TECHNICAL REPORT

Rehabilitation of Caribou Winter Range Following Attack by Mountain Pine Beetle: Monitoring Protocol and Early Post-fire Vegetation Dynamics

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ABSTRACT

Mountain pine beetle (*Dendroctonus ponderosae*) outbreaks have lead to significant changes in the understory communities of lodgepole pine (*Pinus contorta*) forests in recent years, such that they are becoming dominated by dwarf shrubs and terrestrial lichens are declining. This has caused concern about the supply of range for woodland caribou (*Rangifer tarandus caribou*) during winter when their diet is primary composed of terrestrial lichens. Previous studies of post-fire and post-logging succession in these forests have led us to believe that prescribed burning may assist in redirecting the succession of vegetation communities from one dominated by dwarf-shrubs to one dominated by terrestrial forage lichens. Unfortunately, the rate of lichen recovery after a prescribed burn would likely be too slow to know whether or not the technique is effective until well after treatments have been applied. Using surrogate indicators of rehabilitation success may allow managers to determine whether treatments have been effective in a much shorter amount of time, and would allow treatments to be refined or applied to other areas before the short-term supply of terrestrial lichens becomes critically low. In this study, we reviewed relevant literature and conducted a small-scale post-burn vegetation survey to determine how a naturally burned forest recovered in lichen (xeric) and non-lichen (mesic or sub-mesic) sites, and suggested potential indicators for monitoring the effectiveness of prescribed burning. We concluded that post-burn surrogates of a "forage lichen" successional trajectory should be indicated by several plants in addition to the relative abundance of specific plant functional groups. For example, 2-3 years after prescribed burning of a lichen site, we would expect to see the presence of Arctostaphylos uva-ursi, the high abundance of Spiraea betulifolia and Linnaea borealis, and a general increase in the ratio of invaders to sprouters. We also reviewed potential monitoring methods and recommend photo-plot based surveys as the most effective. Based on these results, a monitoring protocol was developed to guide pre- and post-treatment effectiveness assessments for MPB-attached ungulate winter ranges sites chosen to receive rehabilitation.

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PREFACE

This report is the second in a series of technical reports (TR) concerning the use of prescribed burning for management of ungulate winter range (UWR). The broad goal was to develop rehabilitation methods applicable to low-elevation winter range used by woodland caribou (*Rangifer tarandus*) wherever their range had been previously impacted by mountain pine beetle (*Dendroctonus ponderosae*). The series of reports and their general objectives consist of:

- TR #1 - evaluation of UWR within the Laidman Resource Management Zone to identify sites that could potentially form the basis of a replicated adaptive management trial[1];
- TR #2 - development of efficacy indicators and selection of methods for monitoring forest floor vegetation communities (this report);
- TR #3 – development of an adaptive management plan[2]; and
- TR #4 – documentation of site attributes at the selected burn site[3].

[1] Sulyma, R. and R. S. McNay. 2008. Rehabilitation of caribou winter range following attack by mountain pine beetle: field reconnaissance summary UWR U-7-012. Wildlife Infometrics Inc. Report No. 286. Wildlife Infometrics Inc., Mackenzie, British Columbia, Canada.
[2] Sulyma, R and R. S. McNay. 2009. Rehabilitation of caribou winter range following attack by mountain pine beetle: Prescribed burn plan UWR U-7-012. Wildlife Infometrics Inc. Report No. 307. Wildlife Infometrics Inc., Mackenzie, British Columbia, Canada.
[3] Sulyma, R.. 2009. Rehabilitation of caribou winter range following attack by mountain pine beetle: Pre-treatment site monitoring UWR U-7-012. Wildlife Infometrics Inc. Report No. 312. Wildlife Infometrics Inc., Mackenzie, British Columbia, Canada.

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INTRODUCTION

The mountain pine beetle (*Dendroctonus ponderosae*, MPB) epidemic has resulted in millions of hectares of forests with dead pine trees (*Pinus contorta* var. latifolia) throughout the interior of British Columbia. The provincial government targeted these forests for accelerated timber harvest rates to capture the greatest benefit from a declining fibre quality (Province of BC 2007b). However, the volume of timber being processed has not matched the volume of mortality, resulting in large areas of dead pine trees that may have a much slower recovery to pre-MPB conditions (Province of BC 2007b, but see Burton 2006). This unprecedented mortality and incomplete salvage and reforestation of pine forests has negative consequences for the forest industry and can also be detrimental to the supply of some ungulate winter ranges (UWR) used by woodland caribou (*Rangifer tarandus caribou*).

Within the Ministry of Environment's Omineca Region, more than 300,000 hectares of low-elevation pine forests have been designated as Caribou UWR under the Government Actions Regulation (Province of BC 2005a, 2005b, 2007a). Management of these UWRs is guided by General Wildlife Measures (GWMs) that focus on maintaining the quality of the range; particularly a sustainable supply of the terrestrial forage lichens (primarily *Cladina* spp.). Forage lichens tend to dominate the understory of these pine forests (Province of BC 2005a, 2005b, 2007a) during distinct (but not all) stages of natural vegetation succession (Coxson and Marsh 2001). Winter ranges therefore require regular disturbance to perpetuate the overall supply of lichens. To facilitate this regular disturbance, a two-pass timber harvest system implemented on a 140-year rotation has been recommended (Province of BC 2005a, 2005b, 2007a). However, in light of the MPB epidemic, its rate of spread, and changing management priorities for forest licensees; there is a high likelihood much of the pine-lichen UWR will not be scheduled for harvest (McNay et al. 2009).

Wildfire is considered to be the natural disturbance mechanism that allows terrestrial forage lichens to develop (Kershaw 1977, Sulyma and Coxson 2001). However, disturbances from wildfires are rare since the 1970's, perhaps because of the combined effects of aggressive fire suppression and unfavourable weather (Burton 2006). Disturbance resulting from the MPB alone differs from wildfire potentially affecting the development of terrestrial lichen vegetation communities. The different outcomes in MPB-killed stands compared to wildfire are: 1) the organic layer of the forest floor is not returned to basic nutrients, 2) coarse debris from dead wood reaches much higher quantities and spatial coverage, and 3) the forest understory vegetation remains intact continuing as an advanced seral stage relative to the forest overstory. The ultimate effects of these differences is currently unknown and bound by uncertainty as significant variation in lichen community development exists at a variety of scales ranging from regional level (pers. comm., Harold Armleder, BC Ministry of Forests, 09.11.13) to the site level (Haeussler et al. 2008). In the Tweedsmuir-Entiako area, west of the Omineca, creeping shrubs capitalize on a brief increase in availability of soil nutrients and moisture in the early years following beetle-caused pine mortality (Williston et al. 2006). Forecasting subsequent community development is speculative, but a risk is evident that lichen presence could be reduced throughout a stand rotation in the absence of a more severe (relative to MPB) stand level disturbance.

Prescribed burning is a management tool that has potential for restoration of caribou UWRs; especially where changing site conditions promote atypical plant communities relative to those generally observed after stand level disturbances (Williston et al. 2006) and where debris accumulations can potentially restrict caribou access to habitat (Cichowski 2007). Past use of prescribed fire in the north-central interior of BC has, however, generally been associated with silviculture activities and applied in timber harvested areas, rather than in mature forests or in forests with mostly standing dead trees. As a consequence, there is little information available for establishing monitoring procedures and criteria upon which to assess effectiveness of prescribed burning as a management tool to rejuvenate terrestrial forage lichens.

The objective of this project phase was to establish indicators (e.g., plant associations or wildfire severity and/or fire behaviour indices) and sampling methods, which could be used to measure the effectiveness of prescribed burn treatments for rejuvenating terrestrial forage lichens within UWRs. Based on this information, our second objective was to form a monitoring protocol that could guide pre- and post-treatment effectiveness assessments. This phase was part of a larger adaptive management project on understanding factors affecting vegetation community development in pine-lichen forests (McNay and Sulyma 2007).

BACKGROUND

The development of terrestrial forage lichens cannot be monitored directly during initial post-fire stages as one might do for many trees and shrubs, because lichen propagules (vegetative or otherwise) rarely survive a fire in such quantities that they can re-establish or re-grow. Instead, lichens must disperse into a site from unburned areas, a process that can take decades in larger burns (Coxson and Marsh 2001). The slow growth rate that most mat-forming lichens are known to exhibit (Thomson 1967) further delays direct measurement of lichen development and it would be difficult to undertake meaningful direct measurement of terrestrial forage lichen abundance for at least 30-40 years after fire. This time-frame bears an unacceptable risk for managing a sustainable supply of terrestrial forage lichens within UWRs for two compelling reasons. First, there is no direct evidence that prescribed burns implemented today within UWRs will have the desired outcome. Second, in 30 years without management many of the existing UWRs will likely have aged to a stage of vegetation succession where the forest understory will be dominated by feathermoss or shrubs, leaving forage lichens in short supply. It is therefore necessary to test prescribed burning as a tool and find indicators to use in the short term that would provide evidence that a post-burn site is either on or off a vegetation succession trajectory (henceforth lichen trajectory) leading to dominance by terrestrial forage lichens.

