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The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled **Monitoring Ecosystem Restoration of Montane Forests in southeastern British Columbia** submitted by Hillary N. Page in partial fulfillment of the requirements for the degree of *Master of Science* in Rangeland and Wildlife Resources.

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Abstract

It is widely accepted that fire-suppression in the dry forest zones of the Rocky Mountain Trench of BC has contributed to forest ingrowth and encroachment, resulting in closed canopy forests and altered understory plant communities. Sampling of understory and overstory vegetation in dry forested plant communities revealed positive relationships between understory light and several understory variables, including species diversity and bunchgrass cover. Increased tree thinning prescribed to restore open forests was associated with a significant decline in bunchgrass and forb production and cover during the first 2 years post-thinning, it was hypothesized that this was due to the disturbance effects of logging operations. Despite drought conditions, thinning resulted in positive changes in the understory, particularly at the interior Douglas fir site. An experiment using bunchgrass plugs for restoration in recently thinned forests found that plug survival and tiller growth was greater for Richardson's needlegrass than in bluebunch wheatgrass. Fall and spring planting after pinegrass removal generally enhanced bunchgrass development.

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1. INTRODUCTION

"The acid test of our understanding is not whether we can take ecosystems to bits and pieces of paper, however scientifically, but whether we can put them together in practice and make them work."

A.D. Bradshaw, 1983

Restoration ecology is defined by Meffe and Carroll (1994) as, "the process of using ecological principles and experience to return a degraded ecological system to its former or original state". Restoration ecology inherently acknowledges that the ecosystem has been altered in some way as a result of direct or indirect human influences and makes explicit judgements about the desire to reverse this change. Restoration ecology also acknowledges that returning the system to its historical condition generally involves deliberate manipulations to compensate for past human influences (Meffe and Carroll 1994). The restoration of the dry forested systems of the southern interior of British Columbia (BC) is an example of restoration ecology in practice.

Ecosystems can be characterized by their natural disturbance regime. For the purposes of setting biodiversity objectives in BC, five Natural Disturbance Types (NDTs) are recognized occurring in the province. The types range from NDT1, systems with rare stand-initiating events to NDT4, systems with frequent stand-maintaining events. NDT5 systems include alpine tundra and subalpine parkland (Province of British Columbia 1995). NDT4 systems of the southern interior of BC are characterized by grasslands and shrublands mixed with open stands of ponderosa pine (*Pinus ponderosa* Douglas ex Lawson & Lawson var. *ponderosa*) and interior Douglas-fir [*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco]. NDT4 systems historically experienced frequent (every 7 – 50 years), low intensity fires which limited encroachment by most conifer species and shrubs (Leiberg 1899, Cooper 1960, Arno et al. 1995, Province of British Columbia 1995). These plant communities have undergone dramatic changes in structure and losses in diversity hypothesized to be due to forest ingrowth and encroachment brought about with fire suppression policies introduced by the BC Ministry of Forests in the 1940's (Daigle 1996).

Encroachment or outgrowth is tree establishment in previously treeless openings. Ingrowth is excessive tree recruitment, primarily by shade tolerant species, such as interior Douglas fir, within low-density, open forests (Rocky Mountain Trench Ecosystem Restoration Steering Committee 2000). Conifer encroachment has contributed to the rapid disappearance of grassland ranges and open forests in BC (Strang and Parminter 1980, Gayton 1997, Bai et al. 2001). Gayton (1997) estimated that 1% of grassland and open forest are lost annually in NDT4 systems of the Rocky Mountain Trench due to forest ingrowth or encroachment, equivalent to approximately 3 000 ha per year. This rate is similar to estimates made in other areas of British Columbia exhibiting similar ecosystem changes (Bai et al. 2001). Extensive forest ingrowth and encroachment within NDT4 ecosystems of the southern interior of BC has resulted in a loss of wildlife habitat, decreased timber and forage production as well as an increased risk of severe insect outbreaks and catastrophic forest fires (e.g. crown, or stand eliminating fires) (Powell et al. 1998).

