Burning Questions:

Effects of Fire on British Columbia's Ecosystems

Evelyn Hamilton
Julia Chandler
Reg Newman
and
Sybille Haeussler

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Abstract

Key information needs about the effects of fire on ecosystems in BC's interior were derived from our interviews with over 60 decision makers. The "Burning Questions" raised by resource managers encompassed the response to fire of ecosystems and plants in general, groups of plants (e.g., wildlife and livestock forage, invasives) and specific plants (e.g., culturally important medicinal berry species, and intermediate hosts for pine rusts); also considering the implications for reforestation, watershed stability and carbon balance. We attempted to provide answers with the available data about the long-term response of understory plant communities after clearcutting and slashburning in northern and central BC, the monitoring of ecosystem restoration sites in the Rocky Mountain Trench and the response of vegetation to a wildfire. We found that post-fire plant community composition was most highly related to long-term fire history/fire climate and local site moisture and nutrient gradients. Differences in species composition between recently (past 20 years) burned and unburned sites were relatively small, suggesting that these ecosystems are broadly resilient to fire. Management decisions other than whether to burn (e.g., reforestation choices, grazing decisions) may have more impact on understory vegetation.

Burning Questions

- Key Fire Related Information Needs -

General

- Have fire regimes changed over time and if so, how, and what are the likely implications?
- How safe are First Nations and other remote communities with respect to wildfires?
- What values do burned OGMAs still have?

Vegetation

- What are the effects of fire and fire exclusion on understory plant species?*
- Does burning enhance or inhibit trembling aspen encroachment/recovery?*
- Can fire be used to restore/enhance FN food, medicine & cultural plants (e.g., berries, devil's club)?*
- What are effects of fire on invasive or non-native plant species?*

Wildlife

- What are implications of changes in fire regime for wildlife like grizzly bear, moose and caribou especially with respect to forage quantity and quality (e.g., spring forage, berries, aspen/deciduous, lichens)?*
- Have enhancing ungulate populations by burning impacted caribou thru increased predation in the NE?

Watershed

- What are the expected rates of recovery of watersheds after wildfires? Rates of vegetation development?*

Restoration and Reforestation

- How can ecosystem restoration be done in priority wildfire burns?
- What are the implications of severe fires on future reforestation? Rates of natural regeneration? Likely plantation success?
- How does fire damage and/or enhance whitebark pine ecosystems?
- Does wildfire reduce lodgepole pine stem rusts or their alternate host species?*
- Does fire enhance or inhibit Douglas-fir regeneration at northern limits?

*Within Project Scope

Conclusions and Recommendations

- Moderate to low severity prescribed burning is consistent with maintaining ecological values in mesic and wetter spruce, subalpine fir and/or western hemlock communities in sites on mountainous terrain with infrequent high severity wildfire (BCLT1).
- Prescribed burning followed by planting led to successful conifer reforestation on BCLT1sites. Logged and burned sites in the SBS and ICH zones generally provided for good ungulate forage production during the first 20 years post disturbance.
- There was significant cover of berry producing plants in the first 20 years after prescribed burning; differences associated with ecosystem type and fire severity were observed.
- Post disturbance conifer density and herbicide use may have a greater impact on understory vegetation than whether a site was burned or not. Assessment of long-term effects of additional conditions and treatments such as these is recommended.
- Prescribed burning can enhance cover of berry producing shrubs, like huckleberry, over other ericaceous species like rhododendron and false azalea, especially in the ESSF.
- Management to maintain or enhance devil's club should include retention of moist mature forests where it is most abundant as its regrowth on cutblocks can be slow.
- Management for soapberry should focus on drier sites; prescribed burning is compatible with maintaining this species.
- We were unable to detect a significant effect of prescribed burning on the abundance of plant species that are alternate hosts for pine stem rusts (*Comandra* blister rust hosts *Comandra umbellata* var. *pallida* - not present in the database, and *Geocaulon lividum*; and *Stalactiform* blister rust hosts *Castilleja* species, *Melampyrum lineare, Pedicularis bracteosa*, *Rhinanthus minor* and *Orthocarpus luteus* - not present in the database).Dry lodgepole pine community types in plateau landscapes (BCLT 2) have been profoundly impacted by the cumulative effects of recent disturbances including mountain pine beetle, salvage logging and wildfires. Mineral soil and rock exposure on sites that had been burned twice was 2x that found on sites that had only burned once – and was up to 55% on some sites. Further analyses of newly assembled datasets are recommended.
- The condition of grasslands associated with south-facing slopes in the aspen-mixedwood landscape (BCLT3) may be declining and a concerted effort to implement appropriate monitoring is needed to ensure appropriate management practices.
- In the Rocky Mountain Trench of the Southern Interior, restoration burning treatments appeared more successful in the IDF zone than in the PP zone. In both zones conifer regeneration and woody debris was reduced, mineral soil exposure increased somewhat, and some invasive species established on sites where trees were removed, and on sites that were prescribed burned. Development of late seral bunchgrass was slow, likely due in part to consumption by animals. Early intervention to restore these ecosystems before understory tree cover in stands becomes dense is recommended.
- Well planned and resourced monitoring with pre-treatment sampling, controls and exclosures is needed to provide answers to the many important questions related to fire effects.

Executive Summary

We compiled two databases that included vegetation response data from 90 sites throughout the province where fire ecology studies had been carried out. Our Central and Northern Interior database consisted of data gathered from 78 sites or study areas with approximately 3770 plots, 8450 total records (included repeated observations on the same plots) and over 400 plant species. The Southern Interior database had data from 19 treatment units located in 12 sites.

1. Central and Northern Interior

Changes in plant communities caused by prescribed burning and wildfire in the Central and Northern Interior datasets were not substantial enough to shift the burned communities outside of the range of variation found in unburned communities within the same general community type. Vegetation communities found in the BC Interior have been shaped by their past exposure to wildfire: communities with the greatest exposure to fire (drier boreal and sub-boreal community types) possess the most fire-adapted species and can be expected to be most resilient to fire, whereas communities with a history of infrequent fire (wettest ICH and ESSF forests) are more likely to undergo changes following exposure to fire events. Conversely, the effect of several decades of fire exclusion is likely to be most evident in highly fire-adapted plant communities and less evident in communities adapted to long fire-free intervals.

An overview analysis of the Central and Northern Interior database showed that plant community composition was most strongly related to long-term fire history or fire climate and local site moisture and nutrient gradients. Differences between recently burned and unburned sites were statistically significant but comparatively small in relation to broader geographic and site level gradients (2.6% versus 26% of total variation). In the most general terms, our results confirm that these Interior forest ecosystems are broadly adapted to, and resilient to, fire, and that major shifts to alternative ecosystem states following fire are not characteristic of the sites included in the database.

We classified the Central and Northern Interior database (SBS, ESSF, BWBS and a few ICH zone sites) into three broad Community/Landscapes Types (BCLTs) defined by their historic fire regimes:

1. Mesic and wetter spruce, subalpine fir and western hemlock communities in mountainous terrain with infrequent high severity wildfire (BCLT 1)
2. Dry lodgepole pine communities in plateau landscapes with moderately frequent high and mixed severity wildfire (BCLT 2)
3. Aspen-spruce mixedwood and aspen-grassland communities in rolling, low elevation terrain with frequent low and mixed severity wildfires and spring prescribed burns (BCLT 3)

Each of these “landscapes” was well suited to tackling one or more of the Burning Questions.

1.1 Mesic and wetter spruce, subalpine fir (and/or western hemlock) communities in mountainous terrain with infrequent high severity wildfire (BCLT 1)

This BCLT includes a broad range of conifer-dominated plant communities on mesic and wetter sites, at middle and higher elevations in often mountainous terrain with a moist to very wet climate.

Moderate to low severity prescribed broadcast burning is consistent with maintaining ecological values in these ecosystems. Cover of plants associated with mature forests increased over time, forming > 40% of cover by year 20 after a broadcast burn. There was faster recovery and conifer growth in the SBS zone than the ESSF zone, and greater, more persistent deciduous tree and tall shrub cover in the SBS versus ESSF. Ericaceous shrubs were prominent in ESSF by year 20. There was little early seral species cover by year 20. These SBS, ICH and ESSF ecosystems proved fairly resilient to a single burn.

Prescribed burning generally led to successful conifer reforestation on wetter SBS, ICH and ESSF sites. On clearcut and slashburned sites in BCLT 1 monitored for 20 years[1], conifer cover was similar in year 1 (mean < 0.2%) and increased over time in all zones. It was greatest in the SBS by year 20 (42%) followed by the ICH (35%) and the ESSF (13%). In the SBS the tree species planted in young forests (e.g., *Pinus contorta* and *Picea glauca x engelmannii* (Interior spruce)) are more like those typical of mature forests than is the case in the ESSF and ICH, where advanced regeneration of subalpine fir, western hemlock or western redcedar is usually eliminated by prescribed fire**.**

Burned sites provided for good ungulate forage production in the first 20 years after logging in the SBS and the ICH zones. In BCLT 1 cover of deciduous trees and tall shrubs was low until after year 10 when it increased considerably. By year 20, these species formed a significant part of the plant community in the ICH and SBS while in the ESSF their cover was very low. Aspen (*Populus tremuloides*) and paper birch (*Betula papyrifera*) were the dominant deciduous tree species while species of *Salix, Alnus* and *Sorbus* were the main tall shrubs. These species provide forage for ungulates including moose, elk and deer. Burned SBS sites, in particular, provide significant amounts of these forage species. Moose browse species increased rapidly in the first 5 years after burning and continued to increase over the 20-year period in most site series in the Swiss wildfire.

Good cover of berry producing plants was found on burned sites in the first 20 years after disturbance – differences associated with ecosystem type and fire severity were observed. Conifer density and use of herbicides may have a greater impact on berry producing plants than whether a site was burned or not. For berry producing plants and other grizzly bear forage we found that the response trajectories in BCLT 1 varied significantly by plant association and whether or not the site was burned. Moist, rich Devil's-club and Oak-fern plant associations tend to have openings or gaps and a very diverse range of forage plants. The amount of forage continues to increase over longer time periods than in mesic *Vaccinium*-Moss plant associations where abundance stabilizes or declines following overtopping by conifers. Unburned but harvested sites tend to have more forage initially than burned sites, but abundance declines more rapidly**.** Note, however, that abundance of berry producing plants is not a reliable predictor of berry abundance as plants can be abundant yet not have many berries.

[1] These sites are described in Chandler et al. 2017.

A single management regime cannot be recommended for optimum grizzly bear forage management – some sites should probably be prescribed burned for long-term berry production, whereas others should be left unburned to provide herbaceous forage and short-term food supplies. Likewise, temporal and spatial staggering of disturbances so that patches are available at various successional stages to maximize forage diversity in the context of life history traits and range size is recommended*.* In BCLT 1 burning after clearcutting reduced grizzly bear forage production for the first few years after the treatment, but the effect was longer-lasting for late-season forage (berry producers) than for early season forage plants (grasses, ferns, herbs). This study did not investigate the nutritional quality of the forage or berry abundance. Reforestation choices (e.g. spacing and species of tree planted) may greatly affect forage production, and habitat quality (such as hiding cover) for grizzly bears in the second decade and subsequent decades after harvest.

Fire can enhance cover of berry producing shrubs, like huckleberry, over other ericaceous species like rhododendron and false azalea, especially in the ESSF zone**.** Shade tolerant ericaceous shrubs[2] increased after clearcutting and slashburning[3], particularly in the ESSF zone where ericaceous cover was at least 37% at 20 years compared to 5-6% in the ICH and SBS.

Management to maintain or enhance devil's club should include retention of moist mature forests where it is most abundant as its regrowth on cutblocks can be slow. Devil's club provides fruit for birds and bears and is a culturally important plant for many BC First Nations: roots and stems are used to make medicines. It is not well adapted to open environments left after logging. Devil's club cover and abundance in the first decade post- logging was low to nonexistent, regardless of whether the sites were slash burned or mechanically site prepared. It was most abundant on subhygric to mesic sites.

1.2 Dry lodgepole pine communities in plateau landscapes with moderately frequent high and mixed severity wildfire (BCLT 2)

Dry lodgepole pine community types in plateau landscapes across the central Interior have been profoundly impacted by the cumulative effects of recent disturbances. A high priority for further work will be to analyze newly assembled tree regeneration datasets. In BCLT 2 we found that these submesic to xeric lodgepole pine community types in plateau landscapes across the central Interior have been profoundly impacted by recent major wildfires, many of which overlapped. Exposed soil and rock increased to 60% ± 20% on sites exposed to two successive wildfires (compared to 10% ± 5% on sites burned just once) creating extreme risks for soil erosion and non-native species invasion, notably hawkweeds. The regrowth of caribou forage lichens was extremely slow with ≤ 0.2% cover approximately 10-12 years after wildfire, compared to 20-40% in unburned plots. Artificial "reseeding" of lichens is now being tested at several locations to speed the recovery process, and our data analysis supports the need for such interventions. Lodgepole pine ecosystems are, in many respects, highly resilient to wildfire, but multiple fires occurring within a decade have the potential to cause losses in valued resources such as huckleberries and blueberries that generally flourish after a single wildfire**.**

[2] Included: *Vaccinium ovalifolium*, *V. membranaceum*, *V. myrtilloides*, *V. ovalifolium*, *V. parvifolium*, *V. caespitosum*, *V. alaskaense*, *Menziesia ferruginea* and *Rhododendron albiflorum*.
[3] These sites are described in Chandler et al 2017.

Soapberry (Shepherdia canadensis) is an important food plant for First Nations and for some wildlife. It is most common in drier SBS and BWBS sites. Soapberry cover appeared to increase somewhat over time since disturbance on burned and unburned sites. No significant difference between cover or presence on burned compared to unburned sites was apparent in the sites were examined.

1.3 Aspen-spruce mixedwood and aspen-grassland communities in rolling, low elevation terrain with frequent low and mixed severity wildfires and spring prescribed burns (BCLT 3)

The condition of grasslands associated with south-facing slopes in the aspen-mixedwood landscape is declining and a concerted effort to implement appropriate monitoring is needed to ensure appropriate management practices. In BCLT 3 there is a growing consensus that aspen forest health, moose browse, and the condition of grasslands associated with south-facing slopes in the aspen-mixedwood landscape are deteriorating. In general, monitoring data from prescribed burns carried out in this landscape are too inconsistent, short term and lacking in detail and untreated controls to determine whether or not this growing consensus represents real trends and if greater use of fire improves or worsens such conditions. Answers seem to vary greatly depending on local ecological conditions and, as in the Southern Interior analysis, on grazing and browsing by wildlife and domestic livestock. A concerted effort to develop and implement appropriate monitoring is needed. A relatively recent shift from wetter to drier summers across central BC may be complicating our ability to use older datasets and aerial imagery to assess current and future trends.

1.4 All BCLTs

We were unable to detect a significant impact of prescribed burning on the abundance of plant species that are alternate hosts for pine stem rusts. We also looked at response of alternate hosts for stem rusts that affect pines including false toad-flax, members of the Orobanchaceae family and *Ribes* spp. found in all three BCLTs. False toad-flax (*Geocaulon lividum*) is an alternate host for comandra blister rust that affects lodgepole and ponderosa pine. Members of the Orobanchaceae family including scarlet paintbrush are alternate hosts for stalactiform blister rusts that also affect these pines[4]. *Ribes* species are alternate hosts for white pine blister rusts that affect white, limber and whitebark pine.