Potential indicators of the lichen trajectory could be associated with the characteristics of wildfires themselves, or with the early seral, post-fire vegetation communities that develop afterwards. Of particular interest are those indicators from sites known to have been dominated by terrestrial forage lichens. By documenting the plant community and physical characteristics of former terrestrial lichen sites in the first years after wildfire, we gain insight as to what an adequately rehabilitated site should look like after prescribed burning. The assumption is that these sites will continue on a vegetative succession path to be again dominated by terrestrial lichens. The temporal development of terrestrial lichen communities has been documented for several regions of the country (Carroll and Bliss 1982, Kershaw 1977, Johnson 1981, Morneau and Payette 1989) with the observed patterns varying across broad climatic conditions. In the Omineca region of British Columbia, Coxson and Marsh (2001) documented a chronosequence beginning with an early phase (0 to 50 years-old) dominated by deep rooted mosses (e.g., *Polytrichum juniperinum*). The early phase was followed by a club lichen phase (*Cladonia* spp.) (51 to 70 years-old), then a shrub (forage) lichen phase (*Cladina* spp.), (71 to 140 years-old), and finally by a feather moss phase (>140 years-old); primarily the species *Pleurozium schreberi* (Coxson and Marsh 2001). Sulyma (2002) found a similar trend to the vegetation succession in the Laidman lake area of the Vanderhoof Forest District. However, the breadth of the chronosequence assessments in both the Omineca and Vanderhoof forest districts did not provide the necessary resolution to detect distinct species associations in early post-fire years.

STUDY AREA

The study area for this project phase was situated on Vantine Creek in Entiako Provincial Park; it was chosen to take advantage of a wildfire which occurred in July of 2006 (Figure 1). Vantine Creek is approximately 26 km north-west of UWR U-7-012 (the study area for other, related project phases; Sulyma and McNay 2008) and is characterized by similar ecological characteristics (see below). The 700-ha fire that occurred at Vantine Creek was relatively uniform through multiple forest layers (crown, boles and duff), leaving little more than charred, unbranched snags on the site.

The area is situated in the moist cool subzone of the sub-boreal pine spruce (SBPS) biogeoclimatic zone and was heavily impacted by the MPB epidemic in the late 1990's. Large portions of the burn area are typified by the mesic (01a), submesic (01b), and xeric (02) site series, and were set up for vegetation monitoring in 2007 (Cichowski and Williston 2008). Because the area is under the rain-shadow of the Coast Mountains (Meidinger and Pojar 1991), it has a dry and cold climate throughout most seasons. For example, the mean annual precipitation is approximately 450 mm, while the mean monthly temperature ranges from 13° C in the summer to -14° C in the winter (Banner et al. 1993). Forests of the SBPSmc zone are mostly even-aged pine less than 120 years old because of frequent stand-replacing fires; this held true for our study area, where mature dead lodgepole pine or mixed lodgepole pine and white spruce (*Picea glauca x engelmannii*) dominated prior to the burn (Cichowski and Williston 2008). A significant portion of the lodgepole pine was dead prior to the wildfire due to the mountain pine beetle epidemic, which was prevalent in the area from 1999 to 2003 (Eng et al. 2005). Hybrid spruce stands occurred primarily on wetter seepage sites and as bands along wetlands, black spruce (*Picea mariana*) was generally limited to forested wetlands, and both species are rare in mesic and xeric site series. Soils consist of coarse-textured glacial and glaciofluvial deposits, typically greater than three meters thick, with significant organic deposits limited to valley bottoms (Plouffe and Levson 2002). Topography is variable, with the majority of the area forming a series of eskers and valleys, generally oriented in a south-west to north-east direction, with steeper slopes on the south-east facing sides and alternating slopes and benches on the north-west.

METHODS

Post-fire Vegetation Community Characterization

We reviewed the findings of Cichowski and Willlison (2008) who visited the study area prior to us. We decided to supplement their data with additional plot information. Cichowski and Williston established 15 plots located in a band oriented in an east-west

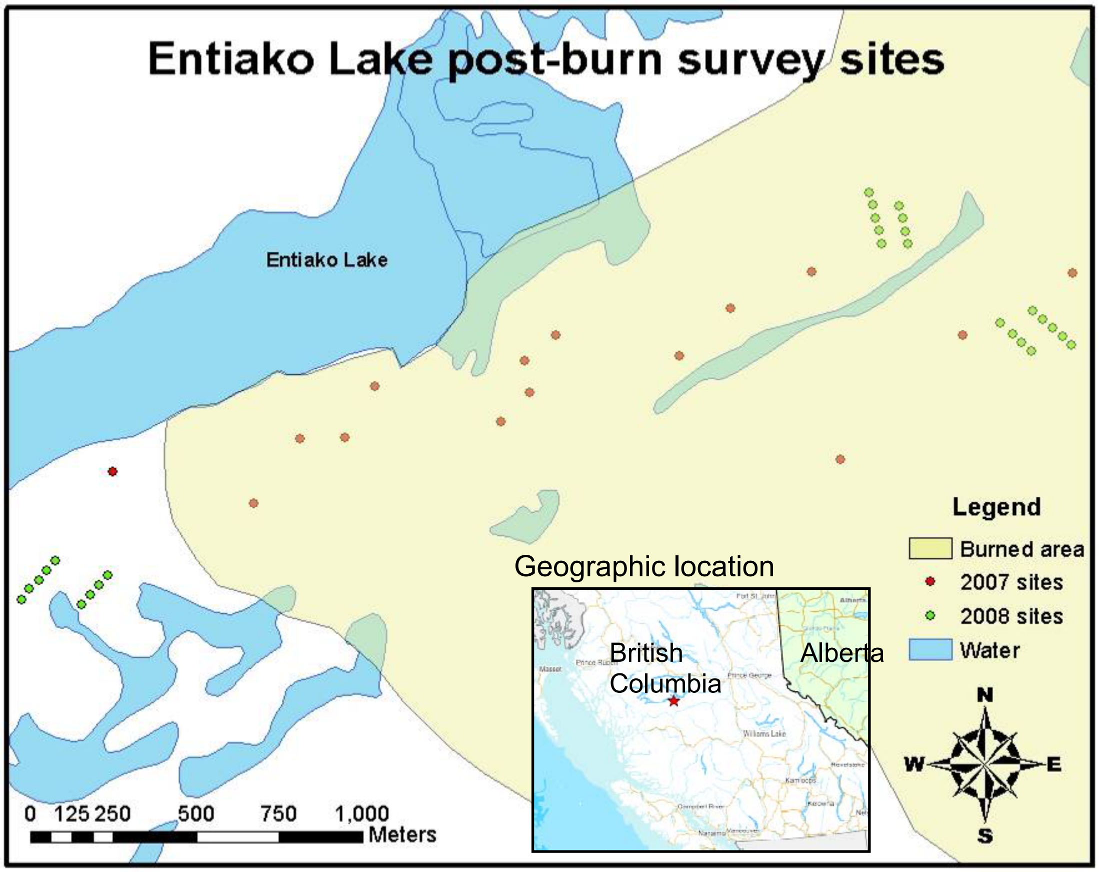

Figure 1. Entiako Lake post-burn survey sites. Note: Western edge of burned extends to the western-most 2008 sites.

direction through the middle of the Entiako Park burn (Cichowski and Williston 2008). They established five plots in each of three different site series observed in the area: the mesic (01a), submesic (01b), and xeric (02). Our supplemental plots were selected with an intentional bias towards xeric (02) site series, since the 02 site series is known to have higher *Cladina* spp. abundance at maturity (Delong et al. 1993), and towards south or east-facing slopes since they were under-represented in Cichowski and Williston's (2008) data.

Spatial information, including forest cover[4] and terrain[5] data, were managed with a geographic information system (GIS) to identify additional sample area. The GIS queries were done in ArcMap (ESRI 2009) and were based on parameters specified in the Caribou Habitat Assessment and Supply Estimator (CHASE) (Brumovsky 2003, McNay et al. 2006) for low-elevation, pine-lichen winter range (site index <15, soil moisture regime submesic or drier, and soil nutrient regime of poor or very poor).

For logistic simplicity, our supplemental plots were regularly spaced along several transects within the targeted site series'. Once potential locations were visited to confirm

[4] Vegetation Resources Inventory (VRI) (Province of BC (n.d.))
[5] Digital Elevation Model (DEM) (Province of BC 2002);

suitability, random distances (within 50 meters) and directions were chosen to begin the first plot of each transect, after which four to eight plots would be surveyed approximately every 40 meters along the transect. Transect orientation was always in a cardinal or sub-cardinal direction, such that at least four plots could be surveyed along its length while remaining in the desired habitat. The precise number of plots was dictated by a combination of environmental boundaries and time limitations, with a minimum of four and maximum of eight plots per transect.