To mitigate these changes, land management agencies (Ministry of Forests, Ministry of Water, Land and Air Protection) in the East Kootenay have adopted ecosystem restoration or habitat enhancement programs intended to restore the required ecological processes of fire-maintained NDT4 communities of the Rocky Mountain Trench (Rocky Mountain Trench Ecosystem Restoration Steering Committee 2000). The primary objective of the Trench Ecosystem Restoration Program is to remove excess immature and understory trees from NDT4 communities over the next 30 years to create an ecologically appropriate mosaic of NDT4 habitats on Crown land. The mosaic is intended to mimic the historical landscape under natural conditions when fire was an integral part of the ecosystem (Rocky Mountain Trench Ecosystem Restoration Steering Committee 2000). The most abundant historical information, both photographic and documentary, on forest conditions and disturbance patterns during the period prior to fire exclusion is from the late 1800's (1850) to early 1900's (Veblen et al. 2001), therefore historical conditions generally refer to this time period. The Trench Restoration Program is the largest, longest running terrestrial initiative underway in the province of BC (Machmer et al. 2001). According to current projections, an estimated 135 000 ha within the Trench is currently being considered for restoration.

In this context, ecosystem restoration is generally a three phase process. In the first phase, ingrown forest stands are thinned to between 20% and 70% of the original basal area. The second phase is 30 years of periodic understory burning. The third phase is 30 years of rest from prescribed fire to allow regenerated tree stems to grow to a height where they can withstand low-intensity burns. This final phase coincides with partial harvest of mature stems on the site. Although not all forest stands will be subject to the same prescription because of inherent variability among initial stand structures, post-treatment stands will be consistent with historic NDT4 vegetation.

Most restored sites will be allowed to recover with minimal intervention. However, some sites in the NDT4 may be altered to such an extent that there is a low probability for recovery without intensive intervention to facilitate recovery. The Invermere Forest District initiated a program in 1994 to investigate the possibility of using native seed in ecosystem restoration programs. Initial success of the seeding trials was low. To increase the probability of native bunchgrass establishment, the Invermere Forest District collected seed (local to the area) to grow into seedling 'plugs' under greenhouse conditions. The Invermere Forest District is now looking at the feasibility of planting bunchgrass plugs to accelerate the process of restoring understory native plant communities.

1.1 Goals and Objectives

1.1.1 Goal

The overall goal of this project was to establish a monitoring protocol that will document the extent and rate of change in the understory of ingrown NDT4 plant communities following thinning and burning in the East Kootenay. Achieving this goal will allow land managers to evaluate and adapt ongoing management practices to meet the specific objectives of ecosystem restoration programs in the area. Ideally, this project will be the first in a series of efforts as part of a long-term monitoring plan complementing ongoing restoration work in the East Kootenay region of BC.

This research project (1) retrospectively assessed the pre-thinning ecological consequences of ingrowth, (2) determined the understory response to initial restoration activities (i.e. thinning) and, (3) evaluate the process bunchgrass plug establishment. Collectively, these components will evaluate whether progress is being made towards meeting the general goals of ecosystem restoration.

1.1.2 Specific Objectives

- ? Quantify the pre-thinning relationship between available understory light and characteristics of the understory plant community,
- ? Determine the initial (2 year) effect of forest canopy thinning on the understory herbaceous and shrub layers; including plant species composition, diversity, and the production of important wildlife forage and browse species,
- ? Determine the general efficacy of habitat restoration projects in the East Kootenay,
- ? Assess the feasibility of bunchgrass transplantation for accelerating the restoration of NDT4 understory plant communities.

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2. LITERATURE REVIEW

2.1 Montane Forests of Western North America

Montane forests of the inland Northwest and southern British Columbia dominated by ponderosa pine (*Pinus ponderosa* Douglas ex Lawson & Lawson var. *ponderosa*) and interior Douglas-fir [*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco] are classified as Natural Disturbance Type 4 (NDT4) (Province of British Columbia 1995). NDT4 systems include grassland, shrubland, and relatively open forest communities that historically experienced frequent, low intensity fires, which limited encroachment by most woody species. The varied intensity and frequency of fires across the landscape historically maintained a mosaic of mostly uneven-aged forest stands interspersed with grassy and shrubby openings (Province of British Columbia 1995). The bunchgrass zone, ponderosa pine (PP) forest zone, and dry interior Douglas fir forest (IDF) zone all characterize NDT4 systems of British Columbia. Frequent surface fire return intervals for the PP and IDF biogeoclimatic zones are in contrast to rare stand replacement fires. It is believed that stand replacement fires did occur more frequently in the IDF zone, anywhere from a 150-250 year interval (Province of British Columbia 1995).

