preparation are examined and the strategies for recolonization and survival used by different species are investigated. Implications of these findings to ecological classification and management to meet silvicultural and wildlife objectives are addressed.

ACKNOWLEDGEMENTS

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# TABLE OF CONTENTS

<table>
<thead>
<tr>
<th>Section</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>ABSTRACT</td>
<td></td>
<td>iii</td>
</tr>
<tr>
<td>ACKNOWLEDGEMENTS</td>
<td></td>
<td>iii</td>
</tr>
<tr>
<td>1 INTRODUCTION</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>1.1 Objectives</td>
<td></td>
<td>1</td>
</tr>
<tr>
<td>1.2 Literature Review</td>
<td></td>
<td>2</td>
</tr>
<tr>
<td>1.3 Study Area</td>
<td></td>
<td>2</td>
</tr>
<tr>
<td>1.4 Treatment History and Prescriptions</td>
<td></td>
<td>7</td>
</tr>
<tr>
<td>2 METHODS</td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>2.1 Field Sampling</td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>2.2 Analysis</td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>2.2.1 Classification</td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>2.2.2 Vegetation structure and composition</td>
<td></td>
<td>8</td>
</tr>
<tr>
<td>3 RESULTS AND DISCUSSION</td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>3.1 General Pattern of Revegetation After Site Preparation</td>
<td></td>
<td>10</td>
</tr>
<tr>
<td>3.2 Rate of Revegetation After Site Preparation</td>
<td></td>
<td>12</td>
</tr>
<tr>
<td>3.3 General Soil and Humus Layer Properties</td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>3.4 Ecosystem-Specific Changes After Site Preparation</td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>3.4.1 Queen's cup ecosystem</td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>3.4.2 Oak fern ecosystem</td>
<td></td>
<td>15</td>
</tr>
<tr>
<td>3.4.3 Devil's club ecosystem</td>
<td></td>
<td>16</td>
</tr>
<tr>
<td>3.4.4 Horsetail ecosystem</td>
<td></td>
<td>17</td>
</tr>
<tr>
<td>3.5 Species Responses to Site Preparation Treatments: revegetation dynamics and implications</td>
<td></td>
<td>17</td>
</tr>
<tr>
<td><em>Epilobium angustifolium</em> (fireweed)</td>
<td></td>
<td>17</td>
</tr>
<tr>
<td><em>Lonicera involucrata</em> (black twinberry)</td>
<td></td>
<td>18</td>
</tr>
<tr>
<td><em>Rubus parviflorus</em> (thimbleberry)</td>
<td></td>
<td>18</td>
</tr>
<tr>
<td><em>Rubus idaeus</em> (red raspberry)</td>
<td></td>
<td>19</td>
</tr>
<tr>
<td><em>Salix spp.</em> (willows)</td>
<td></td>
<td>19</td>
</tr>
<tr>
<td><em>Alnus incana ssp. tenuifolia</em> (mountain alder)</td>
<td></td>
<td>20</td>
</tr>
<tr>
<td><em>Populus tremuloides</em> (trembling aspen)</td>
<td></td>
<td>20</td>
</tr>
<tr>
<td><em>Ribes laxiflorum</em> (trailing black currant)</td>
<td></td>
<td>20</td>
</tr>
<tr>
<td><em>Ribes lacustre</em> (black gooseberry)</td>
<td></td>
<td>21</td>
</tr>
<tr>
<td><em>Vaccinium membranaceum</em> (black huckleberry)</td>
<td></td>
<td>21</td>
</tr>
<tr>
<td><em>Calamagrostis canadensis</em> (bluejoint)</td>
<td></td>
<td>21</td>
</tr>
<tr>
<td><em>Sambucus racemosa</em> (red elderberry)</td>
<td></td>
<td>22</td>
</tr>
<tr>
<td><em>Viburnum edule</em> (highbush-cranberry)</td>
<td></td>
<td>22</td>
</tr>
<tr>
<td>4 SUMMARY AND CONCLUSIONS</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>4.1 Classification Within the Existing Biogeoclimatic Ecosystem Classification System</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>4.2 Rate of Revegetation</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>4.3 Revegetation Strategies of Key Species</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>4.3.1 Establishment</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>4.3.2 Persistence and Longevity</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>4.4 Site Preparation Effects</td>
<td></td>
<td>23</td>
</tr>
<tr>
<td>5 LIMITATIONS OF THE STUDY</td>
<td></td>
<td>24</td>
</tr>
</tbody>
</table>
TABLE OF CONTENTS — Continued

6 RECOMMENDATIONS ........................................................................................................................................... 25
   6.1 Classification ............................................................................................................................................... 25
   6.2 Species Response to Treatment .................................................................................................................. 25
   6.3 Relationship Between Competing Vegetation and Crop Tree Performance ............................................. 25
   6.4 Research Approaches .................................................................................................................................. 25

7 LITERATURE CITED ........................................................................................................................................... 26

APPENDICES

1 Ecosystem classification used in the SBSJ1 subzone variant ........................................................................... 29
2 Code, scientific and common names of species used in this report .................................................................. 31
3 Species composition of seral ecosystems ........................................................................................................ 33
4 Distribution of detailed plots by ecosystem, year, and treatment type ............................................................ 40
5 Vegetation composition of burned seral SBSJ1 ecosystems ........................................................................... 41
6 Environmental attributes of burned seral SBSJ1 ecosystems ....................................................................... 46
7 Vegetation composition of mechanically site prepared ecosystems ............................................................. 51
8 Environmental attributes of mechanically site prepared ecosystems ........................................................... 56
9 Volume of key species in burned seral ecosystems .......................................................................................... 61

TABLES

1 A list of key deciduous tree, shrub and herb species in seral SBSJ1 ecosystems .................................................. 9

FIGURES

1 Map of the distribution of the SBSJ1 subzone variant ...................................................................................... 3
2 Schematic representation of the distribution of ecosystems in the SBSJ1 subzone variant ............................... 4
3 Mature SBSJ1/06 - Queen's cup ecosystem .................................................................................................... 5
4 Mature SBSJ1/01 - Oak fern ecosystem ........................................................................................................... 5
5 Mature SBSJ1/07 - Devil's club ecosystem ..................................................................................................... 6
6 Mature SBSJ1/08 - Horsetail ecosystem ......................................................................................................... 6
7 A mesic site 1 year after burning .................................................................................................................... 10
8 A mesic site 4 years after burning ................................................................................................................... 11
9 A mesic site 9 years after burning .................................................................................................................. 11
10 A mesic site 16 years after burning ............................................................................................................... 12
11 A Queen's cup site four years after burning, an Oak fern site four years after burning ................................. 13
12 Schematic representation of the development of key species in seral (<10 years since burned) SBSJ1 ecosystems ........................................................................................................................................ 14
13 Volume of herbs, shrubs, and deciduous trees in seral (<10 years since burned) ecosystems in the SBSJ1 subzone variant ........................................................................................................................................ 14
14 Cover and height of herbs, shrubs, and deciduous trees in three ecosystems in the SBSJ1 subzone variant ........................................................................................................................................ 16

vi
1 INTRODUCTION

Predictive models of revegetation after clearcutting and site preparation are essential to the development of management prescriptions that will meet integrated resource use objectives. These models will provide information on expected changes in the composition and structure of vegetation over time.

Potential resource management conflicts exist because failure to achieve the silvicultural objective of establishing free-growing plantations is thought to be due, in part, to the presence of shrubby and herbaceous vegetation (Stewart 1984); therefore, reduction in the level of vegetation is a goal of some silvicultural treatments. However, the goals of other management activities may include vegetation enhancement, since these shrubs and herbs may also provide valuable forage and habitat for wildlife.¹ The importance of these species in contributing to long-term site fertility is another important management consideration.