2. Southern Interior (Rocky Mountain Trench)

Restoration burning reduced conifer regeneration and woody debris providing for more open and potentially fire-resistant sites. Mineral soil exposure increased somewhat, and some invasive species were observed. Development of late seral bunchgrass has been slow, likely due in part to consumption by animals. Restoration of bunchgrasses seems more successful in the IDF zone than in the PP zone. Well planned and resourced monitoring with pre-treatment sampling, controls and exclosures is needed to provide answers to the many important questions related to fire effects. Burning reduced cover of young Douglas-fir regeneration (<1.3m height) by 53%-100%. Objectives regarding promotion of grassland plant communities have not yet been achieved, even on sites monitored for 14-17 years post-fire. Calamagrostis and other shade tolerant species were observed post burn - but little or no bunchgrass characteristic of the zone. It is suspected that even under ideal conditions it will require 20 or more years for late seral bunchgrass species to become dominant and

[4] These 4 members of the Orobanchaceae family were observed in the database.

may not ever without intervention (ADD REF). Grazing by livestock and wild ungulates is almost certainly interfering with plant succession.

Prescribed fire increased the amount of exposed mineral soil to some degree at all sites. Exposed soil creates conditions suitable for the establishment of plant seedlings, either from previously buried seed or seed transported from off-site sources. Exposed soil is therefore necessary for new plant establishment of desirable species, but also increases the risk of infestation by non-native species. A summary of 21 sites showed that the most common invasive plants that continue to increase after prescribed fire are of little or no concern (e.g., dandelion, black medic, yellow salsify). There are two FRPA-listed invasive species (Sulphur cinquefoil and St. John’s wort) on three sites that are increasing at trace levels.

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1. Introduction

Fires have burned British Columbia's (BC) forests and grasslands with varying frequency in the past (Parminter 1991). First Nations used fire as a management technique for thousands of years and since the late 1800s settlers burned to clear land. By the mid-1960s prescribed burning of cut-block slash was done extensively to reduce fire hazard and improve reforestation success. Slashburning remained a widely used practice until the 1990s when alternative site preparation techniques became more widely used. Burning to enhance wildlife habitat has been done for many years, especially in the northern regions of the province where guide outfitters seek to maintain or enhance ungulate populations. In the southeastern part of BC, burning to maintain forage for livestock and wildlife and restore ecosystems has been done for decades.

Research to determine the effects of fire on grazing lands in BC began in the 1920s. In the 1980s the BC Forest Service, in partnership with the Canadian Forest Service and universities, began a research program to determine effects of prescribed fire on site productivity and reforestation. John Parminter wrote many papers on expected post-wildfire successional pathways in the northern regions of the province in response to requests by fire managers for information to support decision-making regarding fire suppression.

Various scientists established long-term research installations in the 1980s and Chandler et al. (2017) synthesized the 20-year results from 16 of those sites located in the montane and subalpine zones in central BC. Few additional long-term studies of slashburning were established after 1990 although a few chronosequence studies were published. Some related synthesis was done (e.g., autecology literature review by Haeussler et al. 1990), but to date there has been no comprehensive analysis of fire effects studies in BC. However, research into effects of "restoration" or habitat enhancement burns began several decades ago and has continued, with new monitoring sites being established annually. Data has been consolidated into various databases[5] [6] and analysis of some of this data has been published (e.g., Berg 2006, Page 2014).

There is an extensive body of research on fire effects in the US and a recent publication summarized the results of some of this work (Bartuszevige and Kennedy 2009). The United States Forest Service Fire Effects Information System[7] provides a comprehensive repository for this knowledge.

In 2017 BC experienced an unprecedented wildfire season with larger and more severe fires than those that have occurred in the past (Nicholls and Ethier 2018). Questions regarding the effects of fire and use of prescribed fire have arisen. These range from the value of using prescribed fire to reduce flammability of the landscape to catastrophic fires to the value of prescribed burning to restore or enhance species and ecosystems.

We identified many sources of data on response of BC ecosystems to fire. These included individual studies and larger databases. We consolidated those data, much of which had no established repository

[5] Rocky Mountain Trench Ecosystem Restoration Program (RMTERP) database consists of 24 ecosystem restoration monitoring sites sampled between 1998–2016 in the Rocky Mountain Trench See www.trenchsociety.com
[6] NE burn monitoring database http://www.env.gov.bc.ca/perl/soft/dl.pl/20180119123550-14-gp-45500fda-8f81-4344-a75e-f2e8d1ea?simple=y. Contact Kristen Peck , Ministry of Environment & Climate Change Strategy, Fort St. John, BC
[7] USDA Fire Effects Information System https://www.feis-crs.org/feis/

(e.g., graduate theses) into a database and used it to answer some of the key “Burning Questions” related to the effects of fire. Answering many of these questions was beyond the scope of what we were able to do in this project –- given the limited availability of data and restrictions in terms of time, resources and expertise.

2. Methods

2.1 Database compilation

We compiled two databases that include vegetation response data from sites throughout the province where fire-ecology studies have been carried out. The first being the Central and Northern Interior database and the other is the Southern Interior (Rocky Mountain Trench) database. Related information in the form of research publications that described the individual studies and their results were also compiled. The database is a matrix that includes plot location and description of the plots and plant species cover values (% cover). Plots are in rows and environmental data[8] and species data are in columns. The data was pooled with data we compiled in previous years. The Central and Northern Interior database includes data from 78 sites or study areas, some of which are repeated measures studies and others were sampled in chronosequence studies[9]. There are approximately 3770 plots, 8450 records (1 plot x 1 year of observation) and over 400 plant species in the database. The Southern Interior database has data from 19 treatment units located in 12 sites; there are 195 macroplots, each including a number of transects and subplots. All are repeated measures sites.

2.2 Study Area

Data was gathered from 78 sites from across the Central and Northern Interior and 12 sites from the Southern Interior's Rocky Mountain Trench of British Columbia (Figure 1).

[8] Site and environmental variables for each plot include location (latitude, longitude, easting, northing, UTM zone), elevation (m), BEC zone, BEC site series, block id, plot id/number. Site history variables include year sampled, time since last stand destroying fire, year timber cut or vegetation cleared, type of timber cutting or vegetation clearing (clearcut, sheared, mechanically cut, none), year burned, type of burn (prescribed spring burn, prescribed fall burn, pile and burn, wildfire), year planted, and species planted or introduced (i.e. trees, lichens).

[9] In repeated measures studies the vegetation in plots is characterized, e.g., % cover of each species is estimated repeatedly over time after a treatment. Chronosequence studies attempt to characterize changes over time by sampling different aged sites belonging to the same ecosystem type.

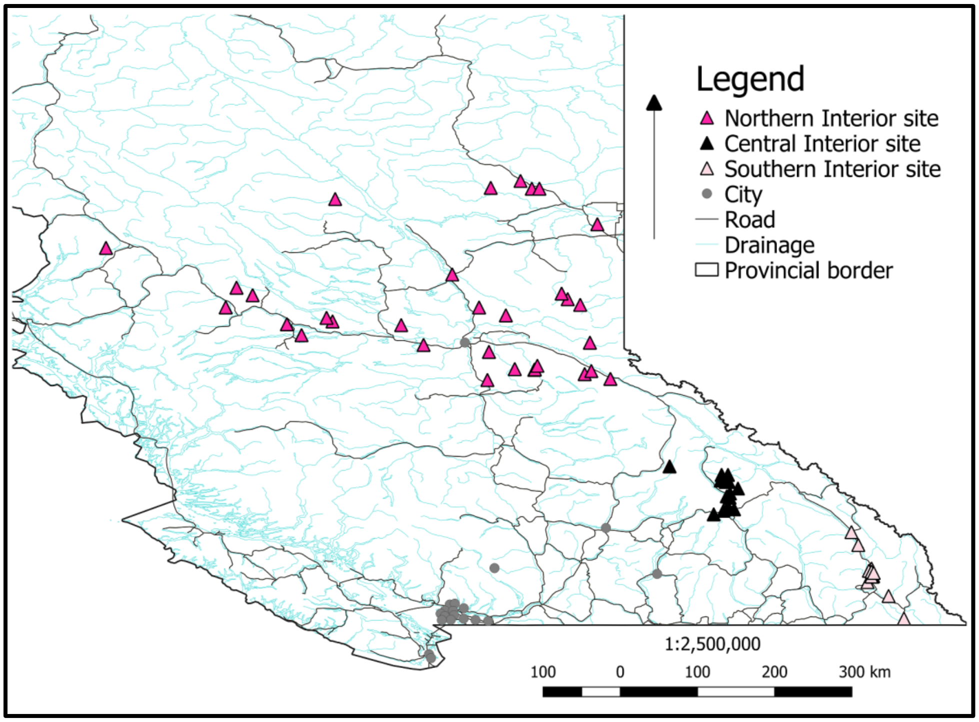

Figure 1. Location of sites included in the study.

2.3 Database and datasets

Subsets of the Central and Northern Interior and Southern Interior databases were used for different analyses. Data from Hamilton and Yearsley 1988 was also used for one analysis (see Appendix 1 for details).

2.4 Analysis

We developed an analysis plan based on the key information needs, thinking about which Burning Questions were within the scope of this project given the nature of the data available and the project timeframe. Our analysis was a combination of data analysis and literature review. Multivariate ordination techniques including non-metric multi-dimensional scaling (NMDS) and partial redundancy analysis were used to identify major plant community trends and subgroups of data for subsequent analyses; and multi-level mixed effects model selection, and other statistical and graphical procedures were used to examine responses over time.

NMDS ordination overlain with a 20-column environmental matrix was run with PC-ORD version 6.19 (McCune and Mefford 2011) using a Sorensen's distance measure, 6 initial dimensions, random starting coordinates, and 250 runs each with real randomized data, stepping down to a 3-dimensional solution with stress of 0.137.

Partial redundancy analysis with the vegan package (v.2.4.1; Oksanen et al. 2016) in R (v. 3.3.2; R Core Team 2016) was then used with a Hellinger transformation of species abundances (Legendre and Gallagher 2001) on the same datasets to separate variation in plant community composition associated with geographic covariables from variation associated with succession after fire (Borcard et al. 2011).

We used forward selection to identify the most powerful geographical explanatory covariables and then constrained the ordination with two variables: burn severity and stand age (normalized with a power transformation of $\lambda = 0.25$).

Within each BCLT we addressed several of the Burning Questions by developing mixed effects models for plant indicator groups or species with the "nlme" (Non-linear Mixed Effects) package (v. 3.1) in R (v. 3.3.2) (Pinheiro et al., 2016; R Core Team 2016). In each case we modeled changes in percent cover with time since disturbance (logging, wildfire and prescribed fire) and tested for significant differences in the trajectories by disturbance type or severity, and site quality (plant community, soil moisture (SMR) or soil nutrient regime (SNR)). Variables were normalized with a power function transformation, if needed. We selected the best model, with and without interactions among the three fixed effects (stand age; disturbance, site quality) based on the lowest AIC and satisfactory distribution of residuals. Random factors for our models were plots nested within (blocks and) sites. We used a continuous autoregressive correlation structure (corCAR1(); Pinheiro and Bates 2000) to address temporal autocorrelation associated with repeated measurement of plots over the years, or in some cases the year of sampling was included as a random factor nested with the plot. A constant variance function (varIdent; Pinheiro and Bates 2000) was also occasionally used to address heteroscedasticity.

Time constraints prevented us from developing predictive models for many important variables and questions. We have included some graphical presentation of results and have summarized results presented in related studies conducted by the authors and other researchers to partly address these issues.

3. Results and Discussion

3.1 Central and Northern Interior Sites – all BCLTs

Data from 49 Central and Northern Interior sites (ICH, ESSF, SBS and BWBS zones) were pooled by site, treatment and year to create a database with 315 observations and 204 vascular plant species.

The NMDS ordination (Figure 2) organized the plant communities according to their historic fire return intervals and site productivity. The fire return interval was strongly associated with the first ordination axis (NMDS1) which accounted for almost 50% of total variation in plant community composition. Dry aspen-grassland complexes on southwest facing slopes with frequent surface fires were situated at far left on NMDS Axis 1, followed by aspen-spruce mixedwood plant communities and lodgepole pine-dominated communities found in landscapes that have experienced regular stand-replacing wildfires (Figure 2a). Mesic and wetter spruce-dominated forests with intermittent stand-replacement wildfires are situated in the centre of the ordination followed by western red cedar and western hemlock-dominated forests with even longer fire cycles. At far right are high elevation wet subalpine fir forests with fire return intervals exceeding 500 years.

The second ordination axis (NMDS2) accounted for 16% of total variation in plant community composition. It was strongly associated with site productivity (e.g., higher site indices, higher soil nutrient regimes), with low productivity-nutrient poor sites dominated by ericaceous species located at the bottom of the graph while rich productive seepage sites with lush vegetation and nutrient rich grasslands were situated closer to the top of graph.

Recently burned plant communities overlapped almost entirely with the range of variability present in unburned communities (Figure 2b), indicating that shifts in species composition related to fire and early post-disturbance succession were relatively minor compared to the more important compositional gradients imposed by geography and climate. The partial redundancy analysis supported this interpretation: four geographic covariables (fire return interval, longitude, soil nutrient regime, soil moisture regime) jointly accounted for 26% of total variation. Although recent burn severity and post-disturbance stand age both significantly influenced plant communities composition (p = 0.001), they accounted for just 2.6% and 1.7% respectively, of the total variation in community composition[10].

In summary, shifts in plant community composition caused by prescribed burning and wildfire in our Central and Northern Interior datasets were not substantial enough to shift the burned communities outside of the range of variation found in unburned communities within the same general community type. Vegetation communities found in the BC Interior have been shaped by their past exposure to wildfire: communities with the greatest exposure to fire (drier boreal and sub-boreal community types) possess the most fire-adapted species and can be expected to be most resilient to fire, whereas communities with a history of infrequent fire (wettest ICH and ESSF forests) are more likely to undergo changes following exposure to unusual fire events. Conversely, the effects of several decades of fire

[10] Notable differences between burned and unburned communities (i.e., black and red triangles located at the very bottom of Figure 2b, outside of the range encompassed by green unburned communities), are due to a bias in the datasets, because examples of the most nutrient poor site series were lacking in our unburned dataset. The NMDS was unable to compensate for such biases, whereas the partial redundancy analysis removes these effects.

exclusion are likely to be most evident in highly fire-adapted plant communities and less evident in communities adapted to long fire-free intervals.