2.1.1 Dry Ponderosa Pine (adapted from Hope et al. 1991a)

The PP zone occurs between 49º and 51ºN at low elevations (335 to 900m) along the very dry valleys of the southern interior of British Columbia. The zone represents the northern limits of a region that is much more extensive in the western USA. PP represents the driest forested zone in BC, resulting from a pronounced rainshadow cast by the Coast Mountains. Mean annual precipitation is 280 – 500 mm with 15-40% falling as snow.

Grasslands and open forests occur throughout the PP zone and are thought to have developed as a result of frequent fire and a combination of edaphic and topographic conditions. Dominant species in good range condition are antelope brush [*Purshia tridentata* (Pursh) DC.)], bluebunch wheatgrass [*Pseudoroegneria spicata* (Pursh) A. Löve)], and fescue species (*Festuca* spp.)

2.1.2 Dry Interior Douglas Fir (adapted from Hope et al. 1991b)

The IDF zone dominates the low- to mid-elevation landscape of the south-central interior of British Columbia between 49? and 52? N. The IDF zone also extends south into Washington, Oregon, Idaho, and Montana, and east into Alberta. The IDF has a continental climate characterized by warm, dry summers, a fairly long growing season, and cool winters. The main factor controlling the climate is a rainshadow created by topographic barriers (coast mountains) to the prevailing easterly flow of air. Mean annual precipitation ranges from 300 to 750 mm, of which 20%-50% falls as snow.

Open to closed, mature forests containing Douglas fir cover much of the IDF landscape. After crown fire, there are mixed stands of Douglas fir and lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. *ex.* S. Wats.), often with scattered large, old fire-resistant trees, otherwise known as veterans. On drier sites, ponderosa pine forms early seral stands on zonal sites, but is eventually replaced by Douglas fir.

Similar conditions that formed the PP zone accommodated the development of large grassland communities in drier parts of the IDF. Grasslands in this zone have been modified by grazing of domestic livestock and influenced by fire suppression. Grasslands in good condition are dominated by bluebunch wheatgrass, together with Idaho fescue (*Festuca idahoensis* Elmer) in the south or rough fescue (*Festuca campestris* Rydb.) and Richardson's needlegrass (*Stipa richardsonii* Link.) in the central and northern parts of the zone.

2.2 Historical Montane Forest Management

2.2.1 Pre-1900 Stand Structure and Fire Regime

Prior to European settlement fire was one of the primary disturbance agents that created and maintained low-elevation forests of the inland northwest (Everett et al. 2000). In both the seral and climax dry forest types, frequent surface fires, termed underburns, kept most stands in an open, park-like condition dominated by veteran trees (Leiberg 1899, Cooper 1960a, Arno et al. 1995). In summarizing the literature on historical and current fire regimes in the PP/IDF types, Arno et al. (1995) stated that most stands had recurring surface fires anywhere between 7 and 50 years. Fire history literature and ecological modeling indicate that a 50-year average fire interval is the near maximum that would allow for the continued perpetuation of pine as a major stand component (Keane et al. 1990). While there is ample evidence supporting the generalization that pre-settlement forests were open and park-like for the most part, there were high density stands throughout the Inland Northwest that contributed to a highly diverse landscape structure (City of Boulder 1999, Kaufmann et al. 2000a). Kaufmann et al. (2000a) found a wide range of variation in both the number of trees and tree basal area within these stands, the apparent result of climate effects on seed production and seedling establishment, insect and disease activity, soils and topographic effects on microclimate.