Models of vegetation development following different site preparation treatments will facilitate the evaluation of silvicultural prescriptions and the scheduling of stand tending and rehabilitation activities.

In cool and moist climatic regions, such as the Sub-Boreal Spruce (SBS) Zone, vegetation may reduce crop tree survival and growth through: snow press damage (B. Richards, pers. comm., 1986); reduction of soil temperature because of shading (D. Spittlehouse, pers. comm., 1986), or reduction in light levels or quality (D. Draper, pers. comm., 1986). However, little quantitative assessment has been made to determine the importance and nature of interactions between crop trees and other vegetation, or the response of vegetation to different site preparation treatments. Considerable logging and rehabilitation activities are centred in the SBS zone and particularly in the Willow River variant of the Wet Cool Central subzone (SBS1). Assessments of plantation performance have indicated that many wetter than mesic sites are classified as “Not Satisfactorily Restocked” (NSR).

The SBS zone is also of noted value for wildlife, particularly moose and bears, which use shrubby and herbaceous vegetation that develops after logging or wildfires. Floristic composition and vegetation structure are important determinants of the wildlife habitat values.

Development of land use prescriptions that will preserve essential resource values requires a common framework for communication. The existing Biogeoclimatic Ecosystem Classification (BEC) system (Pojar et al. 1987) provides such a basis for planning. Information on the changes in the presence and abundance of indicator species and other site characteristics that occur after sites are harvested will facilitate the extension of this classification to seral stands.

1.1 Objectives

The aim of this study is to determine the nature of vegetation re-establishment in recently harvested and site-prepared sites in the SBS1 so as to provide:

1. a basis for predicting rates of revegetation and floristic composition in sites after different site-preparation treatments;
2. insight into the relationships between pre-harvest and post-harvest vegetation and other site attributes, which will facilitate the classification of seral ecosystems within the existing BEC system.

Information on patterns of vegetation development will enable managers to:

1. improve predictions of brush potential after different treatments
2. improve the scheduling of silvicultural surveys and treatments
3. determine priorities for site rehabilitation
4. predict wildlife habitat values

1.2 Literature Review

General patterns of post-fire vegetation development have been described for a range of ecosystems in cool and moist climatic regimes similar to the SBS zone. Summaries of vegetation succession in boreal regions (Viereck and Schandelmeier 1980; Wein and MacLean 1983; Parminter3), in interior cedar-hemlock forests (Antos and Habeck 1981; Stickney 1986), in montane and subalpine forests (Arno et al. 1985), and in coastal areas (Kellman 1969) have been published.

Some monitoring of vegetation development after silvicultural treatments has been done in northern Alberta (Corns and LaRoi 1976) and the western Cascades (Dyreness 1973; Long 1973; Lyon 1976; Wittinger et al. 1977; Irwin and Peek 1979). In British Columbia, preliminary accounts of vegetation development in the Interior Cedar Hemlock Zone and in the Engelmann Spruce Subalpine Fir Zone have been compiled.4,5 Eis (1981) provides information on some of the ecosystems of the SBS zone. However, very little site specific information on the response of vegetation to prescribed burning or mechanical site preparation exists for most ecosystems in British Columbia.

1.3 Study Area

The SBSj1 subzone variant extends east from Prince George to the Rocky Mountains, north towards McLeod Lake, and south along the Bowron and Willow river valleys. It is wetter than the Fraser Basin Moist Cool Central (SBSs2) subzone variant to the west and drier than the Very Wet Rocky Mountain (SBSf) subzone towards the mountains in the east (Figure 1). Many of the SBSj1 ecosystems are comparable to those in the adjacent SBSs2 and SBSf, and those of the Moist Cold Northern Sub-Boreal Spruce subzone (SBSn). Average annual precipitation in the SBSj1 is about 800 mm with a range of 542 to 1102 mm. There is 350 mm (230-408) of seasonal precipitation during May to September. Mean annual temperature is 2.5°C (1.9-3.4°C), with a seasonal mean temperature of 11.3°C (9.8-12.4°C). There are approximately 973 (742-1137) growing degree days above 5°C (McLeod and Meldinger (compilers) 1985).

Rolling morainal landscapes predominate with glacial fluvial, fluvial, and lacustrine sediments along rivers. Soils are typically Luvisols, Brunisols, and Podzols, with Organics in depressions. Humus layers in the mature forests are 5-10 cm thick and are classified as Hemimors and Hemihumimors. Hydromors are found in the wettest sites (DeLong et al. 1986).

Research was concentrated on submesic to hygric sites that support the SBSj1/06 Queen's cup,5 SBSj1/01 Oak fern, SBSj1/07 Deviil's club, and SBSj1/08 Horsetail ecosystems (Appendix 1-Table 1; Figure 2). Hybrid white spruce (Picea glauca x engelmanni)6 and subalpine fir (Abies lasiocarpa) form the climax forests in these four ecosystems. Shrubs including Lonicera involucrata, Ribes lacustre, and Vaccinium membranaceum; herbes such as Cornus canadensis, Gymnocarpium dryopteris, Rubus pedatus, Streptopus roseus, Orthilia secunda, and Tiarella trifoliata; and the mosses Pleuroziun schreberi and Pilium crista-castrensis are consistently found in these mature ecosystems (Appendix 1-Table 2).

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4 Dawson, R. 1985. An initial study of vegetation development following logging in the Cariboo Forest Region. B.C. Min. For., Williams Lake, B.C. Unpublished draft report.
5 Classifications of the SBSj1 ecosystems were developed by DeLong et al. (1986) and R. Coupé and A. Yee (editors). 1982. Identification and interpretation of ecosystems in the Cariboo Forest Region. B.C. Min. For., Williams Lake, B.C. Unpublished draft report. A simplified version of these was used as a basis for this work (see Appendix 1-Table 2).
6 Appendix 2 lists species codes and scientific and common names.
FIGURE 1. Map of the distribution of the SBS|1 subzone variant.
The Queen’s cup association occupies submesic sites and is typically found on middle and upper slope positions on glacio-fluvial terraces and morainal and colluvial blankets (Figures 2 and 3). *Pinus contorta* is a characteristic tree species and *Populus tremuloides*, *Betula papyrifera*, and Douglas-fir (*Pseudotsuga menziesii*) are occasionally present (Appendix 1-Table 2). The shrub layer is moderately well developed and includes *Spiraea betulifolia*, the most characteristic shrub, and *Vaccinium membranaceum* and *Rubus parviflorus*. Herbs include *Cornus canadensis*, *Clintonia uniflora*, and *Aralia nudicaulis*. The moss layer is conspicuous. Soils in these sites are well drained, medium to coarse textured Podzols, Luvisols, and Brunisols that have a 4-7 cm thick Hemimor humus layer (DeLong et al. 1986).

The Oak fern association occurs in mid-slope positions on moderately well-drained, medium textured parent materials (Figures 2 and 4). The moderately well-developed shrub layer includes *Vaccinium membranaceum*, *Lonicera involucrata*, *Rubus parviflorus*, and *Ribes lacustre* (Appendix 1-Table 2). *Gymnocarpium dryopteris*, a species whose abundance in these sites differentiates the Oak fern from the Queen’s cup association, *Rubus pedatus*, *Lycopodium annotinum*, *Veratrum viride*, *Cornus canadensis*, and *Streptopus roseus* are common herbs. Soils are moderately well-drained Podzols, Brunisols, and Luvisols and have a 3-8 cm thick Hemimor or Hemihumimor humus layer. These sites are considered to have good productivity (DeLong et al. 1986).