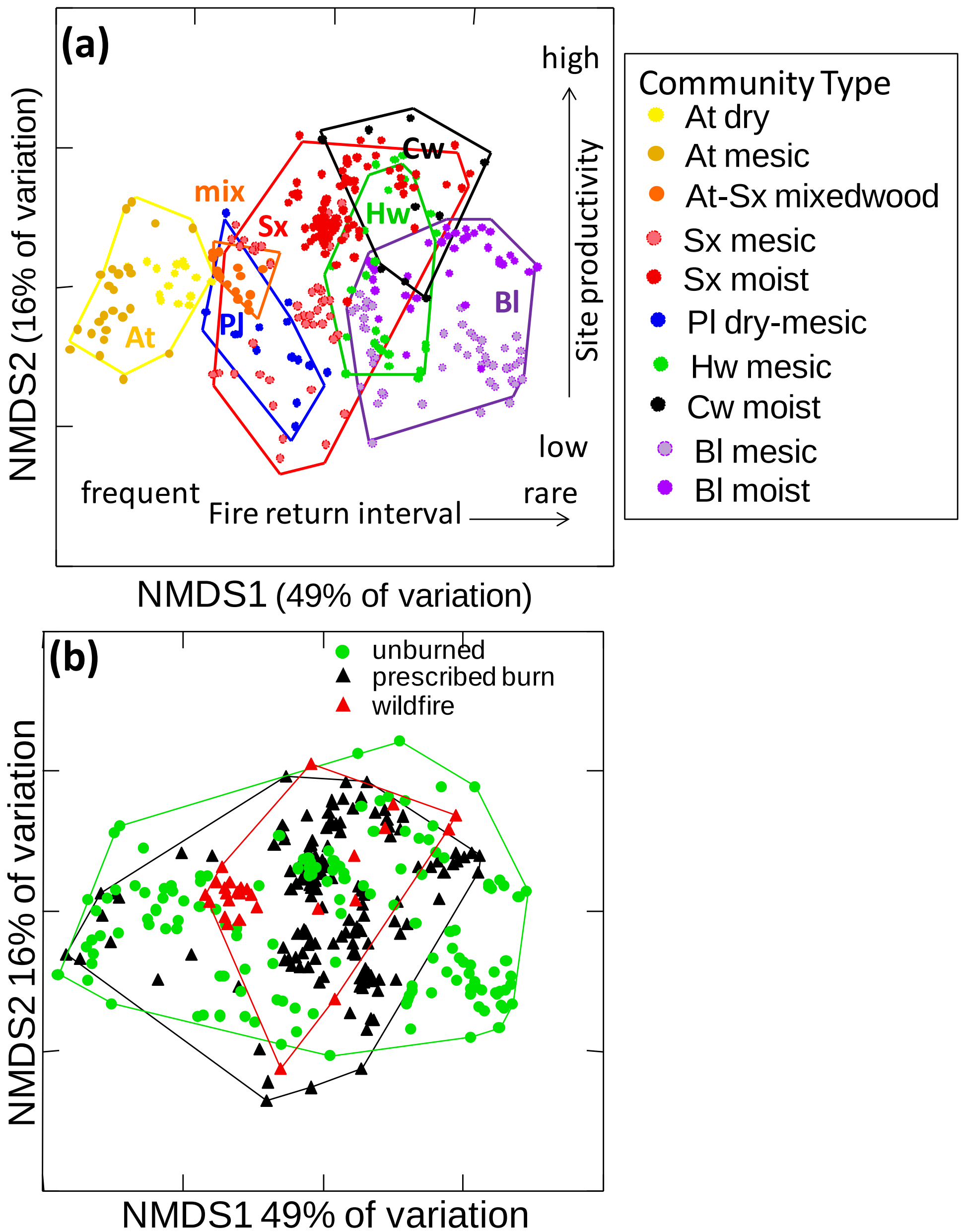

Figure 2. Ordination of northern and wet southern interior plant communities (BWBS, SBS, ESSF and ICH zones) across gradients of fire history (NMDS Axis 1) and site productivity (NMDS Axis 2). Each symbol represents a particular plant community at a point in time. In (a), symbols colours distinguish broad community types with two letter codes indicating the dominant tree species (At = trembling aspen, Sx= interior spruce, Pl = lodgepole pine, Hw = western hemlock, Cw = western redcedar, Bl = subalpine fir). In (b) symbols indicate recent fire history: green circles indicate pre-burn or unburned plant communities, triangles indicate post-burn plant communities (black = prescribed burn; red = wildfire).

Based on the outcomes of the overview ordinations (Figure 2a), we sorted the Central and Northern Interior datasets into three Broad Community/Landscapes Types (BCLTs) defined by their historic fire regimes to better address Burning Questions.

Table 1. The three BCLTs.

Plant Community/Landscape Type (BCLT)	Historic Fire Regime
1) Mesic and Wetter Spruce, Subalpine Fir, Western Hemlock Communities in Mountainous Landscapes	infrequent high severity wildfire
2) Dry Lodgepole Pine Communities in Plateau Landscapes	moderately frequent high and mixed severity wildfire
3) Aspen Mixedwood and Aspen-Grassland Communities in (Sub)Boreal Landscapes	frequent low and mixed severity wildfire and prescribed fire

Each BCLT encompasses a collection of plant associations and site series that cross BEC zone and subzone boundaries. For example, Spruce-Devil's-club plant associations, included with several other plant associations in BCLT 1, are considered mesic and zonal within the very wet SBSvk subzone (SBSvk/01), and are considered hygric in the moist SBSmc subzone (SBSmc/09), but have similar fire regimes, plant communities, and management questions in both subzones and are presumed to respond in broadly similar ways to wildfire and prescribed burning.

3.1.1 Broad Community/Landscape Type (BCLT) 1 - Mesic and Wetter Spruce-Subalpine Fir and Western Hemlock-Western Redcedar Communities in Mountainous Landscapes with Infrequent High Severity Fires

3.1.1.1 Introduction

The BCLT 1 includes a broad range of conifer-dominated plant communities on mesic and wetter sites, at middle and higher elevations in often mountainous terrain with a moist to very wet climate. Late successional understory plant communities dominated by *Vaccinium* spp., *Menziesia ferruginea* and other ericaceous shrubs and feathermosses on well-drained sites; by *Gymnocarpium dryopteris* and/or *Lonicera involucrata* on imperfectly drained sites, and by *Oplopanax horridus* and various ferns on deep, productive seepage sites are included. Mid- and early-successional plant communities with warmer aspects generally have well developed shrub and herb layers that produce a wide range of berries producing plants and other forage for bears and other wildlife. These are important sites for timber harvesting, berry picking and for medicinal plants including devil's-club. Deep snowpacks and lack of deciduous trees make these areas generally less suitable as moose winter range.

Wildfires occur infrequently within BCLT 1 and it has been less affected than BCLTs 2 and 3 by the spate of wildfires since 2000. This may change if the spruce beetle outbreak intensifies and if regional droughts become more severe – as are predicted (Wotton et al. 2010). From the 1960s to 1990s broadcast burning was widely used to reduce fuel loads after logging and to prepare sites for planting. Most of the available datasets we use to examine the response of plant communities to fire comes from slashburning studies established in the 1980s (e.g., Kranabetter and Macadam 2004; Haeussler and Hamilton 2008; Chandler et al. 2017). More recently, fuels left behind after logging have been piled and burned at the roadside. In total, the dataset for BCLT 1 includes 16 fall season slashburn sites, 1 wildfire (portions of the Swiss wildfire study[11]), and 4 study sites with plots that were not recently burned (i.e., unburned controls) (total of 676 plot x year observations). This dataset was used to develop mixed effects models for berry producing species and grizzly bear forage species. Caution should be used in interpreting the results due to the strong bias toward slashburning studies and the lack of a balanced set of unburned controls.

The analysis of general vegetation, conifers, deciduous trees and tall shrubs, ericaceous shrubs and non-ericaceous shrubs was done using Dataset B2 which includes the repeated measures sites from the BCLT 1 Dataset B1. These sites had been clearcut logged and slashburned. Sampling and analysis methods are described in Chandler et al. 2017. The analysis of berry producers, grizzly bear forage and moose browse was done using Dataset B1.

3.1.1.2 General Vegetation[12]

Cover of plants associated with mature forests increased forming > 40% of cover by year 20. There was faster recovery and conifer growth in SBS versus ESSF sites and greater, more persistent deciduous tree and tall shrub cover in SBS versus ESSF. Ericaceous shrubs were prominent in ESSF by year 20. There was little weedy (shade intolerant) species cover by year 20. These SBS, ICH and ESSF ecosystems are fairly resilient to burning in that understory species composition appears to return to pre-burn conditions within a few decades after burning.[13] Moderate to low severity broadcast burning is consistent with maintaining ecological values in these ecosystems.

3.1.1.3 Conifers[14]

In clearcut and slashburned sites in the BCLT 1 monitored for 20 years, conifer cover was similar in year 1 (mean < 0.2%) and increased over time in all zones. It was greatest in the SBS by year 20 (42%) followed by the ICH (35%) and the ESSF (13%), see Figure 3.

[11] The Swiss fire, which was started accidentally, burned approximately 18,000 hectares of forest south and west of Houston in the summer of 1983. http://saythenames.blogspot.com/2011/05/swiss-fire.html

[12] This analysis was done using Dataset B2, which includes only the repeated measures sites from the BCLT 1 dataset B1.

[13] See Chandler et al 2017 for discussion on what constitutes resilience in these ecosystems

[14] This analysis was done using Dataset B2, which includes only the repeated measures sites from the BCLT 1 dataset B1.

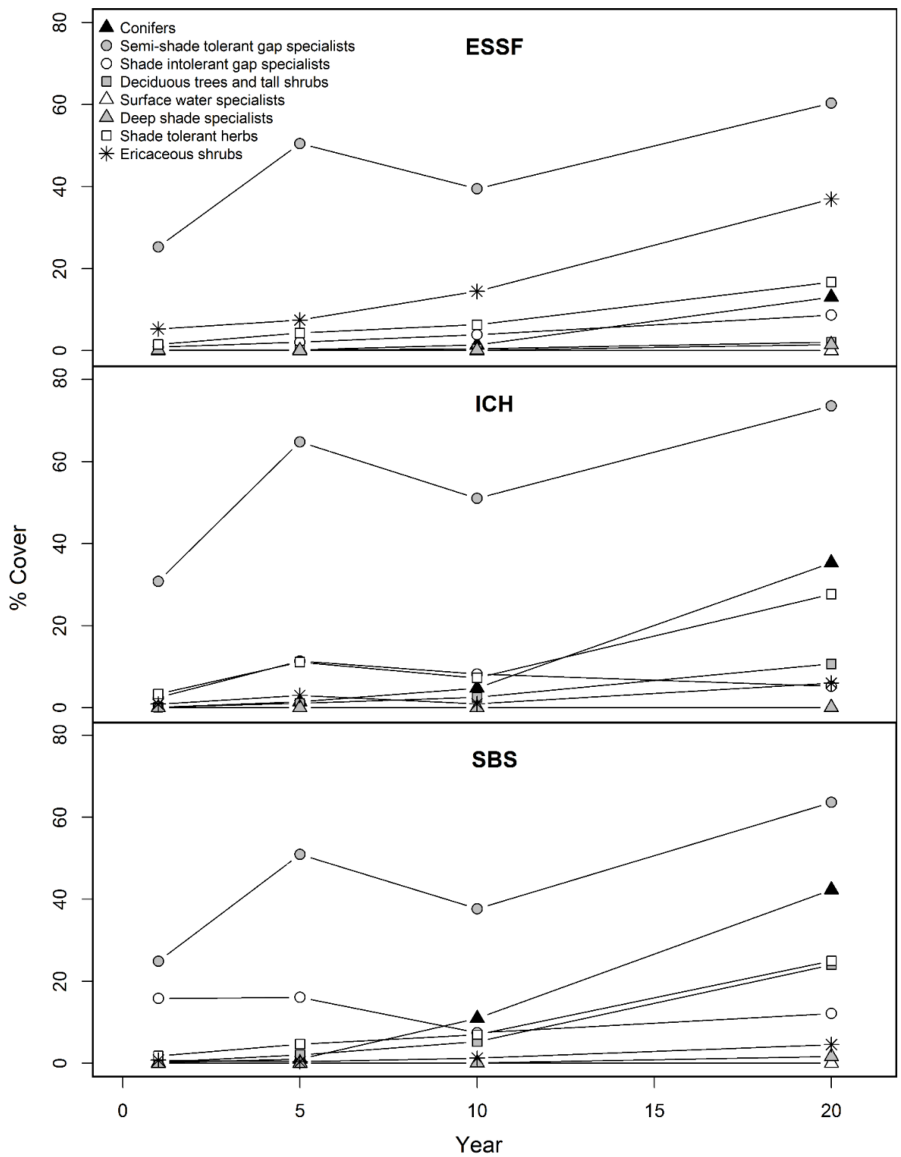

Figure 3. Change in cover of plant functional groups over time in wetter SBS, ICH and ESSF sites. Figure 2 from Chandler et al. 2017.

In the ESSF there was limited conifer cover by year 20 post-treatment. Cover was; *Picea glauca x engelmannii* < 5%, *Pinus contorta* < 5% and *Abies lasiocarpa* < 1%[15]. The mature forests in this zone are dominated by *A. lasiocarpa* and *P. engelmannii*. A certain amount of natural regeneration by these species is expected however the mid-term composition of the young forest will likely be fairly different from that found previously on the sites.

In the ICH zone, mean conifer cover was over 25% and up to 100% by year 20. Planted *P. glauca* x *engelmannii* was the overwhelming dominant, with very minor amounts of naturally regenerated *Thuja*

[15] The conifer cover values are somewhat distorted because on one of the 4 ESSF sites (Otter Creek) trees were not planted in the 3x3 m plots although the surrounding area in the cutblock was planted.

plicata, Tsuga heterophylla and A. lasiocarpa (< 1% cover each). The mature forests in these sites are typically dominated by *T. plicata and T. heterophylla* (10-25% cover each) with lesser amounts of *A. lasiocarpa* and *Pseudotsuga menziesii* var. *glauca* (5-10% cover each) so the tree species composition of the young forests differs from that found prior to logging.

In the SBS zone conifer cover was between 10 and 100% by year 20. *P. contorta* and *P. glauca x engelmannii* were the dominant species with 25-100% and 10-25% cover respectively. *A. lasiocarpa* had < 0.1% cover. *P. contorta* cover was much greater than typical of mature SBS forests (i.e., 10-25%), while *P. glauca x engelmannii* had typical cover (i.e., 10-25%). In the SBS overstory tree species in young forests (i.e., *P. contorta* and *P. glauca x engelmannii)* are more like those typical of mature forest than is the case in the ESSF and ICH. Tree species more typical in mature forests in these other zones (*A. lasiocarpa, T. plicata* and *T. heterophylla*, respectively) may seed in to an extent from adjacent mature forests, if they occur sufficiently close; otherwise the new forests will likely be dominated by *P. glauca x engelmannii*.

3.1.1.4 Deciduous Trees and Tall Shrubs[16]

In clearcut and slashburned sites in the SBS, ICH and ESSF zones monitored for 20 years, cover of 11 deciduous trees and tall shrubs was low until after year 10 when it increased considerably. By year 20, these deciduous species formed a significant part of the plant community in the ICH and SBS with 10.7-24% cover respectively while in the ESSF cover was only 2.1%. Aspen (*Populus tremuloides*) and paper birch (*Betula papyrifera*) were the dominant deciduous tree species with cover of about 3% each by year 20; species of *Salix, Alnus* and *Sorbus* were the main tall shrubs in these sites (Figure 3).

During early succession, these species provide forage for ungulates including moose, elk and deer. Burned SBS sites, in particular, provide significant amounts of these forage species. Broadcast burning is a good option for increasing ungulate forage production in the first 20 years after logging in the SBS and the ICH zones. See Moose browse Swiss fire for an example of the response of deciduous trees and shrubs (moose browse) after a wildfire.

3.1.1.5 Ericaceous Shrubs[17]

Shade tolerant ericaceous shrub[18] cover in clearcut and slashburned SBS, ICH and ESSF zones was monitored for 20 years and increased over time in all zones. This was particularly true in the ESSF where by year 20 ericaceous shrubs had 36.9% cover compared to 6.0% in the ICH and 4.6% in the SBS.

In the burned ESSF sites *Rhododendron albiflorum* had 6.3% cover at year 20. Comparing this with typical cover in mature forest sites of 10-25% suggests that while it survives burning it may take decades to recover. *Menziesia ferruginea* cover increased from 1.7% one year post burn to 12.8% cover by year 20 in the ESSF.

Vaccinium membranaceum, which is typically found in mature ESSF forests with 10-25 % cover, increased in cover steadily from 3.1% one year post burn and had 22% cover by year 20. *V. ovalifolium* increased from 1.2% one year post burn and reached 7.8 % cover by year 20. *V. membranaceum* had 1-

[16] This analysis was done using Dataset B2 which includes only the repeated measures sites from the BCLT 1 dataset B1.
[17] This analysis was done using Dataset B2 which includes only the repeated measures sites from the BCLT 1 dataset B1.
[18] Included *Vaccinium ovalifolium, V. membranaceum, V. myrtilloides, V. ovalifolium, V. parvifolium, V. caespitosum, V. alaskaense, Menziesia ferruginea* and *Rhododendron albiflorum*.