Our plots, at 100 m^2 in size and circular in shape, were half the size of those used by Williston et al. (2006), in order to increase the overall number of locations visited. At each plot we recorded the diameter at breast height (DBH), species, and burn severity class of every standing tree (live or dead). Burn severity classes were recorded on a scale of 0-5, with 5 being the most severely burned and 0 being unburned, following Cichowski and Williston's (2008) methods. We also recorded the canopy closure at plot centre (with a concave spherical densiometer), the slope position, percent slope, slope aspect, and spatial coordinates at plot center (Universal Transverse Mercator, NAD83), and took a planar photo from the south side of the plot. Our plots were distributed from approximately the centre of the burn to the far eastern section near the lake shore, such that each plot was at least 100 meters from the nearest previously established plot or the lake shore, and at least 40 meters from any wetland.

We used both systematic (one quadrat in center of plot and one at 2.5 meters in each sub-cardinal direction) and true random (two random distances, used as x and y coordinates, based on a grid overlaying the circular plot) methods to locate the five, low-vegetation quadrats within each plot, unlike Williston et al. (2006), who used a random distance and random direction from plot centre to locate quadrats. We chose to sample only five quadrats because it ensured a similar quadrat density to Williston et al. (2006). Each quadrat was approximately 0.5 m^2 and square in shape.

Quadrats were surveyed first for the two-dimensional percent cover (by species) of vascular plants, lichens, and identifiable mosses. Non-living cover types were classified as mineral soil, bare rock, coarse woody debris (greater than 2.5 cm in diameter), charcoal, litter, or cryptogamic crust (aggregations of small bryophytes, crustose or young squamulose lichens, algae, cyanobacteria, and fungi). Once cover estimates were complete, the center of each quadrat was excavated to a depth of approximately 20 cm. The depth of litter and humus layers was recorded in centimeters to one decimal place, and the excavated mineral soil was examined for texture and coarse fragment content, following methods outlined in Delong et al. (1993) for soil description.

Resulting data sets (percent cover of observed cover types and depth of litter and humus layers) were entered into a Microsoft Excel database and organized by year of sampling, aspect class, burn-severity class, plant association/site series, vegetation layer, and species functional group. Basic statistical analysis (summaries of descriptive stats and pairwise comparisons) were completed using SAS (SAS Institute Inc., Cary, North Carolina).

Long-term Forest Floor Vegetation Monitoring

We reviewed, summarized, and contrasted two methods commonly used for monitoring terrestrial lichen communities in BC: 1) a line transect procedure used in the Chilcotin (Pucket 1995, Waterhouse 1998) and 2) a photo-imaging method used in the Tweedsmuir-Entiako (Williston et al. 2006) and Omineca (Sulyma and Sulyma 2006) regions. Although other, more standardized, vegetation monitoring methods have been recommended for use in British Columbia forests (Province of BC 2006), these tend to be unsuitable for detection of fine-scale changes and the mechanisms that are responsible for them. In addition to the methods summaries, a survey with a caribou/lichen expert (Deborah Cichowski) was completed to gain an understanding of the *pros and cons* for the two methods. The findings were tabularized for the purpose of comparison and recommendations for future project phases were made.

RESULTS

Post-fire Vegetation Community Characterization

Summary of previous findings

The burn severity at Vantine Creek was judged by Cichowski and Williston (2008) as being uniform across all plots and most trees were identified as burn class 5, where the bole was “heavily burned (>80%) > 1 m above the ground”. Five plots were established in each of xeric (02 site series), submesic (01b) and mesic (01) site conditions where thirty-nine species of vascular plants, bryophytes and lichens were identified (Cichowski and Williston 2008). Species diversity was relatively even across all three site series evaluated (Cichowski and Williston 2008).

Summary of Current findings

Twenty six supplemental plots, distributed among five transects, were established in August, 2008. Of these plots, three were mesic (01a), four were submesic (01b), and 19 were xeric (02). Burn severity was uniformly high throughout most plots, with the exception of plot 1 on transect 3 (S3P1), and plot 0 on transect 2 (S2P0), which were both on a slope facing away from the burn (Table 1). Thirty-six individual species were observed at the plot level and as found by Cichowski and Williston (2008) species diversity was relatively even across all three site series as indicated by our plots (Appendix 1).

Summary of Vegetation in the First Two Years Following Wild Fire

The mean cover by site series for the most prevalent[6] species is presented in Table 2. Twenty-nine species were tallied, but only nine of these were considered abundant (> 1%) in the 01b and/or 02 site series (01b/02). These were: *Arctostaphylos uva-ursi, Calamagrostis Canadensis, Cornus Canadensis, Epilobium angustifolium, Linnaea borealis, Polytrichum juniperinum, Rosa acicularis, Spiraea betulifolia,* and *Vaccinium caespitosum.* Of these nine, *S. Betulifolia* (T(39)=2.552,P=0.147) and *L. Borealis*

[6] “Prevalent” was considered a mean cover > 0.1 percent.

Table 1. Physical plot characteristics, by ecological site series and plot identification, observed in a survey of post-fire vegetation indicators, Vantine Creek Burn, British Columbia, 2008.

Ecological Site Series and Plot Identification	*Topography*			*Mineral soil*		*Organic layer depths (cm)*		*Remnant trees / snags*		
	Slope Position	*% Slope*	*Aspect*	*Soil type*	*% Coarse Fragments*	*Litter*	*Humus*	*Wildlife tree class*	*Burn severity*	*Basal area*
SBPSmc 01a										
S1P3	mid	4	310	sandy loam		0.8	0.4	7	high	37.56
S1P5	lower	5	150	fine sandy loam	34	1.1	0.2	7	high	32.20
S1P6	lower	20	155	loamy sand	48	1.4	0.3	7	high	33.38
SBPSmc 01b										
S2P2	bench	2	54	loamy sand	24	2.5	1.4	6	mod	38.55
S2P3	bench	2	195	fine sandy loam	6	2.1	1.2	6.5	mod	45.09
S1P1	bench	5	300	loamy sand	--	1.2	0.4	7	high	38.66
S1P2	bench	3	355	sandy loam	--	0.6	0.2	7	high	23.93
SBPSmc 02										
S4P1	mid	11	340	sand	60	1.1	0.1	7	high	13.63
S4P4	mid	4	190	sand	65	1.8	0.0	7	high	33.43
S4P2	mid	7	260	sand	74	1.1	0.3	9	high	3.13
S2P0	mid	3	220	loamy sand	21	4.3	0.8	4	low	38.09
S3P1	mid	20	240	loamy sand	15	3.8	0.8	4	low	27.52
S3P2	crest		340	fine sandy loam	2	1.2	0.9	5	mod	25.51
S2P1	upper	7	15	sandy loam	23	1.9	0.9	5.5	mod	21.43
S4P3	mid	13	212	sand	46	1.3	0.0	6	mod	11.64
S1P4	upper	2.5	300	fine sandy loam	--	0.9	0.2	7	high	28.65
S1P7	mid	26	125	loamy sand	24.4	1.0	0.5	7	high	11.33
S1P8	crest	5	5	loamy sand	30	0.8	0.7	7	high	34.42
S1P9	upper	3	330	loamy sand	38	1.4	0.2	7	high	16.56
S2P4	mid	4	175	sandy loam	30	2.5	0.8	7	high	31.26
S3P3	upper	21	153	fine sandy loam	6	5.7	1.0	7	high	26.09
S3P4	mid	8	320	fine sandy loam	6	4.1	0.5	7	high	16.95
S5P1	mid	12	232	sand	30	1.2	0.1	7	high	39.21
S5P2	crest	6	105	sand	17	1.7	0.0	7	high	41.76
S5P3	upper	15	339	sand	30	1.5	0.8	7	high	17.86
S5P4	crest	8	188	sand	10	1.3	0.4	7	high	29.55

Table 2. Percent cover of substrate and plant species, by ecological site series, observed in surveys of post-fire vegetation, Vantine Creek Burn, British Columbia, summarized for 2007, 2008 and 2007/08 combined. Values presented are the mean followed by the standard error in parenthesis.