Generally, pre-settlement PP and dry IDF forests of the Rocky Mountains had open understories, but historical photos (Leiberg 1899) show that stands were heavily stocked with large veteran trees. Veteran trees had clear boles as the lower limbs had been shaded out or scorched by fire. Shrub and small tree development was likely inhibited by the extensive tree root systems utilizing much of the soil moisture and nutrients (Smith and Arno 1999). Generally, tree seedlings died while competing with established herbs or were burned by frequent fires. Seedlings infrequently survived within less severely burned microsites with little fuel build-up. As these seedlings grew, their resistance to fire quickly increased (Cooper 1960a).

Although fire-maintained, pre-settlement forests were capable of supporting extensive stands of Douglas fir, the existing fire regime resulted in the development of uneven-aged stands of ponderosa pine and veteran Douglas fir. Ponderosa pine regeneration tends to be episodic in response to optimal climatic events (City of Boulder 1999). After regeneration pulses, most seedlings were killed by periodic surface fires. Occasionally, individuals would survive repeated fires until their bark was thick enough to protect live tissues and their crowns high enough to escape scorching (City of Boulder 1999). Douglas fir seedlings, which have thin, photosynthetically active bark, were more readily killed by surface fires, whereas ponderosa pine of similar size survived due to a well-developed outer layer of fire-resistant, corky bark (Arno and Gruell 1986, Fischer and Bradley 1987, Arno et al. 1995). Additionally, ponderosa pine has large buds that are better protected from fire than the small buds of Douglas fir (Keane et al. 1990). Due to a combination of these factors, the pre-settlement fire regime of low elevation forests across the inland northwest selected against forests dominated by Douglas fir, a phenomenon acknowledged by W.W. White (1924) (as in Smith and Arno 1999). In higher elevation, cooler areas, where the fire return interval was longer, Douglas fir comprised a greater portion of the forest (Keane et al. 1990). However, even in areas with low fire return intervals, where fire regularly selected against fire sensitive Douglas fir seedlings, fire patterns were patchy enough that some fir trees escaped fire and became veterans. Arno et al. (1995) reported that when a Douglas fir tree reaches 15cm in diameter, it develops the capability to survive surface fires.

Lightning is commonly acknowledged as the principal cause of frequent surface fires, but recent studies indicate Native Americans were also an important ignition source (Arno 1983, Gruel 1983, Arno and Gruel 1986, Habeck 1990, Smith and Arno 1999). Native Americans settled and hunted in areas of the Rocky Mountain Trench for at least several thousand years prior to the first permanent settlement by Euro-American settlers in the late 1850's. There are 19th century reports, from the Rocky Mountain Front of the United States, of Native Americans intentionally setting fires for driving game animals and occasionally in warfare (Veblen et al. 2001). However it is not known how abundant anthropogenic ignitions were in comparison to lightning ignitions (Veblen et al. 2001). Archeologists in BC maintain that Native Americans had little influence on the fire-return interval in the dry forest zone of the Rocky Mountain Trench . Given that the area drained by the Kootenay River was very prone to natural fires, it is likely that the precontact residents did not have to carry out deliberate burning to enhance ungulate carrying capacity (Choquette, consultant archeologist, per. comm. 2002).

2.2.2 The Suppression Era

The art and science of managed forestry was developed in the humid regions of Prussia, Germany and northern France, where fires were considered to be destructive and unnecessary (Smith and Arno 1999). Following the settlement of North America, European concepts became the basis for forestry in the United States (and Canada), within environments where natural fires played a vital role in establishing and maintaining certain desirable forest conditions (Smith and Arno 1999).

Beginning in the late 1800's, selective logging preferentially removed most large ponderosa pine and Douglas fir (West 1969). In the interior of British Columbia, many IDF forests were logged earlier in the century by selective harvesting of larger, more valuable trees (Simpson 2000). Logging occurred in concert with heavy, unregulated grazing and reduced the fine fuels that carried surface fires (Fulé et al. 1997, Kaufmann et al. 2000a). Selective logging combined with fire suppression and heavy grazing, allowed for enhanced understory conifer regeneration in fire-maintained forests across North America (Kaye et al. 1999), resulting in a marked change in stand structure from open, older aged forests to closed, young forests. Large, old growth Douglas fir were mostly removed in selective logging, allowing fir seedling regeneration to increase markedly. Although Douglas fir seedlings are shade tolerant and able to regenerate well in a closed canopy, they experience improved growth in open canopies (Chen 1997).