5% and 5-10% cover in mature forests in the SBS and ICH respectively and 1-5% cover in the SBS and < 1% cover in the ICH at year 20. *V. membranaceum* was much more successful in developing in the ESSF than in these other zones. Greater competition from other plants (especially planted conifers) may limit its development in the lower elevation zones.

3.1.1.6 Non-ericaceous Shrubs[19]

Rubus parviflorus cover increased and then decreased over 20 years going from 12.6% one year post burn to 17.6% by year 5 and then to 12.7% by year 20 in the ICH. *Rubus ideaus* (SBS) cover increased from 1.6% one year post burn to 7.5% by year 5 and then down to 0.2 by year 20. Many buried *Sambucus racemosa* seeds germinated immediately post-burn in the ICH. Cover was 14.1% at year one, then declined to 0.5% by year 20 as only a small percent of germinants survived.

3.1.1.7 Berry Producers[20]

<u>Berry Producers on All Sites</u>

The Swiss fire dataset (B3) was analysed to determine percent cover of all shrub and herb species that produce fleshy berries (and berry-like fruits) consumed by wildlife. Cover was summed to produce the variable "Total Berry Producers". A box plot of total berry producers showed a general increase in percent cover with time since disturbance from 1-22 years (Figure 4). Cover was higher at 0 year (pre-burn) and there was increasingly variable cover in the few, poorer quality datasets that included stands over 22 years of age. Minor undulations in the curve (e.g., dip at year 8, peak at year 17) were interpreted by us as artefacts of the varying length of datasets from sites differing in productivity.

A mixed effects model was developed for total berry producers for young stands only (0-22 years) (Figure 5). The best fitting model was a complex 3rd order polynomial of stand age with two additional fixed effects: (1) an indicator variable for burned sites (red lines) versus unburned sites (green lines); and (2) an indicator variable distinguishing the *Vaccinium*-moss plant association (solid line) from moister Oakfern, Twinberry and Devil's-club plant associations (dashed line). The model did not differentiate between abundance of berry producers on slashburned clearcuts and the Swiss wildfire or between burns of varying severity, likely because most of the 16 slashburns were of low to moderate severity. *Vaccinium*-moss plant communities had higher cover initially when unburned (green solid line) than after a fire (red solid line) but the cover of berry producers in unburned *Vaccinium*-moss sites began to decline 10 years after clearcutting whereas cover in burned sites increased for at least 20 years.

Berry patches in Devil's club and Fern plant communities (dashed lines) had more complex response curves because they support a variety of early seral berry producers (Rosaceae spp., *Ribes* spp., *Sambucus racemosa*) that responded relatively quickly to clearcutting (green dashed line) or fire (red dashed line), but were subsequently replaced by late seral species (*Oplopanax horridus, Lonicera involucrata, Cornus sericea, Vaccinium* spp.) that are able to persist in the gappy spruce and subalpine fir stands that characterize moist seepage sites. In this model, the cover of berry producers in unburned Devil's club-Fern sites began to decline after 17 years, whereas cover in the burned communities peaked at 7 years post-fire and then surged again in the second decade.

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[19] This analysis was done using Dataset B2 which includes only the repeated measures sites from the BCLT 1 dataset B1.
[20] This analysis was done using Dataset B3 which is repeated measures data from the Swiss fire.

Figure 6 illustrates the different trajectories of early-mid versus late seral berry producing species over the first 0-22 years after disturbance. These boxplots do not distinguish between mesic and moist plant associations, nor between burned and unburned sites. Early seral species such as raspberry, red elderberry, wild strawberry and stink currant follow the expected arched trajectory over the first 8-10 years (after which the data were unreliable), while the late seral species follow the expected sigmoidal recovery trajectory for the full 22 years after disturbance.

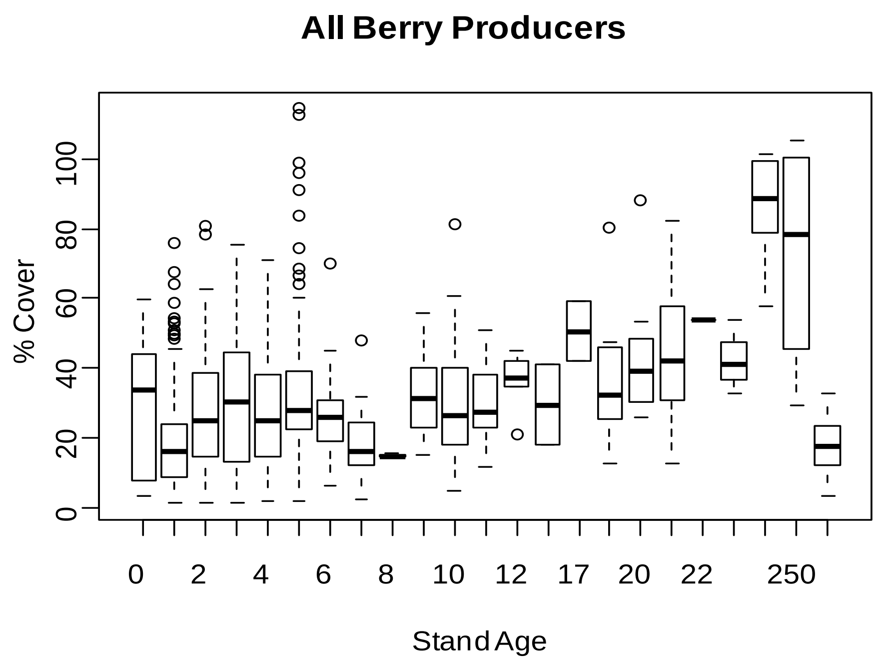

Figure 4. On mesic and wetter Central and Northern Interior sites, the total abundance (percent cover) of all berry producing shrubs and herbs increased with time since disturbance (stand age), regardless of whether sites were burned or clearcut. Note that the stand age axis is not linear after 12 years and age 0 is pre-burn. Bold solid lines = median; box = upper/lower quartile; whiskers = maximum/minimum data points - excluding outliers (points).

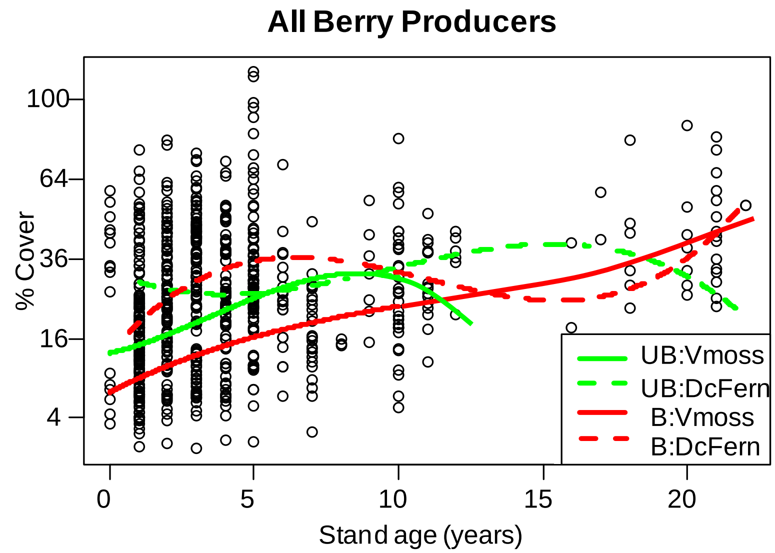

Figure 5. Best fitting model for changes in the abundance (% cover) of all berry producing species for 22 years following disturbance on unburned (UB, green line) versus burned (B, red line) sites, and on *Vaccinium*-moss (solid line) versus Devil's-club and Oakfern (dashed line) plant associations. Note stand age 0 is pre-burn and the vertical axis is non-linear.

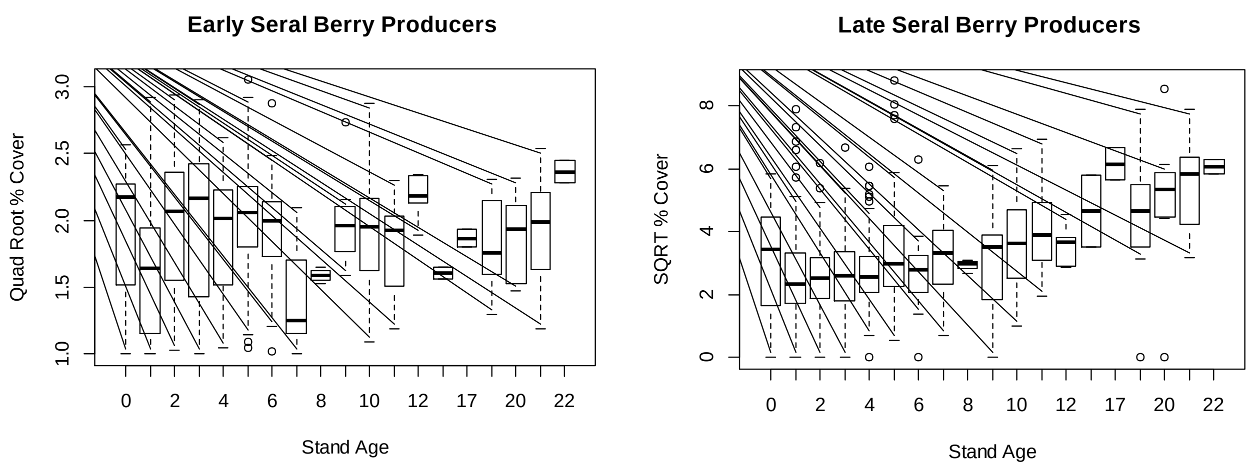

Figure 6. Percent cover trajectories for early (to mid) seral berry producing species (left) versus late seral, shade tolerant berry producing species (right) for 22 years after disturbance. Note that the Y axes, and X axes beyond 12 years, are non-linear.

Berry Producers on the Swiss Wildfire

Plant community development was monitored for 20 years in an ecological reserve located within the Swiss wildfire (burned from May 29 to June 3, 1983), located in the Morice River Ecological Reserve, near Houston, BC. This site is unique because it received no management interventions and allowed us to study the development of naturally regenerated plant communities. Two models were developed for berry producers on the Swiss wildfire: (a) total berry producers, as above (Figure 7a), and (b) berry producers preferred by humans (Figure 7b), which includes just popular edibles such as huckleberry, strawberry, raspberry, saskatoon and highbush cranberry. Both models distinguished between the trajectories on a riparian swamp forest and adjacent upland sites, but the model did not distinguish between mesic and moist upland sites. On the upland sites, the model also distinguished trajectories between a single site that experienced a low severity fire and 5 other sites that experienced a high severity burn.

For total berry producers, low severity wildfire has a similar effect as unburned sites in the analysis described above: cover is high initially, but peaks before ten years, then begins to decline; whereas high severity burns and burns in swamps, have lower cover initially but increase over the entire 2-decade time period. To summarize – with higher burn severities, recovery is slower but more sustained. Riparian swamp forest sites had similar berry cover initially to upland sites, but a faster rate of increase, reaching almost 100% cover 20 years post-disturbance.

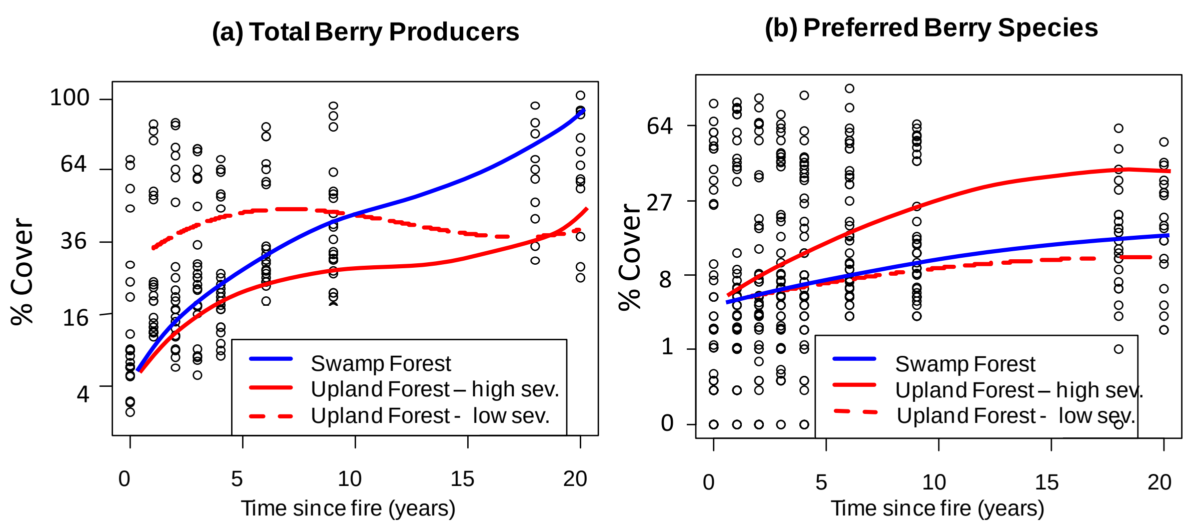

Figure 7. Percent cover trajectories for (a) total berry producing species and (b) berry producers preferred for human consumption on the Swiss wildfire (an unmanaged ecological reserve) near Houston, BC. Both models indicated that trajectories differed between upland mesic to moist conifer forests (red) and riparian swamp forests (blue), and between low severity (dashed line) and high severity (solid line) burns on the upland sites. Note that the Y axes are non-linear.

Plants producing berries preferred by humans made up approximately half of the total cover of berry producers in the Swiss wildfire. Their trajectories differed from total berry cover in at least three ways: (1) all of the modeled trajectories stabilized after 12-15 years rather than decreasing or continuing to increase; (2) on upland sites, the high severity burn had greater cover of berry producers than the low severity burn over the first decade; (3) cover of preferred berry producers was lower in the riparian swamp than on the upland, undoubtedly because the dominant berry producers in the riparian swamp were red-osier dogwood and twinberry – valuable for wildlife, but unpalatable to humans. Thimbleberry and snowberry were important species on the upland sites that quickly resprouted after the low severity burn, whereas preferred edible species such as black huckleberry and highbush cranberry were slower to recover but continued to increase as in the late seral trajectory of Figure 6.

Although an increase in percent cover of berry producing species is generally positively correlated with fruit production, it should not be directly interpreted as an increase in the quantity or quality of edible berries. In agricultural settings, berry production and quality are highest in the first 1 to 5 years after a disturbance or after establishment, and plants are quickly replaced or re-disturbed to maintain berry production.

3.1.1.8 Grizzly Bear Forage Plants[21]

Prime grizzly bear habitat is found in BCLT 1. We wanted to know how fire influenced the availability of grizzly bear forage plants, and whether there were trade-offs between managing for early season vs. late season forage abundance. We also wanted to know how site quality influenced the response to burning, and whether reforestation choices such as planting versus natural regeneration and planting lodgepole pine versus interior spruce reduced the amount of time that forage was abundant after burning.

We obtained lists of grizzly bear forage species from two study areas, the Babine River watershed (MacHutchon 2015) and the Parsnip River (Beaudry et al. 2001). We divided the grizzly bear forage into two seasons. The Early season encompasses spring to early summer when the bears feed mainly on herbaceous plants including willow catkins, grasses, sedges, horsetails, ferns and wide variety of forbs. The Late season encompasses late summer to fall when the bears feed mainly on berries plus some roots and seeds. We weighted each species according to whether it was ranked as being of major importance (weight = 1), medium importance (weight = 0.5) and minor importance (weight = 0.25) in each season and separately summed up the weighted Early and Late season percent covers.

Early and late season mixed effects models were developed by selecting (as above) from a variety of indicator variables for the following fixed effects: (1) stand age; (2) burn severity; (3) site quality; (4) reforestation choices. We found that the indicator variables used for reforestation choices (i.e., planted or not; species planted) did not produce reliable results due to strong biases in the datasets that could not be overcome by the random effects in the model.