Subhygic sites, which are typically in middle to lower slope positions and on north-facing slopes, support the Devil’s club ecosystem (Figures 2 and 5). Parent materials include morainal, fluvial, and lacustrine deposits. *Oplopanax horridus*, which distinguishes this association, and *Lonicera involucrata*, *Ribes lacustre*, and *Rubus parviflorus* are important components of the shrub layer (Appendix 1-Table 2). Ferns, including *Dryopteris assimilis*, *Athyrium filix-femina*, and *Gymnocarpium dryopteris*, are also typical. *Mniium* spp. are characteristic mosses. Soils are moderately well- to imperfectly drained gleyed Podzols and Luvisols that have a 5-15 cm thick Hemimor or Hemihumimor humus layer (DeLong et al. 1986).
FIGURE 3. Mature SBSj1/06 Queen's cup ecosystem.

FIGURE 4. Mature SBSj1/01 Oak fern ecosystem.
FIGURE 5. Mature SBSj1/07 - Devil's club ecosystem.

Flat and depressional hygic sites on fluvial, lacustrine, and morainal materials support the Horsetail ecosystem (Figures 2 and 6). *Alnus incana* ssp. *tenuifolia*, *Equisetum sylvaticum*, *E. arvense*, *Dryopteris assimilis*, and *Hylocomium splendens* are characteristic of this association. Other differentiating but less abundant species include *Calamagrostis canadensis*, *Circe alpina*, and *Geum macrophyllum* (Appendix 1-Table 2). Organic-rich Gleysols with thick (5-40 cm) Hydromor humus layers develop on these sites, where the water table is generally within 50 cm of the surface for most of the year (DeLong *et al.* 1986).

1.4 Treatment History and Prescriptions

All the sites sampled in the SBSj1 were clearcut in the past 24 years. Submesic and mesic sites were usually summer-logged and wetter sites were harvested in winter. Prescribed burning was generally used for site preparation. Exact burning impact could not be determined because fire effects at the time of burning were not assessed. Nevertheless, observations made in this review indicate that, in general, submesic sites were more severely burned because they often have a drier surface duff layer when burned.

Blade scarification and other mechanical treatments were used, either alone or on sites where burns had been unsuccessful. Hybrid spruce was planted on most of the mesic and wetter sites, with natural or planted lodgepole pine regeneration used on the submesic sites.
2 METHODS

2.1 Field Sampling

Detailed sampling was done in 1984 and 1985 by the authors, and reconnaissance sampling of older stands was carried out in 1986 by C. DeLong.

Potential areas for detailed sampling were selected using recent subzone and forest cover maps and air photos were available. Final selection of seventy-seven sites was done in the field. Detailed sampling was concentrated on sites disturbed within the past 10 years, but a few sites where it had been up to 24 years since site preparation were sampled as well. Wherever possible, paired sampling with mature stands was done to ensure that ecological classification of the clearcuts was accurate.

Vegetation was sampled in 10 x 10 m plots. Percent cover of all species within each of six height strata (0-0.25, 0.26-0.50, 0.51-1.0, 1.01-2.00, 2.01-3.00, and 3.01-5.00 m) was recorded. Deciduous trees were included in the shrub layers. A shallow (30 cm) soil pit was dug and the standard description made for the upper soil and humus layers, according to methods in Wimsley et al. (1980).

Reconnaissance sampling, restricted to the determination of species presence, was done in sites which had been burned more than 10 and less than 24 years before 1986. These plots can be differentiated from the detailed plots by the plot numbers that begin with a letter (e.g., G13h34).

Harvesting, site preparation treatments, and planting information were obtained from history records in the Prince George East and Prince George West Forest Districts, and verified as much as possible by field inspection.

2.2 Analysis

2.2.1 Classification

The VEGSORT program (see Meidinger et al. 1987) was used to compare species composition in seral and climax ecosystems, and to group the seral vegetation data according to the ecosystem unit identified in the field. TWINSPLAN (Hill 1979), a polythetic divisive classification program, was also used to evaluate floristic similarity among plots. Field designated classifications were modified on the basis of these results.

2.2.2 Vegetation structure and composition

A preliminary list of key seral species was developed by comparing the species found in the sample plots to those described by Haeussler and Coates (1986) as important competitors. Frequencies of these species were then calculated for all plots, and species having very low frequencies were eliminated. Other species that had high cover and constancy were added. Table 1 lists key seral species.

An estimate of the volume of space occupied by each species was calculated by summing the product of area covered (m$^2$) and average height (m) in each stratum in each plot. Area covered in the 100-m$^2$ plot was determined by subjective cover estimates. Average height was defined as the mid-point of each stratum. An individual plant was considered to belong to only one stratum. Graphs of cover, volume, and height over time were plotted for total shrubs, deciduous trees, and herbs and for individual species within each stratum, and these graphs were compared for differences in vegetation development among ecosystems and site preparation treatments. Hand-fitted lines were added to the graphs. Small sample size limited comparisons with mechanically prepared sites and hygic ecosystems.
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3 RESULTS AND DISCUSSION

3.1 General Pattern of Revegetation After Site Preparation

The floristic composition of stands after disturbance is determined by: 1) the severity of the disturbance and condition and abundance of understory vegetation before disturbance, as this will influence the extent to which the pre-existing vegetation survives and re-establishes; 2) the suitability of conditions for germination and survival of seed-banking species; and 3) the availability and establishment success of off-site seeds.

Although it was not always possible to identify the mechanism of revegetation in sites that were disturbed more than a few years before sampling, it appeared that many of the species in mesic and wetter sites resprouted from pre-existing plants. Several species establish through the germination of buried seeds. Only a few species appear to seed in from off-site sources. Figure 7 illustrated the vegetation development evident 1 year after burning in some sites.

Herbs, especially fireweed, increased in cover over the first 6–8 years after burning and then began to decline in cover. Shrubs and deciduous trees continue to increase in cover. Figure 8 shows a mesic site 4 years after burning. Where plantations are successful, crop trees are expected to overtop and shade, and thus reduce the cover of herbs and shrubs. The development of vegetation 9 and 16 years after burning is illustrated in Figures 9 and 10. Differences between ecosystems are illustrated in Figure 11.

Seed-banking annuals such as Geranium bicknellii and Corydalis sempervirens are present the year after burning, but were gone by the 2nd year. Epilobium angustifolium was usually common by 2 years after burning or mechanical site preparation. Species diversity increased over time as additional species became abundant. Ribes laxiflorum, Rubus idaeus, species of Salix and Carex, and members of the Asteraceae family (including Antennaria neglecta, Anaphalis margaritacea, and species of Hieracium, Agoseris, and Taraxacum) were among those species that were common after sites were disturbed, but not present in the mature forest ecosystems. These species were most abundant on the drier sites, where generally more severe burning and/or greater disturbance from summer logging exposed more mineral soil and destroyed more of the original vegetation. Rubus parviflorus and Equisetum arvense, common in subhygric and hygric forested sites, respectively, invaded all sites after harvesting (Figure 12).

Over time, most of the herb and low shrub invader species were overtopped by taller invading or resprouting deciduous trees, including Populus tremuloides, Betula papyrifera, species of Alnus and Salix, and conifers, including hybrid spruce and lodgepole pine. The more shade-tolerant ground level shrubs and herbs, which had been present in the forest stand, generally persisted (Figure 12).