As a result, forests increased in density and became stagnant. Stagnation persists in the absence of natural or artificial thinning and results in a slowdown in tree growth, and is likely due to intense competition for moisture and nutrients (Stone et al. 1999). Tree stagnation indicates low vigor, high stress, and generally poor health, not only for the dense sapling classes, but for older stand classes as well. Veteran trees of pre-settlement origin are susceptible to disease and insect attacks when highly stressed (Stone et al. 1999).

In BC, with the establishment of the Forest Service in 1912, fire management regulations for forested land were expanded (Dorey 1979). Fire control policy called for the complete suppression of all fires to prevent the loss of valuable forest (Parminter 1978). Effective fire suppression combined with selective logging of old-growth Douglas fir, western larch (*Larix occidentalis* Nutt.), and ponderosa pine resulted in dramatic ecological changes. The loss of open grasslands and semi-open forests to forest ingrowth and encroachment was significant (Gayton 1997). Post-1900 tree densities increased in some northern inland forests by as much as 144% (Arno et al. 1995). Two studies completed in BC approximate that 1% of grassland is lost each year to forested land (Gayton 1997, Bai et al. 2001). In the Rocky Mountain Trench, this change is significant when one considers that 3000 ha/year are lost (Gayton 1997).

Forest ingrowth and encroachment has resulted in negative ecological impacts, including insect outbreaks, loss of forage production, and increased incidence of catastrophic wildfire across the landscape (Powell et al. 1998).

2.2.3 Ecosystem Restoration – the New Paradigm

The destructive nature of ecological change associated with fire suppression policies in North America were first recognized by Harold Weaver (1943). Weaver was the first forester to suggest restoring a semblance of the wildfire process in ponderosa pine forests (Smith and Arno 1999). As awareness of the problem increased, numerous studies documented the extent and impact of ingrowth and encroachment (Cooper 1960a, Arno and Gruell 1986, Lunan and Habeck 1973, Habeck 1990, Arno et al. 1995, Gayton 1996, Gayton 1997, Smith and Arno 1999, Arno et al. 2000b). By the 1970's and 1980's many National Forest managers in the United States were exploring the integration of prescribed fire into management regimes (Smith and Arno 1999). Unfortunately, simple fire reintroduction was not the ecologically appropriate response given the magnitude of change in these forests and the lack of historical information.

Managers initially adopted historic stand models used in the southwestern United States, where much of the research on ingrowth and encroachment has occurred (Kaufmann et al. 2000b). Important differences exist, however, between the ponderosa pine forests found in the southwest, particularly in Arizona, and the ponderosa pine and dry Douglas fir forests in the central and southern Rocky Mountains (Kaufmann et al. 2000b). Fires in the ponderosa pine forests of the southwest historically occurred at average intervals of 2 – 10 years (Covington et al. 1997, Feeney et al. 1998, Mast et al. 1999) and stand replacing fires were considered to be rare (Cooper 1960a, Covington et al. 1997). In contrast, fires in the Rocky Mountains were not as frequent, even at the forest-grassland ecotone (Goldblum and Veblen 1992, Brown et al. 1999, Kaufmann et al. 2000a, Kaufmann et al. 2000b). Less frequent fires may have included areas of non-lethal surface fires and areas where fire burned intensely and created local openings, resulting in landscape patterns that may have been considerably different from the southwest (Brown et al. 1999, Kaufmann et al. 2000b).

Several researchers have documented historical stand replacing fires in the dry forest types on the Rocky Mountain Front (Arno et al. 1995, Brown et al. 1999, Kaufmann et al. 2000b, Veblen et al. 2001). Researchers have suggested that it is imperative management focus on ecological objectives specific to the fire regimes in the area (Mast et al. 1999, Kaufmann et al. 2000a). Therefore, managing forests for long-term ecological sustainability requires information about the condition of historical forests and the processes that regulated their structure and change (Covington and Moore 1992, Kaufmann et al. 2000a). Without an awareness of historical stand structure and fire regimes it is difficult to protect or restore forest characteristics that ensure long-term sustainability (Kaufmann et al. 2000b).