	Growth Strategy[1]	**2007**			**2008**			**2007/08**			**2007/08**	
		01a	**01b**	**02**	**01a**	**01b**	**02**	**01a**	**01b**	**02**	**01a**	**01b/ 02**
Number of Samples (n)		5	5	5	3	4	19	8	9	24	8	33
SOIL		3.1(1.9)	32.6(10.5)	0.4(0.2)	10.3(0.7)	10.5(5.6)	11.6(3)	5.8(1.8)	22.8(7.1)	9.2(2.5)	5.8(1.8)	12.9(2.8)
ROCK		6.9(3.7)	5.8(2.1)	0.7(0.4)	7.7(2.1)	2.2(0.8)	3.9(0.9)	7.2(2.3)	4.2(1.3)	3.3(0.7)	7.2(2.3)	3.5(0.6)
LITTER		80.2(7.8)	56.4(10.2)	93.2(2.1)	43.6(7.6)	49.8(4.1)	42(3.6)	66.5(8.5)	53.4(5.8)	52.7(5.2)	66.5(8.5)	52.9(4)
Achillea millefolium	I				1.4(1.3)		0.2(0.2)	0.5(0.5)		0.2(0.2)	0.5(0.5)	0.1(0.1)
Arctostaphylos uva-ursi	S			0.8(0.8)			2.5(1.6)			2.1(1.2)		1.5(0.9)
Arnica cordifolia	I/B/S		0.6(0.6)		0.3(0.3)			0.1(0.1)	0.3(0.3)		0.1(0.1)	0.1(0.1)
Aster sp.	I						0.4(0.4)			0.3(0.3)		0.2(0.2)
Calamagrostis canadensis	I/S	0.4(0.2)	0.1(0.1)		2.7(2.7)	1(1)	1.6(0.7)	1.3(1)	0.5(0.4)	1.3(0.5)	1.3(1)	1.1(0.4)
Carex spp.	B	0.2(0.1)	0.3(0.1)	0.2(0.1)	1.3(0.6)	0.5(0.4)	1.1(0.5)	0.6(0.3)	0.4(0.2)	0.9(0.4)	0.6(0.3)	0.7(0.3)
Cladina spp.	--			0.1(0.1)			0.1(0.1)			0.1(0.1)		0.1(0.1)
Cladonia spp.	--						0.1(0.1)					
Cornus canadensis	S	0.7(0.3)	0.6(0.4)		3.9(0.4)	1.2(0.7)	2.3(0.5)	1.9(0.6)	0.9(0.4)	1.8(0.4)	1.9(0.6)	1.6(0.3)
Epilobium angustifolium	I/B	1.2(0.7)	1.4(0.9)	0.1(0)	7.1(2.9)	10.7(1.5)	6.5(1.4)	3.4(1.5)	5.5(1.8)	5.1(1.2)	3.4(1.5)	5.2(1)
Epilobium ciliatum	I						0.1(0.1)			0.1(0.1)		0.1(0)
Geranium bicknellii	B		1.1(0.6)		0.2(0.2)		0.4(0.4)	0.1(0.1)	0.6(0.4)	0.3(0.3)	0.1(0.1)	0.4(0.2)
Linnaea borealis	S	0.3(0.1)	1.2(1)	2.4(0.9)	3(1)	4.9(1.5)	5.9(0.8)	1.3(0.6)	2.8(1)	5.2(0.7)	1.3(0.6)	4.5(0.6)
Orizopsis asparifolia	I						0.2(0.2)			0.2(0.2)		0.1(0.1)
Orizopsis pungens	I			0.1(0)	3(3)		0.7(0.2)	1.1(1.1)		0.6(0.2)	1.1(1.1)	0.4(0.1)
Petasites palmatus	I	0.8(0.6)						0.5(0.4)			0.5(0.4)	
Peltigera aphthosa	--						0.1(0.1)			0.1(0.1)		0.1(0.1)
Peltigera spp.	--						0.8(0.8)			0.6(0.6)		0.5(0.5)
Pinus contorta	B					0.2(0.1)			0.1(0)			0.1(0)
Pleurozium schreberi	--						1.5(1.5)			1.2(1.2)		0.9(0.8)
Polytrichum juniperinum	I/B	0.2(0.1)	0.2(0.1)	0.1(0)	1.1(1)	1.1(0.9)	1.5(0.4)	0.5(0.4)	0.6(0.4)	1.2(0.3)	0.5(0.4)	1(0.3)
Populus tremuloides	S			0.2(0.2)			0.5(0.5)			0.4(0.4)		0.3(0.3)
Rosa acicularis	S/B	0.7(0.4)	0.3(0.2)	0.8(0.4)	3.4(1.8)	1.3(1.3)	2.3(0.8)	1.8(0.8)	0.7(0.6)	2(0.7)	1.8(0.8)	1.6(0.5)
Salix sp.	S			0.2(0.2)	0.3(0.3)			0.2(0.1)			0.2(0.1)	
Shepherdia canadensis	S		0.1(0.1)	0.1(0)	2.6(1.4)	2.6(1.9)	1.5(0.6)	1(0.7)	1.2(0.9)	1.2(0.5)	1(0.7)	1.2(0.4)
Solidago spathulata	I			0.1(0)								
Spiraea betulifolia	S		3.5(1)	1.8(0.8)	2.7(1.4)	5.2(1.9)	6.1(1.1)	1(0.7)	4.3(1)	5.2(0.9)	1(0.7)	4.9(0.7)
Spiraea pyramidata	S	0.2(0.2)						0.1(0.1)			0.1(0.1)	
Vaccinium caespitosum	S	0.7(0.4)	0.3(0.2)	0.6(0.4)	3.7(0.6)	2.1(0.2)	2.7(0.7)	1.8(0.6)	1.1(0.3)	2.2(0.6)	1.8(0.6)	1.9(0.4)

[1] The sources for Growth Strategy information are summarized in Appendix 2.

(T(39)=2.532,P=0.0155) were the only two that had significantly greater abundances in the 01b/02 compared to the 01a. *A. uva-ursi, P. juniperinum, Orizopsis asperifolia, Solidago spathulata* and all of the lichen species also appeared to be more abundant in the 01b/02 compared to the 01a, but differences were not significant.

Overall, plant-habitat associations were less variable in the 2008 survey; plots classified as mesic and submesic were less compositionally different from xeric plots in 2008 than in 2007. In addition, total plant cover and cover of each growth strategy group was higher in the 2008 survey (Table 3). Invaders showed the strongest relative increase between survey years (Table 3).

Table 3. Percent cover and rate of increase between survey years of growth strategy functional groups

Growth strategy	Survey 2007	Survey 2008	Rate of increase
Sprouter	5.76	23.84	4.14
Seed-banker	1.97	9.77	4.97
Invader	2.18	13.16	6.04

Comparison of Long-term Forest Floor Vegetation Monitoring

A nested plot design using line intersects was applied by Armleder and Waterhouse (Waterhouse 1998) to monitor trends in vegetation communities in the Chilcotin (Itcha-Ilgatchuz) area of BC. With this method, plot locations are permanently marked using metal pins that also serve to anchor a 0.8 m radius metal hoop (plot frame). During surveys, the hoop is set in place using the anchor pins, and three identical metal rods are situated in a triangular pattern within the hoop. One of the rods is then used as a line-intercept transect, along which the total distance occupied by lichens and bryophytes is recorded by species to the nearest 0.5 cm; the vigor (healthy, sickly, ordead) of individuals or patches that occur along this transect is also recorded. Competing vascular vegetation is assessed by visually estimating percent-cover within the hoop. Various measures can be taken to ensure that the line-intercept transects used at each successive survey are in the same place as for previous surveys, such as ensuring that all transect portions of the triangle have the same orientation, and using leveling devices and adjustable legs for the plot frame (Waterhouse 2008).

Photo-plots have been regularly used to monitor trends in vegetation communities in the Tweedsmuir-Entiako and the Omineca regions of the province (Cichowski and Williston 2008, Williston et al. 2006, Sulyma and Sulyma 2006). Plot locations are permanently marked with metal pins that also serve to anchor a square quadrat frame. Using a tripod with a horizontal boom, a camera is extended and centered over the quadrat, and photos of the plot are taken. Visual estimates of percent cover for individual species (vascular and non-vascular) or groups are also performed. Photos are then analyzed using software products that allow for calibration and rectification of fixed tag points. Williston et al. (2006) have used Gap Light Analyzer software (v.2, Canham 1988) to obtain percent-cover readings by functional group (forage lichen and non-forage lichen). Sulyma and Sulyma (2006) used a manual digitizing procedure based on Sigmascan

Pro V5 (SSI 1999). Regardless of the procedure a quantitative measure representing the area occupied by a functional group can be derived for comparisons among surveys.

The advantages, disadvantages, and potential mitigating measures for each of the two techniques are summarized in Table 4. Other techniques have been used in BC, but these do not typically have the same level of precision as line-intercept's or photo-plot's. For example, Seip and Jones (2007) used a square plot frame with an inset 10 cm x 10 cm wire grid for permanent plot monitoring, and recorded the species of lichen or plant at each point of intersection on the grid. Seip and Jones (2007) marked their plots in similar fashion to Williston et al. (2006), by inserting two permanent stakes outside opposing corners of the plot frame. Rather than having a number of plots clustered within a larger tree survey plot, Seip and Jones (2007) placed their plots at the beginning of a 100m transect, which was further surveyed for vegetation dominance using a line-intercept method. These methods would be better for coarse-scale assessment of a large area at relatively low cost, but may not be able to detect the more subtle, site-specific changes in community composition that occur after wildfire or prescribed burning.

DISCUSSION

Post-fire Vegetation Community Characterization

Much of the past work documenting vegetation composition in pine forests has been done in conjunction with chronosequence work that was not focused on early vegetation response immediately after a wildfire. Chronosequence work on lichen types in the Omineca Region of BC have characterized early seral as 0 to 50 years (Coxson and Marsh 2001, Sulyma 2002). This range is too large to provide detailed characteristics of the responding vegetation complex. The surveys outlined and presented in this report provide the necessary characterization of early post-fire floristic composition and are therefore more appropriate as a foundation for future post-burn monitoring comparisons.