Early Season Grizzly Bear Forage

The best-fitting model for early season grizzly bear forage (square-root transformed) was a fourth order polynomial of stand age (log-transformed), with burn severity (0 = unburned to 4 = high severity burn) as an interaction term, and the actual soil moisture regime (adapted from Delong *et al*. 2011; ASMR: 2 = fresh, 3- 4 = moist, 5 = very moist, 6 = wet) as both an intercept and interaction term (Figure 8). The model indicated that site quality had a much larger effect on early season forage abundance forage than burn severity. Very moist (seepage) sites produced substantially more forage than fresh sites except in the first year after a severe burn. This is unsurprising since horsetails and ferns are important parts of the bears' spring diet. Unburned sites had more forage in the first year after disturbance than burned sites, but thereafter differences were relatively minor. Forage increased more rapidly and peaked slightly earlier after a severe burn than after lesser burns. Forage production was sustained for longest on unburned sites – perhaps due to a transition from herbaceous to woody species. The second surge in forage abundance at 20+ years was probably an artefact of our unbalanced datasets which included relatively few sites monitored for that length of time.

Late Season Grizzly Bear Forage

[21] This analysis was done using Dataset B3, which is repeated measures data from the Swiss fire.

The best-fitting model for late season grizzly bear forage was a relatively simple linear function of stand age with separate intercepts, separate slope terms and an interaction term for burn severity and the actual soil moisture regime (ASMR) (Figure 9). Late season forage, just like early season forage, increased substantially with soil moisture availability, but for late season forage, burn severity had a major impact on successional trajectories over the first 20 years. Unburned sites had considerably more forage initially than burned sites, especially on fresh sites, where huckleberries and blueberries are the most important grizzly foods, and the burned sites did not produce a similar amount of forage until approximately 10 years after disturbance. Moderate burns produced more late season forage initially than severe burns, but severe burns ultimately produced more forage, especially on moist sites. At 20 years, grizzly forage was declining on unburned sites, but continued to increase on burned sites.

Trade-offs Regarding Forage Production

Both models agreed that prescribed burning reduces grizzly forage production for the first few years after clearcutting, but the effect lasted longer for late-season forage than for early season forage. Seven to 15 years after disturbance, the models predicted that early season forage would continue to be more abundant on unburned sites, but that late season forage would be considerably more abundant on burned sites, particularly after a severe burn on moist to wet seepage sites. These trade-offs suggest that a single management regime is not appropriate for optimum grizzly bear forage management – some sites should probably be burned for long-term berry production, whereas others should be left unburned to increase the abundance of ferns and other herbaceous forage species and for short term food supplies. Likewise, it makes sense to stagger harvesting so that cutblocks are available at various successional stages to maximize forage diversity. We don't have data on the nutritional quality of the forage nor berry abundance, both of which may be better on burned than unburned sites (at least for some species) even if the plants themselves have less foliage. Reforestation choices may greatly affect forage production, quality (and hiding cover) for grizzly bears in the second and subsequent decades after harvest, but more data from a variety of sites are needed to include these variables in future models.

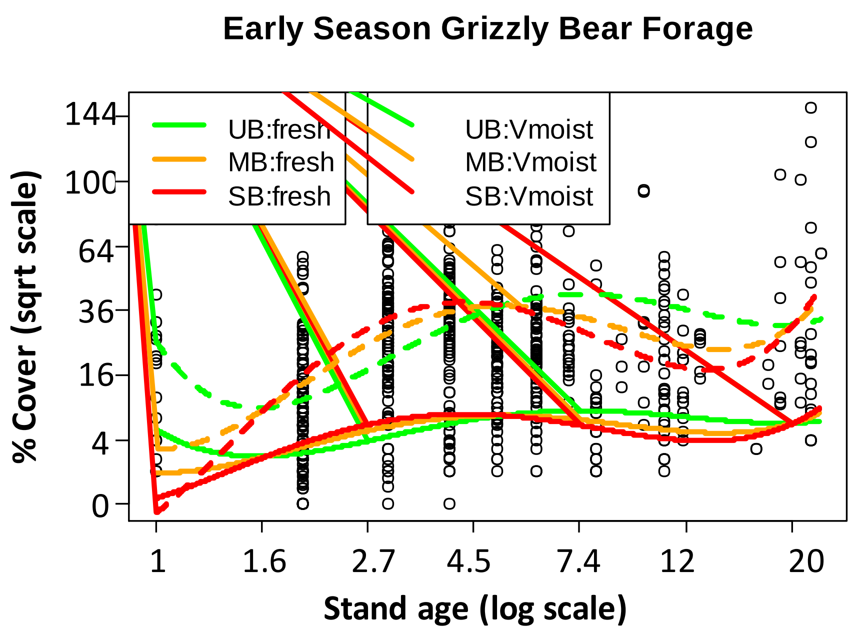

Figure 8. Best fitting model for changes in the abundance (% cover) of early season grizzly bear forage for 22 years following disturbance on unburned (UB, green line), moderate (MB, yellow line) and high (SB, red line) severity burns, and on fresh (solid line) and very moist (dashed line) soil moisture regimes. Note that both the horizontal and vertical axes are non-linear.

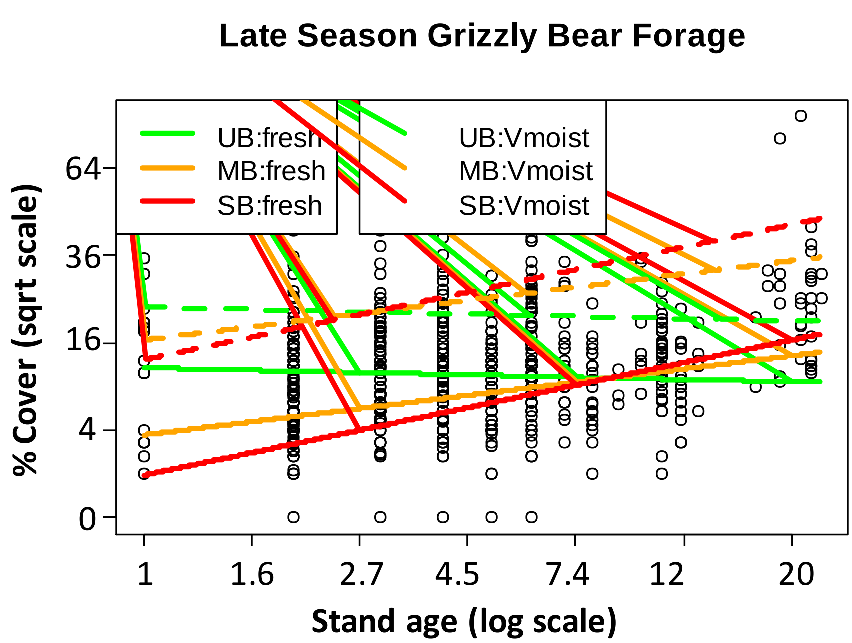

Figure 9. Best fitting model for changes in the abundance (% cover) of late season grizzly bear forage for 22 years following disturbance on unburned (UB, green line), moderate (MB, yellow line) and high (SB, red line) severity burns, and on fresh (solid line) and very moist (dashed line) soil moisture regimes. Note that both the horizontal and vertical axes are non-linear.

Trade-offs involved in managing for grizzly bear forage with prescribed burning were examined in site specific detail using the Kinskuch prescribed burn study, located near northwest of Hazelton, BC (Figure 10). Paired cutblocks, one slashburned (moderately-low severity) and the other unburned, were monitored for 10 years (Kranabetter and Macadam 1998). These were both mesic *Vaccinium*-Moss plant communities in the ICHmc subzone, and the sites that were planted to lodgepole pine. The unburned site had much less early season grizzly bear forage (green circles) over the first 10 years than the burned site (orange circles), but it had three times as much late season forage (dark green triangles) than the burned site (yellow triangles).

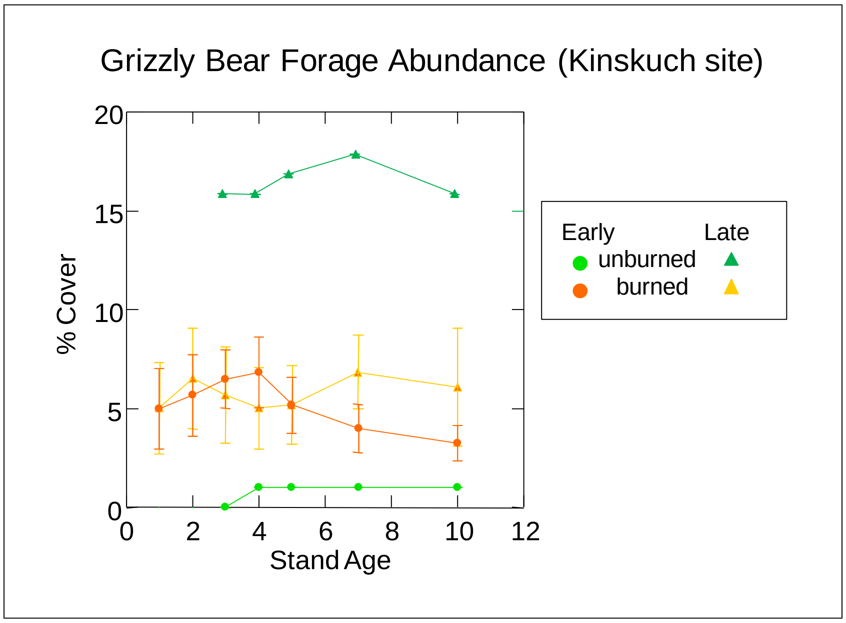

Figure 10. Comparison of early season (circular symbols) and late season (triangular symbols) Grizzly bear forage species abundance for the first 10 years following clearcutting on unburned (green symbols) and burned (yellow & orange symbols) plots at the Kinskuch prescribed fire trial. Error bars indicate within year, among-plot standard errors. There was just one unburned control plot and it was not monitored in years 1 and 2.

The Kinskuch results do not entirely agree with the grizzly bear forage analysis conducted on dataset B3 for fresh sites which indicate that early season forage abundance is similar on burned and unburned sites from 3 to 10 years post-disturbance (Figure 8, orange and red solid lines). But they do agree that late season forage will be substantially more abundant on unburned sites over this time period (Figure 9, solid lines). On burned sites, the Kinskuch data also show the early season forage declining after 4 years while the late season forage is stable or increasing, which corresponds well with the predictions of the results of the multi-site models. Ten years after planting, the lodgepole pine trees on the Kinskuch site were probably having significant impact understory vegetation. Caution is needed in extrapolating the Kinskuch study results because there was just one burned cutblock and one unburned cutblock, thus differences between burned and unburned could have been due to site differences rather than treatment differences. Responses on other ecosystems may be different.

3.1.1.9 Moose browse

On the Swiss wildfire[22], 50 vegetation sampling plots were established immediately post-burn and re-measured after 1, 2, 3, 4, 6 and 9 years. A subset of 25 of the 50 plots was remeasured 18 and 20 years post-burn. Fire severity was estimated for each plot and % cover of all plant species was recorded. Cover of all moose browse species (deciduous trees and select shrubs) was pooled and modeled as a function of time since fire.

The best fitting nlme model for moose browse production after the Swiss wildfire distinguished between a dense submesic Pine stand (BCLT 2) and all other site series (BCLT 1) but did not differentiate the low burn severity site from high burn severity sites. In these other site series moose browse species increased rapidly in the first 5 years after burning and continued to increase over the 20-year period (Figure 10). This period of increasing browse cover is longer than previously been recorded on unburned and prescribed burned clearcuts on comparable ecosystems where the abundance of these important wildlife forage/browse species typically peaks 10 to 15 years after disturbance.

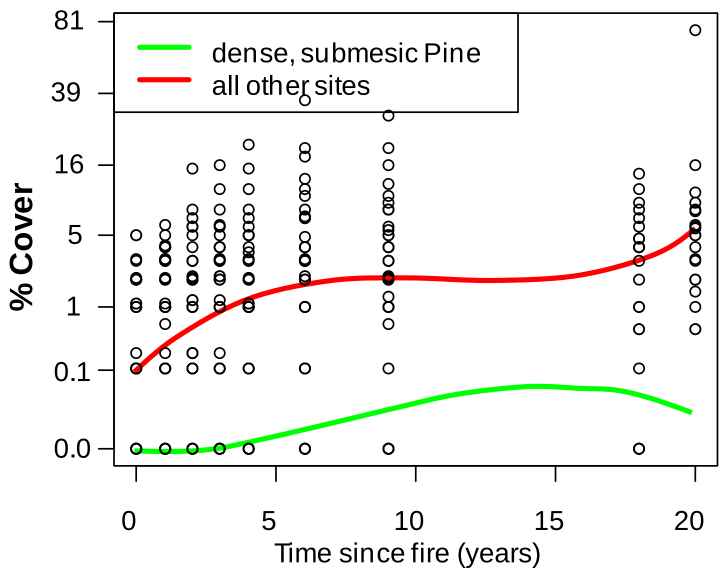

Figure 11. Response of moose browse (deciduous trees and select shrubs less than 2 m tall) over time in the Swiss wildfire.

3.1.2 Dry Lodgepole Pine Communities in Plateau Landscapes (BCLT 2)[23]

These (sub)mesic to xeric lodgepole pine community types in plateau landscapes distributed across the Central Interior have been profoundly impacted by the cumulative effects of mountain pine beetle outbreak, salvage logging, and recent major wildfires, many of which have overlapped. There are justifiable concerns about the resilience of these disturbed ecosystems. Early results have generally found relatively high resilience and small shifts in plant communities and lodgepole pine regeneration

[22] The Swiss fire occurs in landscape that would generally be classified as BCLT1. However, the submesic pine sites within the Swiss fire are more typical of BCLT 2.

[23] Based on Dataset C (see Appendix 1).

after MPB and a single fire (e.g., Edwards et al. 2014; Ton and Krawchuk 2016) and early data from MPB-damaged forests exposed to multiple fires are just beginning to emerge (e.g., Cichowski and Haeussler 2018).

The percentage of exposed mineral soil and rock increased with burn severity (Figure 12) and was approximately twice as high on sites exposed to two successive wildfires than on sites burned just once. Mineral soil and rock exposure were 2-3 times higher on subxeric sites than on submesic to mesic sites across all burn classes, rising as high as 55% on a site that burned twice with high severity compared to 17% on a submesic site exposed to two high severity fires. While exposed soil and rock create high risks for soil erosion and non-native species invasion – notably hawkweeds (Haeussler et al. 2017; Kranabetter et al. 2017) - it also creates opportunities for caribou forage lichen establishment.

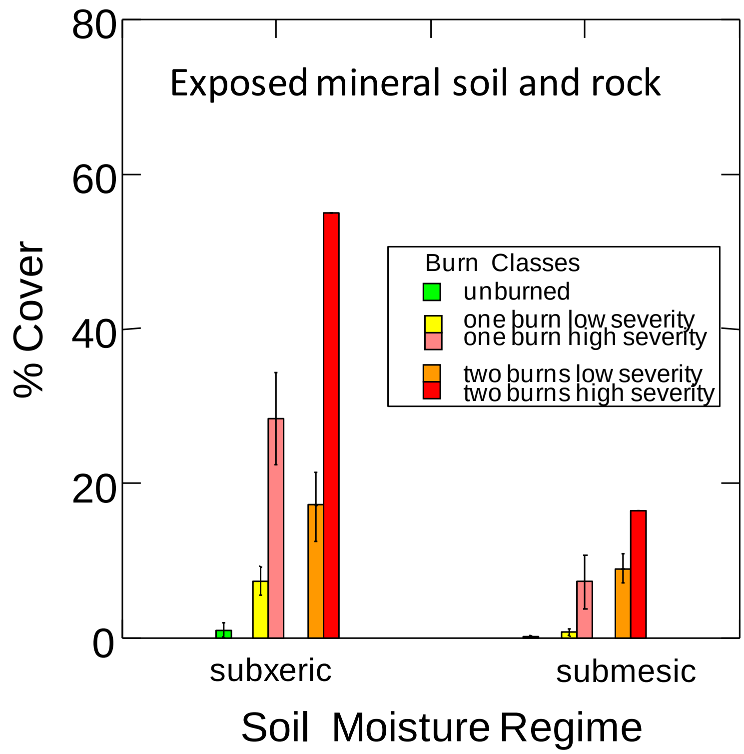

Figure 12. Effect of burn frequency and severity on the percentage of exposed mineral soil and rock on subxeric and submesic sites in BCLT 2. Error bars represent standard errors within site series and burn classes.