3.2 Rate of Revegetation After Site Preparation

On burned sites, the general rate of vegetation development was greatest on the Devil's club sites, followed by the Oak fern and Queen's cup ecosystems (Figure 13). The high shrub and herb volumes associated with the Devil's club sites are likely due to a combination of low burning intensity, which may stimulate shrub regrowth, and higher moisture and nutrient conditions present in these sites. Although species with the potential to develop into dense shrub thickets (e.g., Lonicera involucrata, Vaccinium membranaceum, and Rubus parviflorus) occurred in all of the ecosystems sampled, shrub regrowth differed according to variations in environmental conditions and burning impact. In general the rate of revegetation of ground level species was slower on the drier sites, but these sites also had a larger component of deciduous trees before harvesting and therefore could rapidly develop into deciduous tree dominated stages if extensive resprouting occurred. Shrub and herb development was less in Horsetail than in Devil's club sites, possibly because the cold wet soils of the Horsetail ecosystem inhibit vegetation growth.
FIGURE 7. A mesic site 1 year after burning.

FIGURE 8. A mesic site 4 years after burning.
FIGURE 9. A mesic site 9 years after burning.

FIGURE 10. A mesic site 16 years after burning.
FIGURE 11.  A Queen's cup site 4 years after burning and a Devil's club site 7 years after burning.
FIGURE 12. Schematic representation of the development of key species in seral (<10 years since burned) SBSj1 ecosystems. (See Appendix 5 for species values.)

FIGURE 13. Volume of herbs, shrubs, and deciduous trees in seral (<10 years since burned) ecosystems in the SBSj1 subzone variant.
3.3 General Soil and Humus Layer Properties

Although most soil properties generally remain comparable to pre-logging conditions, soils in submesic sites appear drier and subhygric soils wetter after site preparation. The increase in surface temperature following logging and site preparation results in accelerated decomposition of the litter layer through fungal and faunal activity in these circum-mesic sites. These rapidly changing humus layers are classified as Mormoders.

3.4 Ecosystem-Specific Changes After Site Preparation

3.4.1 Queen’s cup ecosystem

Many of the species present in mature Queen’s cup forests remained after disturbance, although often with reduced abundance. However, some plants took longer to re-establish and other new species invaded, especially after severe disturbances. Important species in the seral (<10 years since disturbed) Queen’s cup ecosystems include *Epilobium angustifolium*, *Rubus idaeus*, *Populus tremuloides*, and *Cornus canadensis* (Appendix 3-Table 3). *Spiraea betulifolia*, a species characteristic of the climax ecosystem, was generally present in the seral state. In general, seral Queen’s cup ecosystems were best differentiated from mesic seral sites by the greater abundance of *S. betulifolia* and *P. tremuloides* and lesser amounts of * Vaccinium membranaceum*, *Lonicera involucrata*, and *Sambucus racemosa* in submesic sites (Appendix 3-Table 1; Appendix 5-Table 1; Appendix 7-Table 1).

Herbs reached their maximum height of about 0.75 m by the 3rd year after disturbance; herb cover continued to increase over the first 9 years after disturbance (Figure 14). The tallest herb layer (0.5-1.0 m) was dominated by *Epilobium angustifolium* with a small component of *Calamagrostis canadensis*.

Although most shrubs remained less than 1 m in height (Figure 14), deciduous trees, including aspen and cottonwood, were sometimes over 4 m tall by 4 years after disturbance. Total shrub and deciduous tree cover increased steadily. *Lonicera involucrata*, *Ribes lacustre*, *Rubus idaeus*, and occasionally *Viburnum edule* reached a height of 1 m, while *Ribes laxiflorum* and *Rubus parviflorus* generally remained less than 0.5 m tall. Willows reached 2 m or more by 9 years after disturbance.

Increased exposure because of canopy and humus layer removal appeared to dry the soil in these sites. Humus layers, typically classified as Orthic Mormoders, were decomposing faster than on mature sites, as evidenced by the presence of an Ah layer not found in mature forest Orthic Hemimors (Appendix 6-Table 1; Appendix 8-Table 1).

3.4.2 Oak fern ecosystem

Dominant species in the seral Oak fern ecosystem included *Epilobium angustifolium*, *Rubus parviflorus*, *Rubus idaeus*, and *Lonicera involucrata* (Appendix 3-Table 4). Natural regeneration of subalpine fir was common. There were no species that clearly differentiated seral Oak fern stands from other seral ecosystems, however, *Sambucus racemosa* and *Vaccinium membranaceum* are most characteristic of mesic sites (Appendix 3-Table 1). Some climax constants including *Lonicera involucrata*, *Ribes lacustre*, *Rubus pedatus*, *Streptopus roseus*, *Valeriana sitchensis*, *Gymnocarpium dryopteris*, and *Cornus canadensis* were also important in early seral ecosystems, while other climax constants such as *Lycopodium annotinum* and several moss species were much less common in seral stages (Appendix 3-Table 4; Appendix 5-Table 2; Appendix 7-Table 2).

Herbs peaked at a height of about 1 m (Figure 14). Total herb cover reached 50-100% by 4 years after disturbance. *Epilobium angustifolium* was the only herbaceous species that reached a height of over 1 m; other species were generally less than 0.5 m tall. Total shrub and deciduous tree cover was generally less than 60% (Figure 14). *Lonicera involucrata*, *Rubus idaeus*, *Rubus
parviflorus, Ribes spp., Sambucus racemosa, and Viburnum edule reached a height of about 1 m by 3 years after disturbance. Vaccinium membranaceum generally remained less than 0.5 m tall. Populus tremuloides and Salix spp. continued to increase in height over time.

Humus layers in the seral Oak fern sites, classified as Orthic Mormodors, were characterized by rapid decomposition of organic material. Those in mature forests were generally Orthic Hemimors (Appendix 6-Table 2; Appendix 8-Table 2).

3.4.3 Devil’s club ecosystem

Dominant species included Epilobium angustifolium, Rubus parviflorus, Rubus idaeus, Ribes lacustre, and Lonicera involucrata (Appendix 3-Table 5). Actaea rubra, Cornus sericea, Veratrum viride, and Ophiopanax horridus differentiated seral Devil’s club ecosystems from mesic seral SBSj1 ecosystems (Appendix 3-Table 1). Gymnocarpium dryopteris, Tiarella trifoliata, Epilobium angustifolium, Rubus idaeus, Ribes lacustre, and Rubus parviflorus were seral constants. All, except Epilobium angustifolium and Rubus idaeus, are also climax constants. Other climax constants, including Ophiopanax horridus, Rubus pedatus, Streptopus roseus, S. amplexifolius, and Dryopteris assimilis were uncommon in the seral sites and therefore are not reliable ecosystem indicators (Appendix 3-Table 5; Appendix 5-Table 3; Appendix 7-Table 3).

Herb cover increased rapidly in the first 4 years after disturbance and maximum height was about 1.5 m by the 7th year after disturbance (Figure 14). Epilobium angustifolium is the only herb
that exceeded 1 m in height; other prominent herbs, including *Calamagrostis canadensis* and *Athyrium filix-femina*, remained shorter.

Shrubs reached about 1.5 m in height (Figure 14) and 65% cover by the 7th year after disturbance. *Cornus sericea* and *Lonicera involucrata* are the tallest shrubs. *Sambucus racemosa* and *Viburnum edule* sometimes reached a height of 1-2 m after 6 years. *Ribes* spp., *Rubus parviflorus*, and *R. idaeus* were usually less than 1 m in height, and *Vaccinium membranaceum* was generally less than 0.5 m. Taller *Populus tremuloides* and *Salix* spp. were found in most sites. *Alnus* species and *Populus balsamifera* ssp. *trichocarpa* were found in some recently disturbed sites and *Betula papyrifera* occurred in some older cutblocks.