We have identified a list of 29 species (Table 2) that can be used to determine if the outcomes of prescribed burning in pine-lichen forests (i.e. 01b or 02 site series) are similar to the results of wildfire on the same forest type. During the early years after fire there appears to be little difference between the vegetation communities on a mesic 01a site compared to drier sites but the presences of lichen remnants as well as the presence a select group of species can help infer if a prescribed burn will eventually contain conditions suitable for a terrestrial lichen community.

Cichowski and Williston (2008) found that *Solidago spathulata* only inhabited xeric plots in a one-year old burn, though it did so in low abundance. This species is known to associate with very coarse textured soils and recently disturbed sites (Mackinnon et al. 1999). We observed *S. Spathulata* more consistently on xeric sites than mesic or submesic sites, but due to the low frequency of occurrence it was not recorded in our quadrats. The presence of *S. spathulata,* albeit at low abundance (0.1 %), is likely a good indicator of sites that will produce lichen dominated vegetation communities over time. Likewise grasses of the genus *Oryzopsis*, were also identified as associates of the xeric site series (Meidinger and Pojar 1991). *O. pungens* is often relatively abundant throughout the early seral stages of lichen woodland development and present during the lichen-dominance stage, but may decline during the feathermoss dominance stage

Table 4. Comparison summary of small-plot survey methods for evaluating forest floor vegetation cover (Deborah Cichowski pers comm.).

Feature	Assessment	Photoplots	Line-intercept (hoop)
Sampling	Method	• Site consists of one 7.98 m radius circular plot with 10 photoplots, alternatively 30+ photoplots established on line transect	• Site consists of 30+ hoop plots on a grid spaced 50 m apart
	Features	• Photoplots are in a consistent site • Includes measurements of trees, seedlings, and saplings within 7.98m plot	• Data are averaged over sites • Includes counts of cones & fecal pellet groups and estimates of arboreal lichens on sample trees
	Issues	• Data from 10 photoplots from 1 site could be biased by 1 event on the site • Difficult on sloping ground to get perpendicular angle from plot	• Site type may vary for hoop plots over a large site
	Improvements	• Keep detailed records with horizontal photos of site conditions	• Maintain plot-level resolution of data for some analyses rather than averaging over site
Equipment	Method	• 75 cm x 75 cm collapsible frame • Tripod and boom for camera • Camera	• 1.6 m diameter aluminum hoop with inlaid triangle with one side being 130 cm measurement line • Aluminum rods & feet for hoop • T-square to measure line length
	Features	• Small and portable • Lightweight	• Rigidity ensures consistency with plot size and location
	Issues	• Small deviations in exact plot location from year to year make re-measurement of individual thalli difficult. • Setting up and using camera may take longer than simple visual assessments	• Bulky • Difficult to transport in helicopter or plane if sampling in remote areas / would need to use the "rod" rather than the hoop for remote areas • Ergonomics can be difficult in bending over and searching centre of hoop and in reading line intercept (especially when there are trees on the line intercept side of the hoop)
	Improvements	•	• Use collapsible or light-weight version of hoop
Permanent marking	Method	• Pigtail stakes/Rebar (2 per plot) • Plot ID on aluminum tags (scribed)	• 20 cm long rebar with 7x7cm steel plate welded to top and painted blue (2 per plot) • Plot ID on top of steel plate – marked with paint pen
	Features	• Light-weight	• Long-lasting
	Issues	• Pins pulled out or bent by animals • Sometimes difficult to push down into stony soil	• Pins pulled out by animals • Difficult to push down flush with ground in stony soil • Heavy (~ 500 grams each) • Rebar is easily oxidized, potentially

Feature	Assessment	Photoplots	Line-intercept (hoop)
			influencing SNR
	Improvements	• Use slightly larger pigtails, or pigtails with barbs on the end	• Use thinner rebar, or galvanized angle-irons instead • Put barbs on the end
Lichen abundance	Method	• Photographs of permanently marked 75 cm x 75 cm (or other sized) square on ground • % cover of lichens analyzed using image analysis software	• Measuring length of each lichen/moss along 130 m line intercept line at a width of about 2 mm (width of the ruler on the t-square)
	Features	• A photographic history of change on permanently marked plots (can see/confirm changes that data is showing) • Objective measure of lichen abundance (image analysis software)	• Can identify lichens/mosses to species/genus • Objective measure of lichen/moss abundance (length along line)
	Issues	• image analysis software can only focus on light coloured lichens (i.e. Cladinas; some of the darker Cladonias cannot be analyzed) • image analysis software cannot distinguish between lichen species • image analysis software cannot analyze % cover of other vegetation • provides a 2 dimensional measurement of lichen abundance (i.e. no measure of volume/mass) • image analysis cannot distinguish between vegetation/lichens covered by plants versus vegetation/lichens that are absent • taking photos in rainy conditions • taking photos in sunny conditions	• method is based on a line about 2mm in width; small changes in position of the measuring line can potentially result in variation in the length of lichens and mosses • provides a 2 dimensional measurement of lichen abundance (i.e. no measure of volume/mass) • competing species measured as % cover in whole hoop rather than just on line intercept so vegetation changes in the hoop as a whole may not reflect vegetation and lichen changes on the line intercept
	Improvements	• % cover of lichen and other species can also be done visually within the 75 cm x 75 cm frame • use waterproof cameras • use a tarp/umbrella to shade photoplots	• add key vegetation species to line intercept (increases amount of time needed) • take a photo of the hoop/transect line (also helps in re-establishing pulled pins)
Vegetation	Method	• % cover of selected species in 75 cm x 75 cm frame • % cover of all species in 7.98 m plot	• % cover of all species within hoop
	Features	• 75 cm x 75 cm frame – good sized plot for estimating % covers • Compatible with standard forest mensuration	• Hoop is a good sized plot for estimating % covers • fixed hoop edge makes it easier to estimate % covers
	Issues	• visual estimates are subjective	• visual estimates are subjective
	Improvements	• Have multiple observers for each plot or rigorous standardized training • Decrease large plot size	• Have multiple observers for each plot or rigorous standardized training

of successional stands (Coxson and Marsh 2001, Morneau & Payette 1988). Though they represent a relatively minor component of the ground layer by percent cover, *Oryzopsis* spp. can be regularly distributed throughout these habitats in later seral stages (Sulyma and Sulyma 2006, Sulyma 2001). Even minor presence in the early years after fire should corroborate that the vegetation community is on track for producing conditions suitable for terrestrial lichen communities.

Despite the fact that *Shepherdia canadensis* and *Spiraea betulifolia* associate with more than just the driest pine-lichen sites, they are an important component of any suite of indicators during the initial post-burn surveys. Both species are considered associates of the xeric site series in mature stands (Delong et al. 1993) and their presence in early years after fire can be used as a measure the success of prescribed burning.

Vaccinium caespitosum was consistent in abundance between the 2007 survey (Cichowski and Williston 2008) and the 2008 survey; most abundant in mesic plots, followed by xeric plots and submesic plots. While no obvious linear relationship was present, the pattern is reminiscent of Tilman's (1985) "interactively essential" resource consumption, whereby the increase of a single resource can increase plant growth almost independently of another resource. However, the variability in abundance was quite high within each site series group, ranging from zero to more than eight percent for both the mesic and xeric sites; it will likely be relatively abundant when present, but may not always be present.

Cornus canadensis, *Calamagrostis canadensis*, and *Carex* spp. all showed similar patterns of abundance in the different site series groups to *V. caespitosum* for 2008, but were not consistent with patterns observed in the 2007 survey, where the two former tended to be more abundant in mesic and submesic plots, and the two latter species showed similar abundance throughout all plots (Cichowski and Williston 2008). The patterns of abundance we observed may simply be an artefact of the sample size and plot locations, and until more plots can be examined their relative abundance should not be indicative of specific post-burn recovery conditions. Rather, they fall into a group of species that will likely be present but are not alone sufficient for influencing predictions of successional trajectory; observers can expect to see these species frequently throughout mesic, sub-mesic, and xeric sites, but their absence doesn't necessarily indicate a poor rehabilitation outcome.

Our data, and those of Cichowski and Williston (2008), agree with Wang and Kemball's (2005) observations that sprouters or seed-bankers dominate in the first or second post-fire years after a severe burn, and we anticipate this will be followed by a rise in dominance of invaders afterward. If this trend occurs, the invader species should begin to increases in abundance more rapidly than sprouters and seed-bankers. The high amount of bare soil and the patch structure observed for many species further supported this developmental trend in that an abundance of available habitat still exists, and many of the small patches of invader species had an abundance of propagules available for dispersal that could quickly increase the patch sizes in coming seasons.