Following examples set by natural resource managers in the United States, studies were completed in British Columbia to establish historical stand structure and mean fire return intervals (MFI's). In southeastern British Columbia, Dorey (1979) determined a MFI of 6.4 years with a range of 2 – 13 years from 1813 to 1840. Studies in similar dry pine communities document MFI's of similar values (Arno 1995). Dorey's study was completed in the PP biogeoclimatic zone so the application of a MFI of 6.4 years to other areas of the East Kootenay Valley should be done with caution, especially in respect to other biogeoclimatic zones (e.g. IDF). Ponderosa pine communities are the driest forests in the region and therefore, likely have the shortest MFI and lightest fire severity due to less fuel accumulation in these areas.

Natural resource managers recognized that restoring a natural fire regime would be difficult after nearly a century of fire suppression because of fuel accumulation and forest growth stagnation related to overstocking increased forest stand vulnerability to fire damage (Smith and Arno 1999). As a result, forest managers started to prescribe fire in conjunction with other treatments, such as silvicultural thinning (Smith and Arno 1999, Arno et al. 1995, Arno et al. 2000a). The objective of thinning was to remove excess understory and weaker trees that could not be safely killed in an underburn (Arno et al. 2000a). This process also served to prevent the understory from acting as ladder fuel that might support a crown fire threatening normally fire-resistant trees.

Overall, there is increasing interest among natural resource managers, biologists, and the public in restoring fire-maintained forests to more natural and desirable conditions (Arno et al. 2000a). New perspectives on forest and fire management reflect a shift in emphasis from commodity production toward ecologically-based management aimed at improving forest health, sustaining productivity, and securing biological diversity (Fiedler and Carlson 1992).

2.3 Ecological Impacts of Fire Suppression

2.3.1 Overstory-Understory Relationships

In first recognizing the effects of fire suppression, researchers often described overstory-understory relationships. Patterns of ground vegetation are directly related to the overstory tree canopy in fire-maintained forests. Mature stands are dominated by large, fire-resistant trees, commonly open and savanna-like, with a ground cover mosaic of grasses and other herbs (Moir 1966).

Early studies documenting the influence of tree clusters on the herbaceous understory observed that under conditions of maximum shading and heavy litter, there is near-total suppression of herbs (Moir 1966, Pase 1958). Later studies quantified understory-overstory relationships in the dry forest types of North America by describing herbaceous plant growth as functions of the overstory (see Ffolliott and Clary 1982 for a review). In addition to light quantity, tree canopies may also have significant effects on light quality (Knowles et al. 1999), a relationship not studied as extensively.

The effect of canopy closure on understory species composition and production, however, is not a relationship solely dependent on light. There are many ecological factors that will be affected by increasing canopy closure, including water yield and soil fertility. Hawke and O'Connor (1993) reviewed changes in forest soil nutrients and pH under different radiata pine (*Pinus radiata*) agroforestry systems and concluded that trees significantly decrease soil pH and some nutrient levels (e.g. nitrogen) while increasing other nutrients. MacClaren (1996) found reductions in soil water content and surface water yield under forests compared to adjacent grasslands.

2.3.1.1 Soil and Water Resources

It is difficult to isolate the single factor that will most affect understory vegetation dynamics. Aboveground competition for light will increase while there will be a significant amount of understory competition for water and nutrients (Riegel et al. 1991, 1992, 1995). Different studies have tried to isolate the critical factor limiting understory growth in forested systems. Krueger (1980) proposed that the critical factor limiting understory production in the Cascade Range of Oregon is primarily water, while nitrogen has been proposed by other researchers to be the primary limiting factor in most Inland Northwest forests (Tilman and Downing 1994, DeLuca and Zouhar 2000).