The regrowth of preferred terrestrial caribou forage lichens (*Cladina* subgenus; *Stereocaulon* spp.) was, however, extremely slow: ≤ 0.2% cover 10-12 years after a single wildfire, compared to 20-40% in unburned forests and clearcuts), and none of the sites that burned twice had visible signs of lichen regrowth 2-3 years following the fire. Artificial "reseeding" of lichens (Rapai et al. 2016) is now being tested at several locations across the central Interior to speed the recovery process, and the data we have compiled supports this intervention.

Lodgepole pine ecosystems are, in many respects, highly adapted wildfire, but multiple burns occurring within a decade in a landscape recovering from the effects of a massive mountain pine beetle outbreak and unprecedented salvage logging have the potential to undermine that resilience and to cause losses in valued resources such as caribou forage lichens, huckleberries and blueberries that generally rebound well (if slowly) after a single wildfire. A high priority for further work will be to analyze BCLT2 tree regeneration datasets compiled for this study, and to follow up on, or expand sampling in, research sites now exposed to multiple successive disturbances.

3.1.3 Aspen Mixedwood and Aspen-Grassland Communities in Boreal and Sub-Boreal Landscapes (BCLT 3)

There is a growing consensus that aspen forest health, moose browse, and the condition of grasslands associated with south-facing slopes in the aspen-mixedwood landscape of the BWBS and SBS zones are deteriorating and that invasive non-native herb species are increasing. We compiled BCLT 3 data from 15 sites, comprising 863 records. Raw data and data tables were acquired from three additional BCLT 3 sites in the Central Interior (Cunningham Lake; Euchiniko Sidehills; Grizzly Valley) but could not be added to the BQ database with the time and resources available. In addition, many of the Northeast BC datasets acquired from the Peace-Liard Prescribed Fire Study could be added to the BCLT 3 landscape dataset. Aside from a single aspen-mixedwood stand in the Swiss wildfire (Figure 13), no quantitative analyses of BCLT 3 data were completed for this study. The summary below is based on earlier analyses of portions of the dataset (Haeussler 2007; Haeussler and de Groot 2008; de Groot and Haeussler 2008, 2016; Helkenberg and Haeussler 2009) and a review of project reports describing prescribed burn studies inherited or initiated by SERNbc (Simonar and Migabo 2006; Sulyma 2008; Albertson 2009; Wilson et al. 2010; Pokorski 2017).

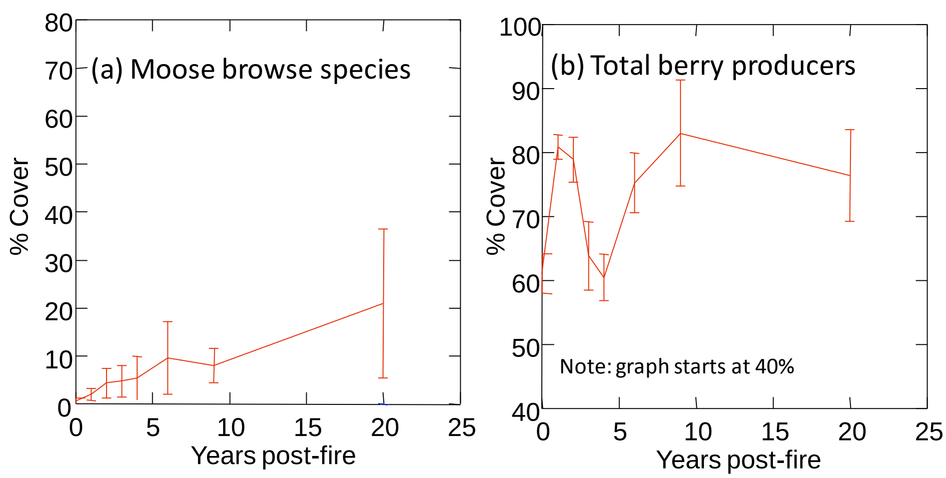

Figure 13. Changes in the percent cover of (a) moose browse species; and (b) total berry producing species on a submesic SBSdk Aspen – Saskatoon seral association over 20 years after the Swiss wildfire, on the Morice River Ecological Reserve near Houston, BC. Error bars represent within-year standard errors (*n* = 5 plots).

For moose browse, only foliage present below 2 m height was tallied; trembling aspen was the dominant browse species, with minor Scouler's willow and osier dogwood. The dominant berry producers were thimbleberry, prickly rose, saskatoon and snowberry. Dandelion was the only non-native species recorded (0.04% cover at 20 years post-fire only) at this protected site.

In general, monitoring data from prescribed burns and wildfires in the BCLT 3 landscape are inconsistent, short term and lack the level of detail and the untreated controls needed to confirm whether forest and rangeland conditions are actually deteriorating and whether greater use of fire will improve or worsen current conditions. Answers seem to vary greatly with local ecological conditions and site history. As in southeastern BC grazing and browsing by wildlife and domestic livestock often confound our ability to distinguish the effects of fire from the effects of herbivory. A relatively recent shift from wetter to drier summers across central BC (BC Ministry of Environment 2016) may complicate our ability to use older datasets and aerial imagery (i.e., prior to ~2000) to assess current trends.

A concerted effort to develop and implement appropriate monitoring is needed. The work of the Peace-Liard Prescribed Fire Study team is important in this regard but needs to be extended to the Omineca and Skeena Regions and should expand upon the work undertaken by SERNbc who do not currently have sufficient resources for comprehensive monitoring. A network of prescribed fire monitoring plots established in grassland-aspen woodland ecotones in the Skeena Region in 2001 (Veenstrat and McLennan 2002; Helkenberg and Haeussler 2009) should be revisited.

3.1.4 Species of special concern

3.1.4.1 *Oplopanax horridus* (devil's club)

Devil's club is a culturally important medicinal plant for some BC First Nations: roots and stems are used to make medicines (Johnson 2005). Impacts of changes in forest condition due to logging, burning and wildfires on this species are therefore of concern. This analysis is based on data from Database A, B2 and B5.

In Database A, devil's club was only found in ICH and SBS sites. Of the 59 ICH and SBS sites in Database A, (which included a total of 5878survey records from 1 to 250 years post-disturbance) there were 518 observations of devil's club across 25 sites. There was an additional 129 observations of devil's club pre-disturbance (year 0; 27 sites). Most of these observations come from repeated sampling of the same plot over time.

Database B2, which included repeated measures studies, showed devil's club was recorded at some point after the site was clearcut and slashburned in just over half (7 of 12) of the ICH and SBS sites[24]. Preburn measurements were also taken at 3 sites: Walker Creek (*n* = 120); Mackenzie (*n* = 6); and Chuchinka (*n* = 21) (Figure 14). Presence and cover decreased immediately after treatment on all three sites. At Walker Creek, mean presence and cover preburn was 88% and 11% respectively while in post-burn years it was 32-47% and 2-4%. At Mackenzie mean presence and cover preburn was 100% and 11% respectively while in post-burn years it was 83-100% and 0.75-3%. At Chuchinka preburn presence and cover was 67% and 8% respectively and in post-burn years it was 19%-5% and 0.05-1%. Devil's club presence decreased over time at Chuchinka, and had not achieved pre-burn cover levels by year 10 at Chuchinka or year 20 at Walker Creek and Mackenzie.

In the other 4 of the 7 sites from Database B2, presence was limited with only 3 observations at Brinks Mill (in the same plot years 3, 5 and 10 but not year 21); 4 observations at Francis Lake (in the same plot years 1, 5, 10 and 21); 12 observations at Haggen Creek (none more than 5 years after slashburn); and 4 observations at Kinskuch (in the same plot years 1, 3, 4 and 5 but not year 10). Cover was very low in all

[24]

of these sites post-burn. Vegetation was not sampled pre-burn at these sites, therefore, pre- and post-burn presence or cover can not be compared.

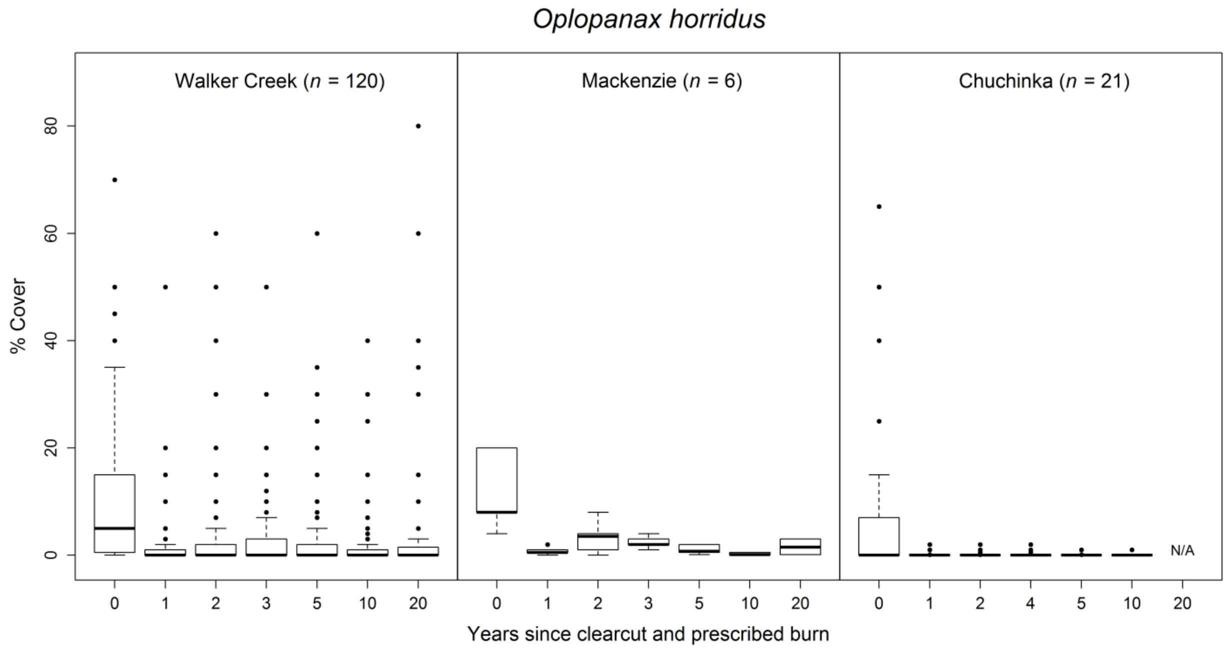

Figure 14. Change in percent cover over time at the three repeated measures sites for which there is pre-burn (year 0) and repeated post burn measurements (years 1-20). Sampling at Chuchinka was conducted until year 10. Bold solid lines = median; box = upper/lower quartile; whiskers = maximum/minimum data points - excluding outliers (points).

We analysed data from a 1988 chronosequence study in the SBSwk1 subzone (Hamilton and Yearsley 1988) that compared cover and presence of plants in mature forests with sites that had been clearcut then slashburned or mechanical site prepared (MSP) (Table 2; Database B5). On submesic (06) sites devil's club had very low cover and was uncommon pre-burn (0.1% cover/<20% presence). It was virtually nonexistent in submesic sites that ranged in time since site preparation treatment up to 24 years. Average cover was 0.3% and presence was 9% on burned sites; it was not found in any MSP sites which ranged in age since treatment up to 7 years. In mesic (01) sites, devil's club had low cover (0.9%) and was moderately common preburn with 21-50% presence, but was not found post treatment in the 16 year period surveyed. In mature subhygric to mesic (07) sites devil's club cover was good (30%) and it was very common – almost ubiquitous (81-100% presence). It had very low cover (<2%) over the 7 year post treatment sample period although presence was good (75% and 43% in burned and MSP sites, respectively). In mature hygric to subhygric (08) sites devil's club cover was good (21%) and it was almost ubiquitous (81-100% presence). It was not observed post treatment over the period sampled in burned (4 years) or MSP (7 years) 08 sites. In conclusion – in the SBSwk1 devil's club disappeared, or virtually did from most logged sites. It did best on subhygric to mesic (07) sites, where it is typically most abundant pre logging; these sites are rich and moist and burn impacts were likely less than on drier sites. No real difference between slashburned and MSP plots or change in presence or cover over time was noted but comparable data is limited.

Table 2. Cover and presence of devil's club in in the SBSwk1 subzone variant in mature forests and early seral sites that were clearcut and either slashburned or mechanically site prepared (MSP) [25]. The data for the mature forests was derived from DeLong et al. 1986. For the early seral plots - the % cover column shows mean % cover, range of cover values, years sampled and number of plots; the % presence column shows % of plots with devil's club, number of plots with devil's club/total plots, years sampled and number of plots.

Site series/Site moisture regime	**Mature forest**		**Early seral, burned**		**Early seral, MSP**	
	% cover	% presence	% cover	% presence	% cover	% presence
(06) Queen's cup *Submesic*	0.1%	<20%	<.03% 0-0.5% 2-24yrs n=22	9% 2/22 2-24yrs n=22	0% 4-7 yrs n=3	0% 0/3 4-7 yrs n=3
(01) Oakfern *Mesic*	0.9%	21-40%	0% 1-16yrs n=19	0% 0/19 1-16yrs n=22	0% 3-8yrs n=9	0% 0/9 3-8yrs n=9
(07) Devil's club *Mesic to Subhygric*	30%	81-100%	1.5% 0 to 5% 1-6yrs n=8	75% 6/8 1-6yrs n=8	2% 0-5% 1-7yrs n=4	43% 3/7 1-7yrs n=4
(08) Horsetail *Subhygric to Hygric*	21%	81-100%	0% 1-4yrs n=6	0% 0/6 1-4yrs n=6	0% 2-12 yrs n=6	0% 0/6 2-12 yrs n=6

In conclusion, we found *O. horridus* was not well adapted to open environments left after logging and burning. It is a shade tolerant plant common in old-growth forests and was restricted to our ICH and SBS sites. It's cover and abundance in the first decades after clearcut logging was low to nonexistent - whether the sites were slashburned or mechanically site prepared. Devil's club is susceptible to fire although thought to resprout from the root crown and/or rhizomes.[26] It may re-establish after wildfires from animal-dispersed seeds after the canopy has closed enough to shade this light-sensitive species. Management to maintain or enhance devil's club should include retention of mature forests where devil's club is most abundant. Severe fire or site preparation treatments should be avoided.

3.1.4.2 Shepherdia canadensis (soapberry) [27]

Soapberry is an important food plant for some First Nations and some wildlife and therefore it is important to determine how management practices affect it.

We determined which of the 75 sites in the Central and Northern Interior database featured *Shepherdia canadensis*. The subset of plots where it had been less than 30 years since the last major disturbance (i.e., cut by logging, shearing or mechanical clearing, or burned by wildfire or prescribed burning) was

[25] Data is from Hamilton, E. and K. Yearsley 1988 (Database B5).
[26] USDA Fire Effects Information System. Available at https://www.fs.fed.us/database/feis
[27] This analysis was done with Dataset A.

identified[28]. Cover and percent presence of soapberry at each of these early seral sites were determined (Table 3).