Overall, post-fire vegetation in prescribed burns of pine-lichen forests will likely be dominated by sprouters, such as *V. caespitosum*, *L. borealis*, *Shepherdia canadensis* and *Spiraea betulifolia*, in the first post-fire year, with approximately five to fifteen percent total plant cover. In the second year, the total plant cover may be two to three times the amount seen in the first year, and invader species (*Oryzopsis* spp., *Calamagrostis canadensis*, *Epilobium* spp.) will become co-dominant with sprouters, reaching an average of approximately fifteen percent cover by themselves. The relative abundance of invaders will probably continue to rise until approximately year four or five (Wang and Kemball 2005), after which it will slowly begin to decline. Sprouters will likely asymptote at a similar time, because tree seedlings begin to dominate resource extraction between five and ten years after fires (Anderson 2003). These changes may be less noticeable on north-facing slopes, where higher initial litter layers will result in the persistence of more favorable conditions for sprouters and seed-bankers, and invaders may never obtain more than twenty percent total cover. Deeper litter layers do not necessarily preclude the future dominance of lichens on north-facing slopes, but it is unlikely that lichens will successfully colonize areas with litter layers of deeper than five to six centimeters and maintain dominance, as these tend to be more favourable to vascular plants and bryophytes (Brown et al. 2000).

Comparison of Long-term Vegetation Monitoring Methods

Both line-intercept and photo-plot procedures yield quantitative assessments of species or functional group abundance, but they differ in several important ways – the most obvious being their dimensionality. The line-intercept method yields a unidimensional abundance measure (distance occupied on a line), whereas the photo-plot method is two-dimensional (area occupied within plot). Both procedures require a height, or vegetation thickness measurement to be taken which can be use to infer productivity or total abundance.

Both methods can be reproduced in a relatively accurate manner between surveys (compared to larger plots). However, data collected using the photo-plot method will be less prone to sampling error between years than the line-intercept method, because it will not be heavily influenced by the marginal (<2cm) plot positioning errors which are likely to occur, even when using permanent plot markers. Marginal positioning errors for a line transect could result in surveying an entirely different line than previous surveys, though the likelihood of floristic composition changing significantly with a marginal shift in position is probably quite low. The photo-plot method would also show less error between surveys than the line-intercept method because it is not as heavily influenced by bias due to multiple observers; use of the same computer software to assess the percent cover ensures consistent and comparable measurements between years. Rigorous training and limiting the number of surveyors in a given year would improve consistency within and among field seasons for the line-intercept method, but can be logistically difficult to ensure.

Species-specific or individual-specific changes are very difficult to detect when using software-based estimations of cover with photo-plot monitoring, because currently used software programs are unable to distinguish between similarly coloured species. The line-intercept method is more accurate in classifying abundance at the species level, and even goes beyond simple species classification in terms of assessing the vigour of individuals over time. However, the taxonomic precision of photo-plot methods may be increased by manually digitizing photos with GIS-type software and by determining species identity in the field, such that researchers already know what species are present when digitizing the photos.

The two methods differ in terms of the amount of equipment and portability of equipment required in the field. The equipment required for the photo-plot method is less cumbersome because it requires only the plot frame, tripod, and camera, of which the former two can be collapsible and made of light-weight materials. In contrast, the line-intercept method requires levels, metal rods, and tapes in addition to a welded metal plot frame. The size and dimensions of the line-intercept plot frame may restrict it from being used at remote sites due to logistics of access. The time required for surveys of the different plots is probably comparable in terms of set-up, but may differ for actual data collection and later analysis. Taking photos is faster and ergonomically easier than surveying line transects, but may ultimately be less expedient because of the time-requirements associated with digitizing photos manually for higher taxonomic resolution.

Considering the importance of the proximity of propagule sources when studying the recovery of dispersal limited species such as lichens (Heinken 1999), it would be useful for investigators to include some measure of dispersal potential in post-fire surveys; for example, documentation of the spatial arrangement of propagule sources on the burn perimeter or in unburned patches. Previous investigations have rarely been able to record or account for distance from propagule source as a variable that influences lichen dispersal; without use of molecular identification techniques it has been difficult to discern how or from where a given patch of moss or lichen originated once it has developed (Nash 1996), especially considering the lack of published information on dispersal ability and rate of propagule success. Even species that are not dispersal limited can deviate from habitat-based predictions of prominence when dispersal potential is not adequately accounted for (Walker et al. 2006). Proximity to intact forest edges or patches should also be carefully documented in post-burn monitoring because it can influence the development of plant communities through changes to the edaphic and micro-climatic environment of the disturbed habitat, an influence commonly known as ‘edge effects’ (Harper et al. 2005). A network of transects and small plots in a regularly spaced grid throughout the burn could provide the necessary spatial data from which both effects can be investigated. The patterns of reestablishment should be indicative of both the dominant dispersal vector and, to a lesser extent, the source, for both lichens and early seral species.

Plot size for stand level sampling should be determined based on the scale at which habitats are most homogeneous, with consideration for how frequent the indicator species are on a given site. For example, a 3.99 m radius (50 m^2) plot is more appropriate than a 7.98 m^2 radius (200 m^2) plot if topography varies significantly on the scale of 10 meters or less, and species of interest are frequently detected. The plot size used in 2008 (100 m^2) should be sufficient for continued post-burn surveys of similar forests, as the trees tended to be small and closely spaced. For stand or site level measurements, plot density must be adequate to address the variation of the elements being assessed. Once determined, a systematic grid with point distances that are calculated as a function of the target density of plots and the total stratum size will provide representative coverage for survey purposes.

RECOMMENATIONS AND CONCLUSIONS:

Several unknowns warrant further exploration with post-burn monitoring. Specifically, two questions that arose from this investigation are: 1) Do differences in dispersal range or dispersal methods among species affect post-burn composition and abundance and

2) Is there a dominant growth strategy for certain species in the region, such that they can be reliably classified as belonging to a single functional group, or should species be recorded by growth strategy at the plot level? Additionally, it may be worthwhile to include some detailed assessments of cryptogamic (a.k.a. biological) soil crust formation in a sub-set of plots, as other studies have found that the recovery of these organisms can either facilitate or inhibit the re-establishment of other biota (Deines et al. 2007), or it may be indicative of the successional trajectory in itself (Ponzetti et al. 2007).

Because the associations between plant species and favourable lichen habitat are based on examination of a single site, these associations should be treated as hypotheses, rather than as an exact successional model for post-burn monitoring. Regular monitoring will be an important part of any lichen recovery strategy, because changes in plant composition and abundance can occur quickly within the first few post-fire years (Wang and Kemball 2005). Specifically, we recommend surveys from post-fire in years 1 and 3, followed by General reconnaissance level assessments should be completed on a 3 to 5-year interval until noticeable changes occur, which would prompt a more detailed survey.

Both of the processes evaluated for long-term vegetation monitoring use detailed procedures to quantify the presence and amount of a species in a small plot. Yet, within a specific region, standardization is still desirable because local expertise and logistic considerations can be better suited to one technique over another. Within the Vanderhoof forest district we recommend the use of photo-plots. This more closely merges with a forestry-lichen adaptive management project (Sulyma and Sulyma 2006) and with work conducted to the west by Cichowski and Williston (Williston et al. 2006).

The implementation of long-term monitoring should begin before a prescribed burn takes place, in order to compare the sites to pre-burn conditions. This will assist in determining which species belong to each growth-strategy functional group for those species that were difficult to classify, and in measuring burn severity by use of more than just tree characteristics (for example, depth-of-burn pins). Sampling intensity must be adequate to provide a basis for future comparisons but it is not necessary to establish permanent plots until the first year after the burn where plot locations may be stratified by burn intensity.

A plot size of 100 m^2 should be appropriate for most lodgepole pine forests, but care should be taken to ensure the plot size is adequate for the site of interest. If tree densities are low, larger plot sizes (ie. 200 m^2) may be used. The density of sampling quadrats should be 30 to 40 per stratum (McCune and Lesica 1992). The spatial location of plots relative to each other, to unburned patches, and to the edge of the intact forest should be documented during surveys so that patterns of recolonization can be studied as the community develops. This will assist not only in helping to understand which species disperse primarily from outside of the burn, but also in designing future burns to maximize the speed of lichen recovery.

We would expect after prescribed burning of a lichen site to see: the presence of *Arctostaphylos uva-ursi*, the high abundance of *Spiraea betulifolia* and *Linnaea borealis*, and the general increase in the ratio of invaders to sprouters during the first two to three years after the burn.

In Appendix C, we provide a protocol to guide pre- and post-treatment effectiveness assessments for MPB-attached ungulate winter ranges sites chosen to receive rehabilitation.

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APPENDIX 1: RAW PLOT SUMMARY DATA

Two-dimensional percent cover of substrate and plant species, by ecological site series, observed in a survey of post-fire vegetation indicators, Vantine Creek Burn, British Columbia, 2008.