Humus layers in burned Devil's club sites were classified as Orthic Mormoders, reflecting the rapid decomposition typical of these sites. Those in forested sites are usually classified as Orthic Hemimors and Hemihumimors (Appendix 6-Table 3; Appendix 8-Table 3).

### 3.4.4 Horsetail ecosystem

Dominant species in the seral Horsetail ecosystem included *Lonicera involucrata*, *Rubus idaeus*, *Rubus parviflorus*, *Alnus incana*, *Ribes lacustre*, *Epilobium angustifolium*, *Erigeron annuus* ssp., and *Calamagrostis canadensis* (Appendix 3-Table 6). Climax constants, including *Lonicera involucrata*, *Ribes lacustre*, *Gymnocarpium dryopteris*, *Equisetum sylvaticum*, and *Cornus canadensis*, were generally present in seral Horsetail sites; other climax constants, including *Rubus pedatus* and *Streptopus amplexifolius* were uncommon (Appendix 3-Table 6; Appendix 5-Table 4; Appendix 7-Table 4).

Herbs reached up to 75% cover 1 year after disturbance and up to 80-100% cover after 3 years. Most herbs were a maximum height of about 0.75 m. *Epilobium angustifolium* reached 1-2 m height in the first few years after disturbance, while the prominent herbs, including *Athyrium filix-femina* and *Calamagrostis canadensis*, generally remained less than about 0.5 m in height. Shrub cover peaked at about 50% after 7 years. *Lonicera involucrata* dominated the upper strata along with *Rubus idaeus* and occasionally *Viburnum edule*. *Ribes lacustre* had a significant cover in the 0.5-1 m stratum by 7 years after disturbance. *Rubus parviflorus* and *Ribes laxiflorum* generally remained less than 0.5 m in height. *Alnus incana* ssp. *tenuifolia* and *Salix* spp. developed into tall shrubs.

Humus layers in forested Horsetail sites included poorly drained Mors, Moders, and Mulls, while those in recently burned sites were typically very poorly drained Saprimulls with thick organic layers (Appendix 6-Table 4; Appendix 8-Table 4).

### 3.5 Species Responses to Site Preparation Treatments: revegetation dynamics and implications

*Epilobium angustifolium* (fireweed)

*Epilobium angustifolium*, a minor component of the mature forest stands in the SBSj1, resprouted from rhizomes and established from seeds throughout cutblocks soon after burning and mechanical site preparation (Appendix 3-Tables 3-6; Appendix 5; Appendix 7). Site preparation appears to reduce competition from other species and provide a suitable seedbed (Watson et al. 1980). Fireweed will resprout from rhizomes the year after burning in cedar-hemlock sites (Stickney 1986). Seeds, which are reported to be viable for less than 2 years (Myerscough 1980), often arrive on-site soon after site preparation.

Peak abundance of fireweed is reached by the third growing season in coastal sites and by the 5th year in cedar-hemlock sites, with declines in abundance evident by the 10th year (Stickney 1986). In the

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SBSj1, however, where site occupancy by other species was probably slower, fireweed grew to a height of about 1 m on submesic sites and about 1.5 m on mesic and wetter sites within a few years. It achieved over 50% cover, and then appeared to decline in abundance (Appendix 5; Appendix 7). In Alaska, fireweed continues to increase in cover for up to 30 years on boreal spruce sites, declining as the forest canopy closes (Foote 1983).

Maximum fireweed development occurs in the most severely burned boreal and cedar-hemlock sites (Ahlgren 1960; Mueggler 1965; Morris 1970). A similar trend is evident in the SBSe2, where E. angustifolium is most abundant on the ecosystems that correspond with the SBSj1 Queen's cup and Oak fern associations, is less abundant in the SBSe2 Devil's club ecosystem, and is virtually absent from the SBSe2 Horsetail ecosystem. This pattern suggests that fireweed is a poor competitor in these wetter sites where other pre-existing vegetation often resprouts soon after burning (Eis 1981). In the SBSj1, however, E. angustifolium was most abundant on mesic and subhygric sites, in spite of greater abundance of other vegetation (Appendix 9-Figure 1). This suggests that in some situations the greater availability of moisture and nutrients may be more important than potentially negative effects of other vegetation.

Fireweed may contribute to seedling snow press damage (J. Pollack, pers. comm., Jan. 1985, cited in Haussler and Coates 1986; B. Richards, pers. comm., 1985; Brand8) and to reducing soil temperature by shading (D. Spittlehouse, pers. comm., 1986) in mesic and wetter sites in the SBSj1. On submesic sites, the species did not appear to have sufficient volume or density to cause appreciable snow press damage to lodgepole pine, which is usually planted on these sites and generally grows fast enough to avoid light competition or mechanical damage from fireweed. The importance of moisture competition between lodgepole pine and E. angustifolium on drier sites and the potentially beneficial effects of shading and other positive contributions of the species, which absorbs nutrients released after burning and thus maintains them on site (Watson et al. 1980), have not been determined.

**Lonicera involucrata** (black twinberry)

*Lonicera involucrata*, a shade-tolerant species, was one of the most common shrubs in mature and seral SBSj1 ecosystems (Appendix 1-Table 2; Appendix 5; Appendix 7). In cedar-hemlock sites other species of *Lonicera* survive burning and resprout from the root crown the following year (Stickney 1986). The species often disappears after severe burning in boreal sites, but resprouts vigorously after light burns (Ahlgren 1960). In seral SBSj1 ecosystems, black twinberry had greater abundance on mesic and wetter sites and was consistently present after burning in the Oak fern ecosystem (Appendix 9-Figure 2). Resprouting and growth of the species after prescribed burning was fairly slow in the Queen's cup ecosystem, where it was more likely to be killed by burning (Appendix 3-Table 2; Appendix 5; Appendix 9-Figure 2). There was no evidence of re-establishment from buried seeds during the first ten years after disturbance in sites in the SBSj1.

Eis (1981) reported that *L. involucrata* increased on sites in the SBSe2 that are comparable to the SBSj1 Oak fern, Devil's club, and Horsetail ecosystems, achieving the greatest cover and height on the wettest, mechanically prepared sites. Eis (1981) also found that 5 years after logging, *L. involucrata* was common on burned and mechanically prepared SBSSe2 sites that are comparable to the SBSj1 Oak fern ecosystem, and cover increased to 25% in the burned SBSSe2 Devil's club ecosystem and to 40% on mechanically prepared SBSSe2 Horsetail sites.

Black twinberry reached a height of 1.2 m within 2 years on burned SBSSe2 Horsetail sites and after 7 years on SBSSe2 Devil's club sites. In the SBSj1 black twinberry reached a maximum of about 1.5 m in height. The greatest cover of black twinberry was in the Horsetail association in the SBSj1 and SBSSe2.

**Rubus parviflorus** (thimbleberry)

*Rubus parviflorus*, a common understory species in mature SBSj1 forests, (Appendix 1-Table 2) was consistently present in both burned and mechanically prepared sites with an average cover of about 5-10% (Appendix 3-Table 1; Appendix 5; Appendix 7; Appendix 9-Figure 3). Thimbleberry remained less than about 1 m in height and was generally shorter on drier sites.