Table 3. Number and percentage of early seral sites in each plant community type with *Shepherdia canadensis*.

Plant community type	**Early seral sites with *Shepherdia canadensis*** (number and percentage)	
	#sites with soapberry/total #sites of each plant community type	% sites with soapberry
At dry	1/1	100
At mesic	3/5	60
At-Sx	1/6	17
Pl	2/3	67
Sx-mesic	2/5	40
Sx-moist	0/7	0
Bl mesic	0/3	0
Bl-moist	0/3	0
Cw	0/31	0
Hm	0/11	0

S. canadensis occurred in drier early seral plant communities (i.e., At-dry, At-mesic, At-Sx, Pl, and Sx-mesic) in the SBS (dw3, dk, mc2) and the BWBS (mw and mk) subzones. It was not found in the moister early seral plant communities (i.e., Sx-moist, Bl mesic, Bl-moist, Cw, and Hm) (Table 3). In sites where soapberry occurred, it generally had very low cover; it was found in 0.8-58% of the plots on these sites. Cover appeared to increase somewhat over time after the sites were disturbed by cutting or burning. No consistent difference between cover and presence on burned versus unburned sites was apparent.

We plotted the cover and presence over time in each site with repeated measures data (Database B2) and cover in burned (spring burn, fall burn , wildfire) and unburned sites (Figure 15).

[28] Burned treatments include wildfires and broadcast burns (spring and fall). Unburned treatments include clearcut, shear and manual brushing. There were 702 burned and 235 unburned samples.

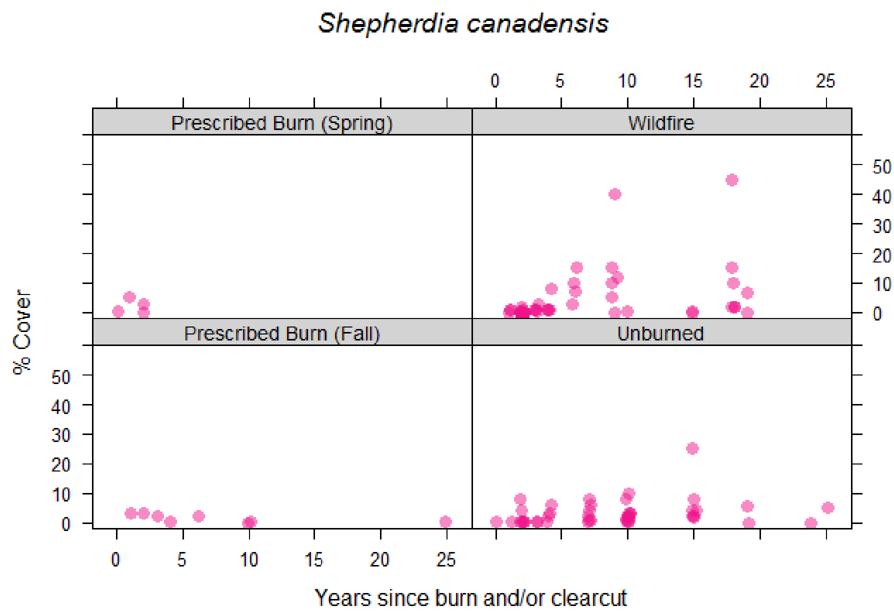

Figure 15. Cover of *Shepherdia canadensis*, where present, in plots that were 1) clearcut or selectively logged and then spring burned (found in 4/218 plots measured up to 10 years), fall burned (found in 8/1801 plots measured up to 25 years) or unburned (found in 46/836 plots measured up to 25 years); or 2) unlogged but wildfire burned (found in 85/502 plots measured up to 24 years.

Three sites containing soapberry had two or more treatments - including burned and unburned (i.e., Inga Lake, Joel Lake, Bednesti); these three sites span a range of site series and treatments as well as monitoring periods. Cover and presence were low in these sites and no significant differences attributable to treatment effects were evident. In most of the other sites in which soapberry occurred there was either only one treatment (e.g., wildfire - Swiss Fire, Mesilinka or prescribed spring burn - Dielman, Stuart River), or the unburned control which was a much older forest (i.e., Boreal LTSP) or the post-treatment cover and presence was extremely low (i.e., Hubert Hills, Dielman) and/or the site was only sampled once at year 2 (Mesilinka, Stuart River). Because of these limitations, no generalizations regarding effects of burning in comparison to other treatments can be made for this species.

In conclusion, *S. canadensis* was found in some of the drier early seral plant communities in our database but not the moister ones. This is consistent with results reported elsewhere that characterize soapberry as a plant of drier habitats (Klinkenberg 2017)[29]. Soapberry cover appeared to increase somewhat over time since disturbance. No clear change in cover or presence associated with burning was evident. Our results are consistent with studies that found soapberry to be moderately fire resistant and able to re-establish after burning[30] since it was observed on our burned sites.

3.1.5 Alternative hosts for stem rusts affecting pines[31]

[29] USDA Fire Effects Information System. Available at https://www.fs.fed.us/database/feis

[30] USDA Fire Effects Information System. Available at https://www.fs.fed.us/database/feis

[31] This analysis was done with Dataset A.

3.1.5.1 Introduction

Pine stem rusts cause significant damage to the trees reducing the health of the forests and the economic value of the timber. A number of understory plants are alternative hosts for different stem rusts.

False toad-flax (*Geocaulon lividum*) is an alternate host for comandra blister rust (caused by *Cronartium comandrae* fungus) that affects hard pines (i.e., 2- and 3-needle pines including lodgepole and ponderosa pine). Members of the Orobanchaceae family including scarlet paintbrush (*Castilleja miniata*), lousewort (*Pedicularis bracteosa*), yellow-rattle (*Rhinanthus crista-galli*) and cow wheat (*Melampyrum lineare)* are alternate hosts for stalactiform blister rust (caused by the *Cronartium coleosporioides* fungus) that also affect hard pines[32]. *Ribes* species are alternate hosts for white pine blister rusts (caused by the *Cronartium ribicola* fungus) that affect soft pines (i.e., 5 needle pines like white, limber and whitebark pine).

We evaluated available data (Database A) to determine how different forest management practices (i.e. burning) affect the abundance of these alternative host species.

3.1.5.2 Geocaulon lividum

False toad-flax (*Geocaulon lividum*) was rarely observed on any of the sites in our database. It occurred with very low cover in 3 unburned sites[33], and 3 burned sites - including 1 wildfire and 2 prescribed burns. *G. lividum* occurred across a range of ecosystems from dry SBS (dk, dw), to moist ICHmc and wet ESSFwc sites. It was not observed on any of our early seral BWBS sites, although it commonly occurs in that zone. No conclusions regarding its response to fire could be drawn from our analysis of the database. False toad flax is common in dry to mesic forests and bogs in the montane and subalpine zones throughout BC (Klinkenberg 2017). In BC it is most abundant in the BWBS zone (Haeussler pers. comm 2018). It is thought to be top killed by fire, its survival dependent on the depth of burial of rhizomes from which it can resprout; elsewhere its abundance appeared to increase over time since disturbed.

3.1.5.3 Orobanchaceae family

- Scarlet paintbrush (*Castilleja miniata*) was the most abundant alternate host for stalactiform blister rusts. It was found on 15 sites with varying amounts of cover (up to 40%) from 1 to 50 years post-disturbance. We found it almost exclusively on SBS sites and particularly on older drier SBS (dk, dw) sites although also in the moist to wet SBSmk, wk, and vk subzones. It was rarely observed on recently disturbed (< 10 year old) sites. Scarlet paintbrush occurred across the range of unburned logged, sheared/cleared and uncut sites as well as in various types of burned sites. It was found in 5 CC, 1 MC, 2 S, and 2 UC sites and in 6 FB, 3 SB, 3 WF and 5 UB sites[34]. It is typically found in wet to dry meadows, grassy slopes, wetlands, clearings, and open forests from the lowland to subalpine zones and is common throughout most of BC (Klinkenberg 2017).

[32] These 4 members of the *Scrophulareaceae* family were observed in the database.
[33] Unburned sites are Kinskuch, Otter Creek and Bednesti, wildfire is Swiss Fire, prescribed burn sites are Kinskuch and Stuart River.
[34] Clearcut=CC, mechanically cleared=MC, uncleared=UC, fall burn=FB, spring burn-SB, wildfire=WF, unburned =UB.

- Lousewort (*Pedicularis bracteosa*) was observed with very low cover on 3 sites including burned and unburned sites in the BWBSmw (Bl, IL) and ESSFwc2 (OC) subzones.
- Yellow-rattle (*Rhinanthus crista-galli*) was observed with very low cover on 5 sites including prescribed burn, wildfire burn and unburned sites in SBSdk & wk and BWBSmw subzones.
- Cow wheat (*Melampyrum lineare*) is the second most abundant alternate host for stalactiform rusts. It occurred with very low cover on 5 sites (some with multiple treatments) including sheared/clearcut logged and unlogged (S (2), CC (2) UL (3)) and burned and unburned (WB (1), SB (1), FB (2) and UB (5)) sites in the BWBSmw/101, SBSdk/81, SBSdw3/05 and SBSmk/01-07 sites.

There is not sufficient data in this dataset to draw reliable conclusions about the implications of forest disturbances and management treatments, including fire, on the presence or abundance of these plants.

3.1.5.4 Ribes spp.[35]

To further investigate specific species addressed in section 3.1.1.7 - Berry Producers, we examined *Ribes* species in the ICH zone. We found no significant difference in the cover of *Ribes* spp. by site series in early seral stages in the ICH zone (Figure 16). Mean cover was generally less than 5%. Comparison of early seral wildfire, broadcast burned and unburned sites in the ICH showed no difference in cover of *Ribes* spp. associated with treatment history (Figure 17). Average cover was less than 5% in all treatments.

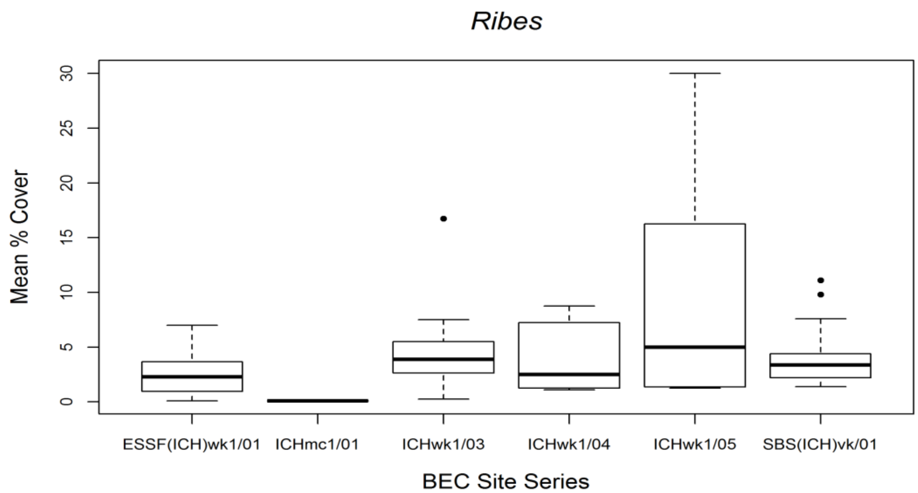

Figure 16. Cover of *Ribes* spp. by site series in selected subzones in the ICH zone. Transitional ICH sites are indicated by parentheses. Bold solid lines = median; box = upper/lower quartile; whiskers = maximum/minimum data points - excluding outliers (points).

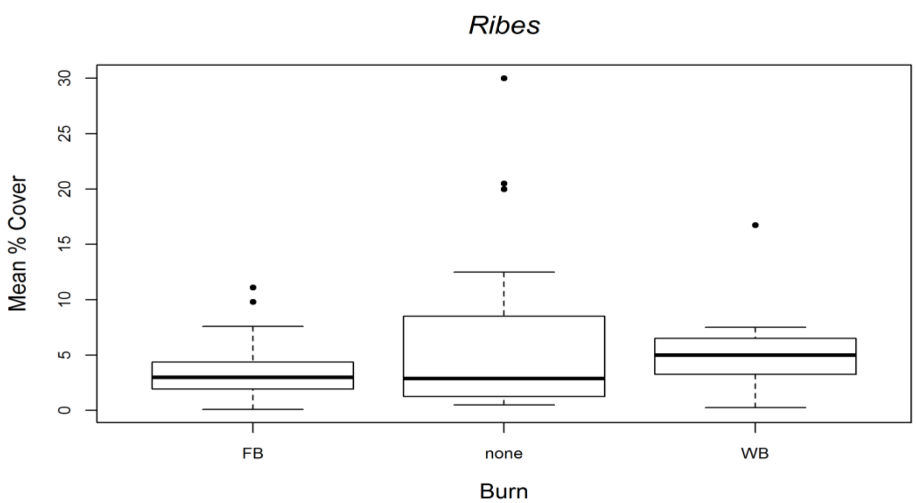

Figure 17. Cover of *Ribes* spp. in clearcut and fall broadcast burned (FB), clearcut and unburned (none) and wildfire burned (WB) sites in the ICH zone. Bold solid lines = median; box = upper/lower quartile; whiskers = maximum/minimum data points - excluding outliers (points).

[35] Based on Dataset A - all ICH data – BD and PLOSONE (no ICH in other SH database).

Further analysis of 10 sites (1561 records) with repeated measures (years 1, 5, 10, and ~20) was also done[36]. Data on three *Ribes* species (*Ribes glandulosum*, *R. lacustre*, *R. laxiflorum*) is reported (Table 4); other *Ribes* spp. - *R. hudsonianum* and *R. oxyacanthoides* - were recorded five times on three sites.

Table 4. Overall % cover for all years at each site by *Ribes* species.

	Ribes lacustre		***Ribes laxiflorum***		***Ribes glandulosum***	
	mean	sd	mean	sd	mean	sd
Brinks Mill (*n*=18)	1.3	(2.7)	0.3	(0.6)	0*	(0.0)
Francis Lake (*n*=36)[Ŧ]	1.3	(2.3)	0.3	(0.4)	0.2	(0.5)
Genevieve Lake (*n*=24)[Ŧ]	0.6	(0.7)	0.5	(1.1)	0*	(0.0)
Goat River (*n*=42)[^]	0.1	(0.3)	2.6	(2.3)	0*	(0.0)
Helene (*n*=135)[Ŧ]	0.4	(0.6)	0.3	(0.6)	0.6	(1.0)
Herron (*n*=70)[^Ŧ]	1.4	(3.2)	0.6	(1.1)	0.6	(0.8)
Mackenzie (*n*=42)[^]	2.0	(1.9)	1.0	(2.0)	0*	(0.0)
Otter Creek (*n*=224)[^Ŧ]	2.3	(2.8)	0.0	(0.1)	0*	(0.0)
Walcott (*n*=130)[Ŧ]	0.1	(0.4)	0.3	(1.0)	1.3	(1.6)
Walker Creek (*n*=840)[^]	1.8	(4.4)	3.8	(6.4)	0.0	(0.0)

*Species not recorded at this site.
^These sites have preburn measures
Ŧ Site was planted with P. contorta var. latifolia

Ribes lacustre and *R. laxiflorum* were found on all 10 ICH sites: there was no significant change in cover over time (Figure 18). *Ribes glandulosum* was only found in 5/10 sites and they were in the drier subzones: no significant change in cover over time was noted (Figure 18). *R. laxiflorum* is more typical of ICH zone; *R.glandulosum* is more typical of the SBS zone (Chandler et al. 2017). Presence of *Ribes* pre-burn was determined to be a good predictor of presence post-burn. Where present pre-burn, *Ribes* spp. increased in cover over time on clearcut and burned SBS, ICH and ESSF sites due in part to stimulation of buried seed germination as well as by resprouting. Burning did not reduce *Ribes* cover over pre-burn levels in the midterm (20 years) (Chandler et al. 2017).