	Ecological Site Series																										
	SBPS mc01a			***SBPS mc01b***				***SBPS mc02***																			***SBPS mc03***
	S1P3	**S1P5**	**S1P6**	**S2P2**	**S2P3**	**S1P1**	**S1P2**	**S4P1**	**S4P4**	**S4P2**	**S2P0**	**S3P1**	**S3P2**	**S2P1**	**S4P3**	**S1P4**	**S1P7**	**S1P8**	**S1P9**	**S2P4**	**S3P3**	**S3P4**	**S5P1**	**S5P2**	**S5P3**	**S5P4**	**S4P5**
Substrate																											
Cryptogamic crust)	3.0	10.8	10.0	3.4	5.0	20.0	3.4	5.4	8.2	2.8	8.3	3.2	3.0	0.8	2.0	10.2	7.0	10.4	4.2	6.2	0.8	2.5	2.3	2.8	3.4	2.5	5.2
Mineral soil	10.2	9.2	11.6	5.0	2.3	7.5	27.0	34.0	25.0		1.0	2.5	4.4	4.0	15.2	5.4	11.2	5.8	11.4	6.5		3.3	3.3	45.8	11.4	29.4	7.5
Charcoal	9.2	8.6	8.6	7.0	6.4		10.0	6.6	4.2	9.6	5.8	6.0	8.2	6.6	10.8	8.2	3.6	7.0	4.2	10.2	6.6	15.8	8.8	7.0	8.2	7.6	5.8
Coarse woody debris	9.6	7.8	13.3	21.8	13.6	6.2	4.0	19.0	19.7	15.0	8.5	15.8	9.4	7.3	17.7	3.3	10.5	6.7	22.3	6.3	11.0	15.8	12.0	12.8	4.7	8.0	20.5
Litter	58.8	37.0	35.0	59.0	50.0	51.0	39.0	21.2	29.6	26.0	60.4	43.0	53.0	69.0	39.0	41.0	38.0	48.6	32.0	50.6	76.0	49.6	36.8	14.3	39.6	31.0	53.4
Bare rock	4.0	11.2	7.8	4.0	0.0	2.0	2.8	5.4	3.2	14.6	0.0	4.5	0.0	0.0	5.5	7.5	2.8	4.0	7.2	8.3	0.0	4.0	0.0	5.5	2.0	0.0	24.0
Species																											
Achillea millefolium		0.1	4.0													0.1	4.1	0.1									
Agrostis scabra												0.1	0.1														
Arctostaphylos uva-ursi												24.0		3.7	19.0												
Arnica cordifolia			0.8																0.1								
Aster sp.																	7.0										
Calamagrostis canadensis		8.0	0.1		0.1	4.1									3.1	7.3	1.7						5.0		9.0	5.0	
Carex spp.	2.0	1.8	0.1		0.1	0.4	1.6	0.1	1.0	1.0	0.6		1.3		0.1	1.0	1.1	3.0	0.7		0.1	0.1	0.7	9.3			
Castilleja mineata																0.1											
Cetraria sp.												0.1															
Cladina mitis												1.1															
Cladina rangiferina												2.0															
Cladonia spp.												1.0															
Cornus canadensis	4.1	3.0	4.5	0.1		2.8	2.0	3.0	3.0	5.3				3.0	5.5	7.0	2.4	1.5	4.0	1.1	2.0	1.8	3.3				4.5
Dicranum polysetum											1.0	1.0															
Epilobium angustifolium	2.6	6.0	12.6	9.0	14.0	7.4	12.3	8.5	3.5	1.5	1.0	18.5	2.0	4.0		5.2	4.5	8.0	4.7	21.0		6.0	12.2	1.1	5.5	15.6	4.5
Epilobium ciliatum															1.4					0.1							
Galium boreale			0.1																								
Geranium bicknellii	0.7										0.1		0.6			0.1		0.1		7.0							
Linnaea borealis	4.5	3.5	1.1	8.5	6.0	1.5	3.5	5.7	4.2	1.5	9.5	5.5	10.0	6.3	11.7		4.7	1.0	10.0	8.0	2.7	7.0	9.0	2.0	4.2	9.0	1.0
Marchantia polymorpha					0.1								0.1							0.1							
Orizopsis asparifolia														1.0													
Orizopsis pungens		9.0							3.0			1.1			1.1		1.4		1.0				3.0	2.0	0.7	0.1	0.1
Orthelia secunda									0.1																		

	Ecological Site Series																											
	SBPS mc01a			SBPS mc01b				SBPS mc02																				SBPS mc03
	S1P3	S1P5	S1P6	S2P2	S2P3	S1P1	S1P2	S4P1	S4P4	S4P2	S2P0	S3P1	S3P2	S2P1	S4P3	S1P4	S1P7	S1P8	S1P9	S2P4	S3P3	S3P4	S5P1	S5P2	S5P3	S5P4	S4P5
Peltigera aphthosa											2.0	0.1															
Peltigera spp.											0.1	15.3															
Pleurozium schreberi											28.0	0.6									0.1						
Pinus contorta	0.4												0.2			0.2	0.1	0.2	0.1		0.1			0.2			
Polytrichum juniperinum		3.1	0.1	0.1	0.1	3.6	0.6	0.6	4.5	1.3		0.1	0.3	0.4	0.1	3.8	4.8	0.8	0.7	0.7	0.1	0.1	4.7	3.4	1.2	0.4	0.1
Populus tremuloides																0.1						8.7					
Rosa acicularis		4.3	6.0				5.3	12.0				1.6	4.3	2.0	1.0	3.0	3.0	9.7							6.3		
Salix sp.		1.0																									
Shepherdia canadensis		5.0	2.8			8.0	2.3	7.0	2.0							7.5	3.0	3.3	4.0				0.1	1.0			1.0
Spiraea betulifolia	4.5	3.5		4.7	0.1	9.0	7.0		11.0	5.3	4.5	2.0	16.5	3.4	6.0	8.0	3.4	13.0	3.5	5.0		7.0	4.8	8.0	14.0		5.3
Vaccinium caespitosum	3.0	4.8	3.2	1.6	2.3	2.0	2.6	2.0	1.8	7.5	0.4			8.8	0.1	2.3	4.4		2.4	5.0	2.5	2.4	0.1	0.1	8.7	2.0	12.0
Viburnum edule																											1.0
Viola spp.															2.0												
Total % plant cover	***21.8***	***53.1***	***35.4***	***24.0***	***22.8***	***38.9***	***37.2***	***38.8***	***34.2***	***23.5***	***47.2***	***73.9***	***35.4***	***32.6***	***51.0***	***45.8***	***45.5***	***40.7***	***31.2***	***48.0***	***7.7***	***33.1***	***42.9***	***27.1***	***49.6***	***32.1***	***29.5***
Species richness	***8***	***13***	***12***	***6***	***8***	***9***	***9***	***8***	***10***	***7***	***10***	***16***	***10***	***9***	***12***	***14***	***14***	***11***	***11***	***9***	***7***	***8***	***10***	***9***	***8***	***6***	***9***

APPENDIX 2: PLANT CLASSIFICATION BY GROWTH STRATEGY

Classification of plant species, by growth strategy functional groups: I = Invader (by seed or spore from unburned areas), B = seed-banker, S = resprouter (from buried roots, rhizomes or buds), observed in a survey of post-fire vegetation indicators, Vantine Creek Burn, British Columbia, 2008.

Species	Strategy	Source(s)
Achillea millefolia	I	Aleksoff 1999, Wang and Kemball 2005
Agrostis scabra	I	Haeussler et al. 2002, Matthews 1992, Sirois 1995
Arctostaphylos uva-ursi	S**	Crane 1991, Rowe 1983
Arnica cordifolia	I/B/S	Haeussler et al. 2002, Reed 1993
Aster sp.	I*	Wang and Kemball 2005
Calamagrotis canadensis	I/S	Haeussler et al. 1990, Haussler et al. 2002, Tesky 1992, Wang and Kemball 2005
Carex spp.	B*	Haeussler et al. 2002
Cornus canadensis	S	Crane 1989, Haeussler et al. 2002, Rowe 1983, Sirois 1995
Corydalis aurea	I/B	Haeussler et al. 2002, Matthews 1993, Wang and Kembal 2004
Corydalis sempervirens	I/B	Haeussler et al. 2002, Rowe 1983, Sirois 1995, Wang and Kembal 2005, Williams 1990
Epilobium angustifolium	I/B	Haeussler et al. 1990, Haeussler et al. 2002, Rowe 1983, Sirois 1995, Wang and Kembal 2005
Epilobium ciliatum	I*	Wang and Kembal 2005
Galium boreal	B/S	Gucker 2005, Wang and Kemball 2005
Geranium bicknellii	B	Haeussler et al. 2002, Rowe 1983, Wang and Kemball 2005
Ledum groenlandicum	S	Gucker 2006, Rowe 1983
Linnaea borealis	S**	Howard 1993, Sirois 1995
Lonicera involucrata	S*	Munger 2005, Wang and Kemball 2005
Marcantia polymorpha	I/B	Haeussler et al. 2002, Matthews 1993
Orizopsis asperifolia	I*	Tirmenstein 1999
Orizopsis pungens	I*	Tirmenstein 1999
Petasites palmatus	I	Wang and Kemball 2005
Pinus contorta	B	Haeussler et al. 2002, Rowe 1983, Sirois 1995
Polytrichum juniperinum	I/B	Fryer 2008, Haeussler et al. 2002
Populus tremuloides	S	Haeussler et al. 1990, Haeussler et al. 2002, Rowe 1983
Rosa acicularis	S/B	Crane 1990, Haeussler et al. 1990, Wang and Kemball 2005
Salix sp.	S*	Anderson 2001, Haeussler et al. 2002, Sirois 2005, Wang and Kemball 2005,
Shepherdia canadensis	S	Haeussler et al. 2002, Walker 1991
Solidago spathula	I*	Sirois 1995, Wang and Kemball 2005
Spiraea betulifolia	S	Habeck 1991
Spiraea pyrimidata	S**	Habeck 1991
Vaccinium caespitosum	S	Tirmenstein 1990

* General strategy inferred from information on genus or similar species and on-site observations
** On site observations assisted in classification

APPENDIX 3: MONITORING PROTOCOL

Assumptions:

- The protocol will be used to monitor vegetation dynamics on sites where a prescribed burn has been implemented for the purpose of rejuvenating the succession pathway of terrestrial forage lichens used by woodland caribou.
- The burn plan for the project has been constructed and implemented as planned and has thereby led to availability of information about the general study area as well as specifics about sample plots within treatment and control sites.
- The protocol would be used to assess vegetation prior to, as well as after, the burn has been implemented.