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Algren (1960) found *Rubus* species to be resistant to — and often enhanced by — burning in the boreal. *Rubus parviflorus*, a prolific seed producer and a long-term seed-banker, establishes immediately after burning from seed and by resprouting from rhizomes in the SBSj. Similar results are reported for the cedar-hemlock (Stickney 1986), coastal (Kelpas 1978), and intermountain areas (Wright 1972). Thimbleberry was sometimes abundant immediately after burning but appeared to decrease in volume over time in the drier mesic and submesic SBSj1 ecosystems (Appendix 9-Figure 3). This suggests that the species may be well adapted to the high nutrient availability and low competition from other species found immediately after burning, but is less successful once other species have re-established. Thimbleberry is found at low light levels, but is most abundant at 60-100% of full light levels in Oregon sites (Emmingham 1972). Eis (pers. comm., 1987) also found a decrease over time in the cover of *R. parviflorus* in the SBSSe2 equivalents of the SBSj1 Oak fern and Queen's cup ecosystems as other vegetation increased in height. Once established, thimbleberry increases through rhizomatous extension in larch/fir sites in Montana (Stickney 1981) and British Columbia (Marchant and Sherlock 1984).

Witterger et al. (1977) reported that thimbleberry was more abundant after burning in cedar-hemlock sites, but that it declined in abundance over 25 years. On coastal sites, establishment occurred immediately after burning, maximum height growth was reached in the first few years, and the species remained abundant for 5 years.9

Thimbleberry was most abundant on the subhyglic SBSSe2 (Eis 1981) and SBSj1 Devil's club ecosystems where it was likely most abundant before harvesting. It appeared to expand to occupy sites left vacant by the die-back of devil's club, which does not tolerate canopy removal. In the SBSj1 Horsetail sites, lack of an initial source for *R. parviflorus* appeared to limit development of the species.

*Rubus idaeus* (red raspberry)

*Rubus idaeus* is generally absent from mature SBSj1 forests, although present in openings in the mature SBSj1 Devil's club ecosystem (Appendix 1-Table 2). The species, which re-establishes through resprouting and germination of stored seeds in the SBSj was consistently present after sites were disturbed and had an average cover of 7-20% for the first 10 years after disturbance (Appendix 5; Appendix 7). Burning promotes the development of the species by stimulating germination of buried seeds (Sharp 1970; Stickney 1986) and resprouting of underground rhizomes, which are quite resistant to burning (Ahlgren 1960; Wright 1972).

Foote (1983) found that *R. idaeus* invaded black and white spruce sites the 1st year after burning in Alaska, but declined once trees were established. Ahlgren (1960) reported that red raspberry took at least 5 years to become dominant on jack pine sites in northern Minnesota, and then declined in abundance. *R. idaeus* generally remained less than 1 m in height in all SBSj1 sites and increased in volume over the first 10 years after site preparation (Appendix 9-Figure 4). By 14 years after burning, red raspberry was generally no longer present in mesic and submesic sites in the SBSj, presumably because it was shaded out by other species (Appendix 5-Tables 1 & 2). *Rubus idaeus* does not tolerate shade and devotes more energy to seed production as it becomes shaded (Whitney 1982).

The most rapid growth rates in burned sites appeared to be in the SBSj1 Devil's club ecosystem, where burning impacts were probably less severe and nutrient and water availability greater, followed by the Oak fern and Queen's cup ecosystems (Appendix 9-Figure 4). Raspberry has a high demand for soil nutrients and is most abundant where they are plentiful (Wright 1972). No difference in the abundance of the species on burned compared with mechanically prepared sites was apparent.

*Salix* spp. (willows)

Willows are generally shade-intolerant (Lyons 1952; Rawson 1974) and are not typically found in most mature SBSj1 forests (Appendix 1-Table 2). They seeded into clearcuts after disturbance and were

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a common component of seral stands, particularly wetter sites, by 6 years after site preparation (Appendix 9-Figure 5). *Salix* spp. were important in mesic and submesic SBSj1 sites up to at least 16 years after burning (Appendix 5-Tables 1 and 2). Willows resprout readily from the root crown after fire (Wright 1972; Stickney 1986). Resprouting is maximized by quick, hot fires and inhibited by longer burns (Wright 1972; G. MacKinnon, pers. comm., Jan. 1985, cited in Haeussler and Coates 1986). Although seeds are widely dispersed (Stickney 1986), seed viability is short lived (Zasada et al. 1983) and development from seedlings is slow in larch/fir sites in Montana (Stickney 1981). Comparisons of burned and unburned sites in Alaska (Viereck and Dyrness 1979) and Idaho (Mueggler 1965) show that willows are favoured by burning. There was no difference in the abundance of the species after burning, compared to mechanical site preparation in the SBSj (Appendix 3-Tables 3 to 6).

*Alnus incana* ssp. *tenuifolia* (mountain alder)

*Alnus incana* ssp. *tenuifolia* was common in the mature hygric Horsetail ecosystems in the SBSj1, but was rarely found in drier forested sites (Appendix 1-Table 2). Although quite shade-tolerant (Krajina et al. 1982), mountain alder is usually found where there is good exposure to sunlight (Lyons 1952). It is generally absent from mature Devil’s club sites, but will establish in these subhygric sites after burning. Mountain alder was restricted to seral Horsetail and Devil’s club ecosystems in the SBSj1. Increases in abundance after canopy removal have been reported, especially on wetter sites, and alder resprouts after burning or if damaged (Healy and Gill 1974; Stickney 1986). Seedling viability is low (USDA 1974) and germination of the species may require the saturated soils that are generally only available in wet sites (Healy and Gill 1974). This could explain why it was not found in mesic and submesic SBSj1 sites (Appendix 3-Table 1). No difference in the abundance of the species on burned compared to mechanically prepared areas was observed (Appendix 3-Tables 3 to 6). Although total cover was generally low in sites in the SBSj1, mountain alder will grow to a height of several metres within a few years after being disturbed.

*Populus tremuloides* (trembling aspen)

*Populus tremuloides* was a common component of the mature submesic SBSj1 Queen’s cup ecosystem, which has a history of frequent fires. Although generally not abundant in the mesic and wetter forest stands at the time of harvesting, aspen was found in the range of submesic to subhygric sites after site preparation. Suckering from the root system is common. No evidence of establishment from seeds was observed in the sites sampled in the SBSj. Strotherman and Zasada (1957) found that the seed was viable for only 2 or 3 weeks, and Barnes (1966) suggested that reproduction by seed is limited because this short period of viability rarely occurs when there is enough moisture to ensure seedling survival. Seedling survival is also low in the boreal (Rowe 1983). There was no apparent difference in the abundance of the species after either mechanical site preparation or burning in the SBSj1 (Appendix 3-Tables 3 to 6).

Aspen suckers will grow very quickly and can be over 2 m in height within a few years in the SBSj1. Total cover was generally around 10-20% in the mesic and submesic sites by 16 years after sites were disturbed (Appendix 5).

*Ribes laxiflorum* (trailing black currant)

*Ribes laxiflorum*, although relatively uncommon in mature SBSj1 forests, germinated from buried seeds and was fairly abundant by the 3rd year after sites were disturbed (Appendix 9-Figure 6). Other species of *Ribes* are also reported to be seed-bankers in cedar-hemlock (Stickney 1986) and boreal (Rowe 1983) sites. No appreciable difference in the cover of the species was found in different seral ecosystems or in burned vs mechanically prepared sites (Appendix 3-Tables 3 to 6). *Ribes laxiflorum* did not usually exceed 0.5 m in height, even on the richest SBSj1 sites. Other species apparently outcompeted *R. laxiflorum* within a short time, as it was not common in older burned sites (Appendix 5-Tables 1 and 2). Other *Ribes* species are also shade-intolerant (Viereck and Little 1972) and nutrient-
demanding (Haeussler and Coates 1986), and *R. laxiflorum* is probably not as successful once the initial flush of nutrients is depleted and taller vegetation shades it. The lack of rhizomatous sprouting and the reliance on seed germination (M.Newton, pers.comm., cited in Haeussler and Coates 1986) noted in other *Ribes* species could also explain the decline of this species over time, as the surface seedbank is exhausted and shading by other vegetation makes conditions for new seeding establishment less favourable.