[36] These sites include Brinks Mill (*n*=3), Francis Lake (*n*=6), Genevieve Lake (*n*=4), Goat River (*n*=6), Helene (*n*=5), Herron (*n*=10), Mackenzie (*n*=6), Otter Creek (*n*=34), Walcott (*n*=5), and Walker Creek (*n*=120).

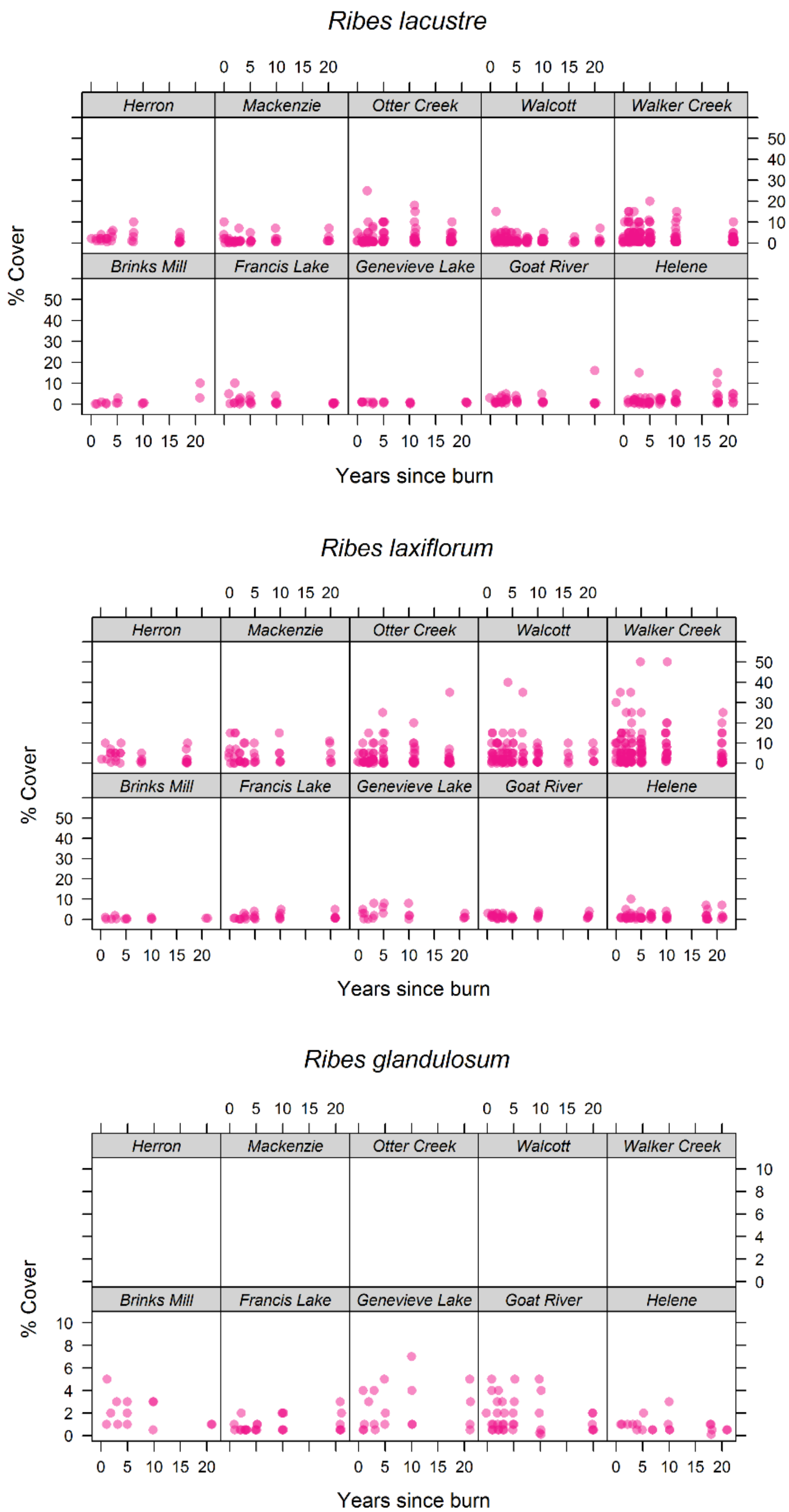

Figure 18. Repeated measures of % cover of three species of *Ribes* (*R. lacustre*, *R. laxiflorum* and *R. glandulosum*) up to 20 years after clearcutting and prescribed burning at 10 sites.

4. Conclusions

4.1 North and Central Interior

An overview of the Central and Northern datasets showed that the plant community composition across central and northern BC was most highly related to long-term fire history/fire climate and local site moisture and nutrient gradients. Differences between recently burned and unburned sites were statistically significant but comparatively small in relation to broader geographic and site level gradients (2.6% versus 26% of total variation). In the most general terms, our results confirm that these interior forest ecosystems are broadly adapted and resilient to fire, and that major shifts to alternative ecosystems states are not characteristic of the sites included in the database.

Based on the outcomes of the overview ordinations, we classified the Northern and Central Interior datasets (SBS, ESSF, BWBS and a very few ICH zone sites) into three Broad Community Landscapes Types (BCLTs) defined by their historic fire regimes:

BCLT1 Mesic and wetter spruce, subalpine fir (western hemlock) communities in mountainous terrain with infrequent high severity wildfire;

BCLT 2 Dry lodgepole pine communities in plateau landscapes with moderately frequent high and mixed severity wildfire;

BCLT 3 Aspen-spruce mixedwood and aspen-grassland communities in rolling, low elevation terrain with frequent low and mixed severity wildfires and spring prescribed burns.

Each of these “landscapes” was well suited to tackling one or more of the Burning Questions.

In BCLT 1 we modelled the recovery of berry producing plants consumed by wildlife and humans, and spring and fall Grizzly bear forage after logging with or without broadcast burning of logging slash. Wildfires are rare in this landscape and we had only one wildfire dataset to compare with the slashburning results. For both the berry producers and the Grizzly bear forage we found that the response trajectories varied significantly by Plant Association and whether the site was burned. Moist, rich Devil’s club and Oak-fern plant associations tend to have open, gappy stands with very diverse range of forage plants. The amount of forage continues to increase over longer time periods than in mesic Vaccinium-moss plant associations where abundance stabilizes or declines following overtopping by conifers. Unburned sites tend to have more forage initially but fall off more rapidly.

With respect to berry production and grizzly bear forage - a diversity of management techniques including burns of varying severity and unburned areas on different ecosystems are recommended. Planting decisions are important (i.e. tree species, patchiness). Cover of berry producers increased with time since disturbance, regardless of whether sites were burned or clearcut. Some communities produce earlier, others later; however, severe burns produced more later. After the Swiss wildfire total and people preferred berry producers cover increased over time; people preferred berry plants cover was greatest throughout time in upland sites with severe burns. Optimal grizzly bear forage management requires a variety of habitats including burned and unburned areas and different severity burns applied across the landscape. Some treatments produce more early season and others more late season forage; abundance will vary over time and by ecosystem type. Burning reduces all grizzly bear forage for the first few years but the effect lasts longer for late season forage. Very moist (seepage) sites produced substantially more early season forage which increased more rapidly after a severe burn but was sustained for longest on unburned sites. Late season forage was most abundance on moister sites. Unburned sites had more initially, especially on fresh sites, but cover was declining after 20 years while continuing to increase on burned sites. Moderate burns had more late season forage than severe burns initially, but the latter ultimately produced more, especially on moist sites.

In BCLT 2 we found the submesic to xeric lodgepole pine community types in plateau landscapes across the central Interior have been profoundly impacted by the cumulative effects of mountain pine beetle outbreak, salvage logging, and recent major wildfires, many of which overlapped. There are justifiable concerns about the resilience of these battered ecosystems. Exposed soil and rock increased to 60% ± 20% on sites exposed to two successive wildfires (compared to 10% ± 5% on sites burned just once) creating extreme risks for soil erosion and non-native species invasion – notably hawkweeds. The regrowth of caribou forage lichens was extremely slow (≤ 0.2%, compared to 20-40% in unburned) ten to 12 years after wildfire. Artificial "reseeding" of lichens is now being tested at several locations to speed the recovery process. Lodgepole pine ecosystems are, in many respects, highly resilient to wildfire, but multiple burns occurring within a decade have the potential to undermine that resilience and to causes losses in valued resources such as huckleberries and blueberries that generally flourish after a single wildfire. A high priority for further work will be to analyze newly assembled tree regeneration datasets.

In BCLT 3 there is a growing consensus that aspen forest health, moose browse, and the condition of grasslands associated with south-facing slopes in the aspen-mixedwood landscape are deteriorating. In general, monitoring data from prescribed burns carried out in this landscape are too inconsistent, short term and lacking in detail and untreated controls to determine whether these trends are real and whether greater use of fire improves or worsens such conditions. Answers seem to vary greatly with local ecological conditions and, as in the southeast, on grazing and browsing by wildlife and domestic livestock. A concerted effort to develop and implement appropriate monitoring is needed. A relatively recent shift from wetter to drier summers across central BC may be complicating our ability to use older datasets and aerial imagery to assess current and future trends.

4.2 Southern Interior (Rocky Mountain Trench)

Our analysis (Appendix 2) showed that where significant amounts of Douglas-fir regeneration (< 1.3 m height) occurred pre-fire, decreases in cover due to fire ranged from 53-100%. Objectives regarding promotion of grassland plant communities have not yet been achieved, even on sites monitored for 14 to 17 years post-fire. It is suspected that even under ideal conditions it will require 20 or more years for late seral bunchgrass species to become dominant - and they may not. Grazing by livestock and wild ungulates is almost certainly interfering with plant succession.

Prescribed fire increased the amount of exposed mineral soil to some degree at all sites. Exposed soil creates conditions suitable for the establishment of plant seedlings, either from previously buried seed or seed transported from off-site sources. Exposed soil is therefore necessary for new plant establishment of desirable species, but also increases the risk of infestation by non-native species. A summary of 21 sites showed that the most common invasive plants that continue to increase after prescribed fire are of little or no concern (e.g., dandelion, black medic, yellow salsify). There are two FRPA listed invasive species (sulphur cinquefoil and St John's wort) that were found to be increasing at trace levels on three sites. There is some evidence that prescribed fire may have differing effects on plant species composition in the PP versus IDF zone. Restoration burning to increase bunchgrass seems more successful in the IDF zone than in the PP zone.

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Appendix 1. Description of and contents of datasets used in various analyses. CNI refers to Central and Northern Interior.

Dataset Name	**A**	**B1**	**B2**	**B3**	**B4**	**B5**	**C**	**D**	**E**
Data included	CNI - JC[37] version[38]	CNI -SH version)[39]SBS (wetter), ICH & ESSF	CNI - Chandler et al, 2017 repeated measures, SBS (wetter).	CNI - Swiss Fire wildfire	CNI- Kinskuch CC & Slash Burn	FRDA 018 chrono sequence data[40]	CNI (SH version)	CNI (SH version)	Subset of the SI Rocky Mt Trench restoration burn sites[41]
Zones and subzones included	BWBS, ICH, ESSF, SBS	Wetter SBS, ICH ESSF	ICH, ESSF sites	SBSdk	ICHmc	SBSmc	Dry SBS (dk, mc) SBPSmc	BWBS, SBSdk	PP, IDF
Broad Community Landscape Type	BCLT1	BCLT1	BCLT1	BCLT1	BCLT1	BCLT1	BCLT 2	BCLT3	n/a
Description	Mesic and wetter spruce, subalpine fir (hemlock) communities in mountainous terrain						Dry lodgepole pine communities in plateau landscapes	Aspen-spruce mixedwood and aspen-grassland communities in rolling, low elevation terrain	Dry warm Douglas-fir and Ponderosa pine forests and grasslands
Historic fire regime	Infrequent high severity wildfires						Moderately frequent high and mixed severity wildfires	Frequent low and mixed severity wildfires and spring prescribed burns.	Frequent low severity stand maintaining fires
Number of sites	75 sites	SH 21 (includes all B2 sites except OC	16	1 in B1	1 in B1, B2	??			12 sites, 27 treatment units

[37] Initials of lead analyst Julia Chandler = JC, Sybille Haeussler = SH, Reg Newman = RN, Evelyn Hamilton = EH.
[38] CNI JC version of the CNI database includes all C&NI plots (including all Chandler et al 2017 plots) except 6 only SH has. Includes Swiss fire.
[39] SH version the CNI database: includes all C&NI plots (including Chandler et al. 2017 plots) except Otter Creek and Dupuis ICH sites. Includes Swiss fire. It has an additional 6 sites.
[40] Data is from Hamilton and Yearsley 1988.
[41] Analysis for all sites and for some individual sites.

Number of plots	3770 plots (8378 plot x years observations)	676 plot x year observations		50 plots					195 macroplots: varying #s of sub-plots
Years sampled	Up to 250 yrs		Up to 22 yrs	Up to 22 yrs.	Up to 10 yrs.	Up to 24 yrs.			Up to 17 yrs.
VEGETATION									
Total vegetation			EH						
Conifers			EH						Df Py regen
Deciduous trees and tall shrubs			EH						
All berry producers[42]		SH					SH		
Early seral berry producers [43]		SH							
Late seral berry producers [44]		SH							
Producers of people -preferred berries[45]									
Grizzly bear forage species – early season[46]		SH			SH				
Grizzly bear forage species – late season[47]		SH			SH				
Ericaceous shrubs		SH included in all and late seral berry producers	EH	EH					
Vaccinium spp.		SH included in all, late seral and	EH	EH					

[42] All berry producers include shrubs, dwarf shrubs and a few forbs. In addition to early and late seral berry producers they include producers of berries preferred by people and soapberry *Symphoricarpos albus and* thimble berry *(Rubus parviflorus)*.
[43] Early seral berry producers include rose family plants (Rosaceae), currents (*Ribes* spp.), and elderberry (*Sambucus racemosa).*
[44] Late seral berry producers include devil's club *(Oplopanax horridus),* twinberry *Lonicera involucrata,* red-osier dogwood *(Cornus sericea),* huckleberry (*Vaccinium* spp.).
[45] People-preferred berry species include huckleberries (*Vaccinium* spp.), strawberry (*Frageria virginiana*), raspberries (*Rubus* spp.), saskatoon (*Amelanchier alnifolia*), and high-bush cranberry (*Viburnum edule*).
[46] Early season grizzly bear forage includes willow catkins, grasses, sedges, horsetail, ferns and a wide variety of forbs. Data was weighted by importance to bears.
[47] Late season grizzly bear forage includes berries, roots and seeds. Data was weighted by importance to bears.

		people preferred berry producers							
Rubus spp.		SH included in all and people preferred berry producers	EH	EH					
Sambucus racemosa		SH included in all and early seral berry producers	EH	EH					
Shepherdia canadensis	EH								
Ribes spp.	JC	SH included in all and early seral berry producers	EH	EH					
Oplopanax horridus	JC	SH included in all and late seral berry producers	EH	EH		EH			
Forbs		SH some included in all berry producers and early season Grizzly bear forage					SH		
Orobanchaceae (e.g., *Castilleja*)	JC								
Geocaulon lividum	JC								
Invasive forb species									RN
Graminoids (grasses and sedges)		SH included in early season grizzly bear forage						SH	RN
Bunchgrasses									RN

(total and 8 individual bunchgrasses)									
Mid seral bunchgrasses									RN
Late seral bunchgrasses									RH
Horsetails and ferns		SH some included in early season grizzly bear forage							
Early vegetation response							SH		RN
Caribou terrestrial forage lichens							SH		
SITE									
Exposed mineral soil							S		RN
Fuel consumption (Df regen)									RN

Appendix 2. Newman and Hamilton 2018. Available as a separate document.