Applicable Standards:

- Province of BC. 1998. Field manual for describing terrestrial ecosystems. Land Management Handbook 25. Resource Inventory Branch, BC Ministry of Environment Lands and Parks & Research Branch, BC Ministry of Forests. Victoria, BC.
- Trowbridge, R., Hawkes, B., Macadam, A., and Parminter, J. 1989. Field Handbook for prescribed fire assessments in British Columbia: logging slash fuels. FRDA Handbook 001. BC Ministry of Forests, Research Branch and Forestry Canada, Pacific Forestry Centre. Victoria, BC. 63p.
- Habitat Monitoring Committee. 1996. Procedures for environmental monitoring in range and wildlife habitat management. Version 5.0. British Columbia Ministry of Environment, Lands and Parks and British Columbia Ministry of Forests, Victoria B.C. 225pp.

Required Equipment:

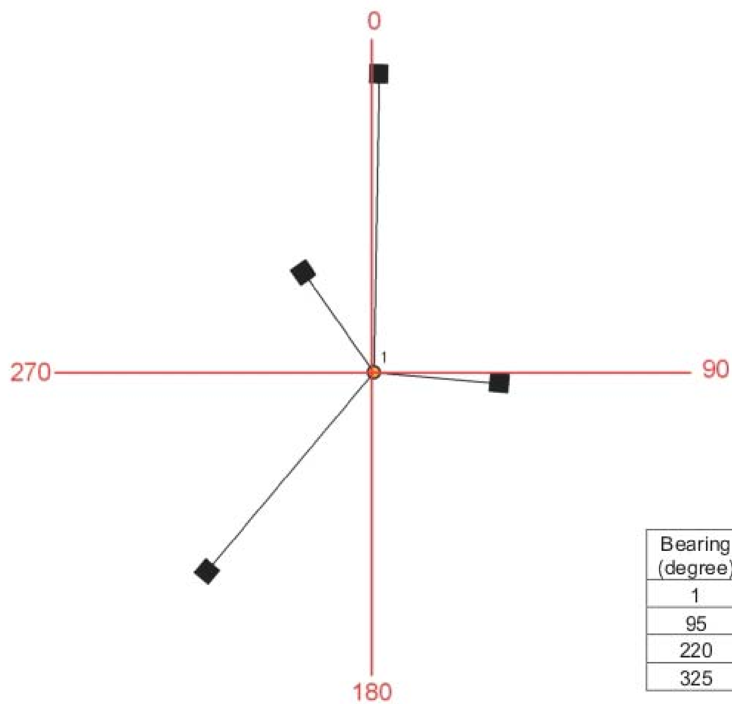
- GPS
- 5.64/3.99 m Plot Cord
- Shovel – for inspecting surface soils
- Loggers/Eslon Tapes
- Data logger/Field notebook – with forms
- Species List
- Map
- Field guides/plant ID guides
- Quadrat frame (0.5m^2 or 0.71m X 0.71m)
- Hand lens/magnifying glass
- Compass
- Clinometer
- Callipers/diameter tape
- Carpenters tape
- Small (15cm) metal ruler
- Go-no-go gauge for measuring debris (See Trowbridge et al 1989 pg 21)
- Camera
- Wooden stakes/rebar for marking plot centres
- Pig-tail pins or rebar for marking quadrat corners
- Numbered tags
- Flagging tape
- Paint

Office Procedures:

- Pre-field planning:
  - The study area should be stratified into control and treatment sample units. Also, stratification of post-burn conditions will result in a minimum of three burn severity strata to be delineated within the treatment sample units.
  - Three types of plots will be developed all referenced to a plot center: circular plots of two different radii and a series of quadrat plots (see plot establishment below). First, develop a systematic grid for establishing plot centers within each sample unit. Grid spacing will depend on the size of sample units where a minimum of 5 plot centers are desirable.
  - Establish 30 to 40 quadrats plots in each sample unit to provide acceptable representation of the vegetation community. The number of quadrats per plot centre will be a function of the target divided by the number of plot centres within the treatment unit.
- Post-field data analysis:
  - All field data should be entered into, and summarized in, Excel Workbooks (Microsoft Corporation, Redmond, Washington) and statistical analysis can be completed using SAS (SAS Institute Inc., Cary, North Carolina). Statistical values presented should include the mean $\pm$ 1 standard error ($\mu \pm$ 1se), or be summarized as frequencies expressed as percentages. All tests should be applied with an $\alpha = 0.05$. Tests for simple effects evaluating the potential influence of site attributes on strata designations can be conducted using one-way ANOVA. Tukey's HSD (honestly significant difference) can be used to express the strength of the statistical findings. Eta (η^2) can be calculated for f-stats where $\eta^2 \approx 0.02$ is considered small, $\eta^2 \approx 0.15$ is medium, and a $\eta^2 \approx .35$ is large (Kirk 1996).

Field Procedures

- Plot establishment
  - Plot centers should be established according to the paper plan. At each plot center, establish a 5.64 m radius plot within which to measure trees and shrub vegetation. Nest within that, a 3.99 m radius plot for measuring vegetation regeneration. The percent cover of herbs, mosses, and lichens can be recorded at a third plot using a modified cluster survey design where 0.5 m^2 (0.71 m X 0.71 m) quadrats are established around each plot centre (Figure A). Quadrats are located by placing them at a random bearing and random distance (<20 m) relative to the plot centre. To ensure equal distribution around a plot centre, the azimuth should be divided into identical sized sections equivalent to the number of quadrats that were to be established at the plot and a random bearing was selected for each section. For example, where 4 quadrats are to be established, the azimuth would be divided into four sections ranging from: 1 to 90 degrees, 91 to 180 degrees, 181 to 270 degrees, and 271 to 360 degrees and a random number picked within the range of each section. Random distances, ranging between 0 and 20 m from the plot centre are also selected. To improve the distribution of the quadrats and to avoid the potential for quadrats to overlap, only one quadrat per plot can be



Bearing (degree)	Distance (metres)
1	15.9
95	7
220	13.9
325	6.8

Figure A. Example of a modified cluster approach for establishing sampling quadrats at each plot centre for the purposes of measuring percent cover of herbs, mosses, lichens, dwarf, and creeping shrubs.

permitted within a ring of one metre of the plot centre, and only two quadrats can be permitted within a ring of three metres.

- Measurements
  - Surveys completed at each plot center will characterize:
    - Vegetation – Tally all trees larger than 12.5 cm diameter at breast height (dbh). Tree dbh, height, index of beetle attack and wildlife tree class should be recorded using standard protocol. An estimate of the crown closure can be made using a concave densiometer following procedures outlined by Trelenberg and Hodder (2006). The percent cover of shrubs less than 10 m tall can be made using an ocular estimate. Undertake a layered survey within the vegetation regeneration plots to record the number of stems by species and health (I.e., live or dead) by the following classes: <1.3 m tall, >1.3 m tall to 7.4 cm dbh, and 7.5 to 12.4 cm dbh. Percent cover of herbs, mosses, lichens, dwarf, and creeping shrubs is measured at the quadrat plots by an ocular estimate and by the photo methods.
    - Down woody debris / fuel loading – At each plot centre, establish two 30 m transects. Measurements along each transect should be based on standard procedures outlined by Trowbridge et al. (1989). The total number of intersections of debris less than

seven centimetres diameter should be recorded in specified transect segments relative to the diameter class being assessed. All pieces greater than seven centimetres diameter are individually tallied collecting species and diameter information for each piece. In the field each piece of fuel greater than 7 cm diameter should be marked with a nail at the intersection with the transect line

- Biophysical site characteristics – At each plot centre site attributes, such as terrain and soil characteristics, should be recorded on Ground Inspection Forms[7] following provincial standards.

Literature Cited

Trelenberg, J., and Hodder, D.P. 2006. The concave densiometer: a reliable method for measuring canopy closure on mule deer winter range. John Prince Research Forest - University of Northern B.C. Fort St. James, B.C.

Kirk, R.E. 1996. Practical significance: A concept whose time has come. Educational & Psychological Measurement 56(5): 746-759.

[7] Information on Ecosystem field forms can be found at the following BC Ministry of Environment web page - http://www.env.gov.bc.ca/ecology/dteif/forms.html (accessed September 4, 2008).