*Ribes lacustre* (black gooseberry)

*Ribes lacustre*, a fairly shade-tolerant species, was often present with low cover in mature SBSj forest stands (Appendix 1-Table 2). It was usually present with less than 10% cover in both mechanically prepared and burned sites (Appendix 5; Appendix 7). Site preparation does not appear to have a detrimental effect on established plants and average cover in mature stands was comparable to that of clearcuts in the SBSj1. There may be an initial increase in abundance of the species in the first few years and then a decline by 9 or 10 years when taller vegetation had begins to shade it out (Appendix 9-Figure 7). Like other species of *Ribes*, *R. lacustre* is a long-term seed-banker that establishes immediately after burning in the SBSj and cedar-hemlock sites in Idaho (Stickney 1986). Rhizomatous extension is limited (Stickney 1986). In the SBSj1, plants remained less than 1 m tall.

*Vaccinium membranaceum* (black huckleberry)

*Vaccinium membranaceum* is a fairly shade-tolerant species common in the mesic and submesic mature SBSj1 forests, where it typically had about 10% cover. No obvious increase in the abundance of the species was evident for up to 10 years after burning in the SBSj1 (Appendix 9-Figure 8). Eis (pers. comm., 1987) observed similar results in the SBSj2. The species was set back by burning and seeding establishment was limited in these sites in the first 10 years. The greater abundance of *V. membranaceum* on mechanically treated compared to burned SBSj1 clearcuts also indicates that burning can kill the species (Appendix 5; Appendix 7; Appendix 9-Figure 10). In the SBSj, re-establishment after burning occurs only through resprouting from the rhizome; no seeding establishment has been observed. Similar results were reported in Idaho (Stickney 1986). Although *V. membranaceum* increases in vigour and cover after canopy removal, no increase in stem density or seedings has been reported in northern Idaho (Mueggler 1965) or in the Pacific Northwest (Minore et al. 1979). Miller (1977) also reported very little seeding establishment in burned sites in Montana and recovery from burning is described as slow in Oregon and Washington (Minore et al. 1979). In coastal sites, light fires stimulate sprouting from dormant vegetative buds on underground rhizomes, but intense fires kill rhizomes (Minore et al. 1979). In boreal sites, other species of *Vaccinium* have comparable abundance on light, heavy and unburned sites, indicating that response to burning is variable (Ahlgren 1960). Because *V. membranaceum* is an important component of mature forest stands, it is expected to maintain a presence on sites over time as the canopy closes.

In the SBSj1, plants were generally less than 0.5 m tall in <10 year old sites. The species is not perceived to be an important competitor in the SBSj1 or in the SBSj2 (Eis 1981), or in areas in Oregon and Washington (Minore 1979, cited in Haeussler and Coates 1986).

*Calamagrostis canadensis* (bluejoint)

Although common only in the hygric forested sites, *C. canadensis* was frequently present with low cover (<10%) in SBSj1 sites after disturbance. No difference in abundance was observed on mechanically prepared vs burned sites (Appendix 3-Tables 3 to 6). Rhizomes survive and increase in abundance after burning and the species also rapidly seeds-in to disturbed sites in Alberta (Watson et al. 1980). Bluejoint establishes immediately after and is enhanced by burning in the boreal in Minnesota (Ahlgren 1960) and Alaska (Foote 1983). Other species of *Calamagrostis* also survive burning and establish by seed in cedar-hemlock sites (Stickney 1986). *Calamagrostis canadensis* did not appear to be a serious competitor on any of the sites in the SBSj1, although it develops into continuous mats 3 or 4 years after logging in the boreal near Dawson Creek and in eastern Canada (Frisque et al. 1978).
**Sambucus racemosa** (red elderberry)

Although not common in mature submesic and mesic SBSj1 sites (Appendix 1-Table 2), *S. racemosa* establishes from buried seeds and through resprouting from rootstocks immediately after burning and is common by 3-4 years after burning (Appendix 5; Appendix 7; Appendix 9-Figure 9). *Sambucus racemosa* was found to be a seed-banking species on the coast (A. McGee, pers. comm., 1987), and is apparently enhanced by burning in cedar-hemlock (Mueggler 1965) and coastal sites (Lafferty 1972; Wright 1972). The species also resprouts from rootstocks in Oregon (M. Newton, pers. comm., Dec. 1984, cited in Haeussler and Coates 1986) and in cedar-hemlock sites (Stickney 1986). In some coastal sites, no increase in cover is apparent after burning.10 *Sambucus racemosa* increased in abundance over the 6 years studied in the SBSSe2 (S. Eis, pers. comm., 1987).

**Viburnum edule** (highbush-cranberry)

*Viburnum edule* was present with limited cover in submesic to hygic forests and in burned sites in the SBSj (Appendix 1-Table 2; Appendix 5; Appendix 9-Figure 10). Low impact fires stimulate germination of highbush-cranberry seeds and resprouting of stems in the boreal forest (Rowe 1983). Parminter also found that shallow burns did not kill rhizomes and noted sprouting at the base of stems after fires in northern British Columbia.11 No appreciable increase in the abundance of *V. edule* was evident in the SBSj1 (Appendix 9-Table 13), although the species appeared to become more frequent over time (Appendix 5; Appendix 7).

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10 Brand 1984.

4 SUMMARY AND CONCLUSIONS

4.1 Classification Within the Existing Biogeoclimatic Ecosystem Classification System

Attributes used to classify sites, including moisture and nutrient regime, and floristic composition can change markedly when sites are disturbed. Changes in moisture regime are particularly evident on severely disturbed submesic sites, which may appear drier, and on wet sites, where the water table may rise after forest canopy removal. Floristic composition changes considerably on severely disturbed sites and indicator species important in identifying ecosystems may be lacking. In areas where disturbance has been minimal, e.g., on lightly burned sites, or where undisturbed vegetation remains after patchy burns or mechanical site preparation, floristic composition is similar to pre-treatment conditions and existing ecosystem guides can easily be used to provide accurate site classification. More information is needed on revegetation patterns in severely disturbed sites.

4.2 Rate of Revegetation

The rate of vegetation development was greatest on the Devil’s club sites, followed by the Oak fern and Queen’s cup sites. The high vegetation volumes associated with the subhygric Devil’s club sites are likely due to a combination of low burning intensity, which may stimulate shrub regrowth, and to greater availability of moisture and nutrients.

4.3 Revegetation Strategies of Key Species

4.3.1 Establishment

Species establish in sites through the germination of seeds already on-site or newly arrived after disturbance, and/or through the resprouting of underground rhizomes that have survived burning or other disturbances. Sambucus racemosa, Ribes lacustre, R. laxiflorum and Rubus idaeus establish through seedbank germination. Populus tremuloides re-establishes through the resprouting of underground stems or roots. Some species, including Epilobium angustifolium, Calamagrostis canadensis, and Rubus parviflorus, may establish by seed but also extend their range by vegetative reproduction. Mode of re-establishment of some species, including Viburnum edule, Lonicera involucrata, and Vaccinium membranaceum has not been well established. Betula papyrifera and Salix spp. appear to seed-in over time.

4.3.2 Persistence and Longevity

Plants that increased in abundance over the first 10 or more years included Populus tremuloides, which will form the forest canopy on drier sites; Alnus incana ssp. tenuifolia, which persists in the understory in wetter sites; and Salix spp., which are greatly diminished once the forest canopy has developed. Calamagrostis canadensis increased after disturbance, particularly in the hygic sites. Lonicera involucrata and Vaccinium membranaceum were slow to regrow after disturbance, but persist and are important in the understory of mature forests.

Some of the species common immediately after disturbance, although initially abundant, are not very shade-tolerant and are outcompeted by taller species over time. Epilobium angustifolium persisted for up to 20 years; Rubus idaeus was replaced on submesic and mesic sites by about 14 years after the initial disturbance; and Rubus parviflorus, although common in mature coniferous forests, appeared to decline as the deciduous forest canopy developed. Ribes laxiflorum was abundant initially, but was short lived.

4.4 Site Preparation Effects

Burning diminished the abundance of a number of the original plant species on submesic and mesic sites, however fireweed rapidly occupied these sites. In moister sites, burning reduced the abundance of shrubs and herbs initially, but significant regrowth of these species and invasion by fireweed occurred in a short time. There were not enough similarly treated, mechanically prepared areas to allow an adequate assessment of the effects of this treatment.
5 LIMITATIONS OF THE STUDY

The use of a chronosequence approach to developing ecosystem-specific predictive models of vegetation development after different site preparation treatments is limited by the following factors:

1. lack of information on pre-treatment vegetation and site conditions. This caused difficulties in classifying sites disturbed by harvesting and site preparation, according to the BEC system, when no comparable adjacent mature forests existed.

2. difficulty in determining site history — such as time of year harvested — because of limitations of the site history record system.

3. lack of information on, and considerable year-to-year and site-to-site variation in, treatment impacts. Specifically, variations in climatic conditions at the time of burning partly determine burning intensity, which influences whether vegetation is killed or only temporarily set back by burning.

4. variation in local alphanumeric factors (including phenological status of the vegetation at the time of treatment and presence of off-site seed sources), which, although they influence vegetation development patterns, cannot be determined through sampling years later.

The impacts of different levels of vegetation on crop tree performance were not specifically assessed. It was not feasible to make this type of an evaluation without a detailed knowledge of site history, including information on seedling condition at time of planting, site preparation impacts, and climatic conditions. Such information could only be obtained through detailed monitoring.
6 RECOMMENDATIONS

6.1 Classification

Further investigations into the nature of vegetation development after clearcutting and site preparation should be done to facilitate the development of field guides to the classification of seral ecosystems. Particular emphasis should be placed on determining how floristic composition and site moisture and nutrient regimes change following the most commonly used site preparation treatments. Priority should be given to ecosystems where silvicultural rehabilitation or habitat enhancement activities are under way to ensure that ecological classification can be readily used as a framework for the development of management prescriptions.

6.2 Species Response to Treatment

Research should focus on determining the response to site preparation treatments of dominant species (e.g., *Populus tremuloides*, *Epilobium angustifolium*, and *Rubus parviflorus*), which may be important competitors with crop trees, and of other components of the mixed shrub complex, including *Salix* spp. and *Betula papyifera*, which are of particular importance to wildlife. Determination of growth rates and mode of re-establishment (vegetative reproduction, seedbank germination, or invasion of new seeds) should be a priority.

6.3 Relationship Between Competing Vegetation and Crop Tree Performance

Studies should be initiated to determine the impacts on crop trees of the type and volume of vegetation established after burning in Devil's club sites. Hypotheses that spruce growth is limited by competition for light, water, or nutrients or by maintenance of low soil temperatures, should be tested. Research should also focus on determining the possible beneficial effects of non-crop vegetation, such as protection from loss of winter snow cover and desiccation damage, shading in drier sites, and improvement of site nutrient regime. Determination of the level of non-crop vegetation that is optimal for long-term productivity is essential.

Although impacts of non-crop vegetation on crop trees were not specifically assessed in this study, observations suggest that the ground level vegetation on submesic and mesic sites is not generally dense enough or tall enough to have a significant negative impact on spruce tree performance as a result of shading or reduced soil temperatures. However, snow press damage may occur on the mesic sites, especially where fireweed is abundant soon after burning. In the Devil's club and Horsetail sites, where vegetation regrowth was greatest, reduction in spruce tree performance may be expected.

6.4 Research Approaches

Predictive models of revegetation should be developed through the establishment of experimental sites that are classified ecologically prior to treatment and monitored to determine specific impacts of site preparation treatment and subsequent vegetation development patterns.
7 LITERATURE CITED


Lyons, C.P. 1952. Trees, flowers and shrubs to know in British Columbia. J.M. Dent and Sons Ltd., Vancouver, B.C.


Wein, R.W. and D.A. MacLean. 1983. The role of wildfire in northern circumpolar ecosystems. J. Wiley and Sons, Toronto, Ont.


APPENDIX 1. Ecosystem classification used in the SBSj1 subzone variant

TABLES

1. Ecosystem classification in the SBSj1 subzone variant
2. Common and differentiating species in mature ecosystems in the SBSj1 subzone variant

TABLE 1. Ecosystem classification in the SBSj1 subzone variant

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<thead>
<tr>
<th>Symbol</th>
<th>Current classification *</th>
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<td>SBSj1/02</td>
<td>Pine lichen</td>
<td>NS b</td>
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* This classification was developed in 1985 and represents a revision of the SBSj1 classification in the Prince George Forest Region (DeLong et al. 1986) and in the Cariboo Forest Region (Coupé and Yee (editors), 1982).

b NS: Not sampled.
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| PINU CON     | V 11.5 III 3.4 I 1.7 I 0.7 |
| POPU TRE     | II 0.3 I 0.2 |
| BETU PAP     | II 1.0 I 0.4 II 0.5 |
| PSEU MEN     | II 3.2 I 0.2 |
| SPIR BET     | IV 1.8 II 0.5 II 0.2 II 0.0 I 0.0 |
| RUBU PAR     | III 3.6 IV 6.3 V 4.5 III 1.2 |
| ARAL NUD     | IV 2.9 III 2.7 II 1.2 I 0.1 IV 3.7 |
| CLIN UNI     | V 2.5 IV 1.0 III 1.3 II 0.0 I 0.1 |

| LYCO ANN     | IV 5.5 V 4.2 IV 1.8 III 0.3 III 0.6 |
| VERA VIR     | II 0.2 IV 1.2 III 0.9 III 0.1 |

| OPLO HOR     | I 0.1 II 0.9 V 30.4 V 20.8 III 1.0 |
| DRYO ASS     | II 0.1 III 3.0 V 8.2 V 3.4 IV 2.1 |
| ATHY FIL     | I 0.0 II 0.4 IV 6.8 IV 6.7 V 1.6 |
| MNUI MED     | I 0.0 II 1.2 II 3.2 I 2.2 III 3.6 |
| MNUI NUD     | I 0.5 II 2.1 II 5.3 |

| ALNU INC 2   | II 5.9 IV 3.5 |
| EQUI SYL     | I 0.0 II 0.1 III 2.0 V 2.4 V 8.9 |
| EQUI ARV     | I 0.1 I 0.1 III 1.3 V 13.5 V 19.0 |
| CALA CAN     | I 0.1 I 0.0 III 0.1 III 1.3 |
| CINC ALP     | I 0.0 II 0.2 III 0.3 III 0.3 |
| HYLO SPL     | III 5.3 IV 11.8 III 12.0 III 26.2 V 15.1 |

Key to Presence Classes

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* This classification was developed using data collected by the Prince George Forest Region and Cariboo Forest Region ecologists and pedologists. Not all ecosystems are shown.

* This table presents a partial species list only.

* See Appendix 2 for full species names.