Competition for nitrogen plays an important role in determining plant growth and species composition (Tilman and Wedin 1991, Riegel et al. 1992,1995, Herron et al. 2001). Nitrification is considerably greater in open, fire-maintained forests compared to high density forests (Moir 1966, Kaye and Hart 1998a), likely the result of the contribution of pine litter to organic matter. Pine litter has a relatively high content of resins, lignins and other organic compounds that are generally resistant to chemical breakdown. As long as the C:N ratio remains high, nitrogen will be immobilized by soil microorganisms, resulting in nitrogen deficiencies in the upper soil. Subsequent rates of humus mineralization may be too slow to provide a satisfactory supply of nitrogen to the understory (Hunt et al. 1988, DeLuca and Zouhar 2000). Earlier studies observed this phenomena although the relationship was not documented empirically (Pase 1958, Moir 1966). In addition to decreased N mineralization, increased canopy cover will have a negative impact on water yield due to increased water use by overstory trees (Maclaren 1996).

Ingrowth and encroachment, as well as intensifying competition for belowground resources, such as nitrogen and water, collectively affect carbon storage pools. Recent global warming trends have drawn attention to terrestrial carbon storage. Studies have documented the accumulation of carbon in the forest floor as a result of fire suppression (e.g. Cooper 1960a, Kaye and Hart 1998b). More than a century of fire exclusion has likely affected regional carbon cycles by reducing the number of surface fires and by increasing the number of stand replacing fires (Kaye and Hart 1998b).

Understory responses to ingrowth and encroachment will ultimately depend on which variable is monitored. Riegel et al. (1991,1992, 1995) found that a combination of belowground and aboveground competition for resources limited growth. Thus, it is essential that these various factors be considered when interpreting the impact of ingrowth on understory dynamics.

2.3.1.2 Understory Vegetation Production

In temperate regions, where light is a limiting factor for understory growth, canopy closure and reduced light has generally reduced forage production for wildlife and livestock (Pase 1958, Cooper 1960b, Moir 1966, Borjoquez et al. 1989, Knowles et al. 1999). This is likely why the influence of ingrowth on graminoid, forb and shrub species biomass has received considerable attention (Naumberg and DeWald 1999).

Regression coefficients describing overstory impacts on herbage production are generally strong (r^2>85%). Grasses suffer the greatest negative response from increasing tree cover, while forbs exhibit moderate responses and shrubs generally little response. This trend is likely due to highly variable shrub production in temperate ecosystems, whereas herbage production is less variable (Pase 1958, Bojorquez et al. 1989, Knowles et al. 1999). There appears to be no consistent pattern in the type of relationship found between overstory and understory production: some relationships are curvilinear (accelerating) (Bojoroquez et al. 1990, Pase 1958) while others are linear (Cooper 1960, Moir 1966, Knowles et al. 1999).

When analyzed by soil type (Moir 1966, Bojorquez et al. 1990), overstory-understory relationships are often site-dependent. In other words, factors besides decreased light levels negatively impact forage production, for example, decreased soil nitrification (Moir 1966, DeLuca and Zouhar 2000) and soil water (Maclaren 1996).

2.3.1.3 Understory Species Composition and Diversity

Individual species' and functional group responses to ingrowth and encroachment have not received much attention, and has even been considered relatively unimportant because of low overall productivity under dense conifer stands (Naumberg and DeWald 1999). Those studies that have been completed, have found strong relationships between canopy closure and the cover and diversity of plants (Klinka et al. 1996, Thomas et al. 1999).

In northern inland forests of the U.S. and Canada species presence is related to resource availability (e.g. overstory light transmission) (Moir 1966, Riegel et al. 1991, 1992, 1995, Naumberg and DeWald 1999). Relationships of species presence and abundance to forest characteristics indicate that any change in the structure of the overstory will undoubtedly have implications for the understory. Presumably, each individual species has an extinction point along a resource gradient. As resources decline over time the more productive and resource-demanding species disappear and create room for the establishment and growth of other species more suited to the changing conditions (Knowles et al. 1999). Alterations in forest structure and corresponding changes in the understory can also be discussed within the context of succession.

Disturbance regimes and how they affect community succession are important to the extent that they influence probabilities of species extinction and colonization, and thereby, patterns of diversity in the landscape. Understanding succession is essential to ecological forest management. Successional changes are due to the competitive abilities of plant species, which are determined by such attributes as tolerance for shade and allelopathic chemicals, reproductive strategies, and longevity (Steele and Geier-Hayes

1993). Disturbance frequency determines the direction and extent of successional trends and thus, the historical range of plant communities in a particular system (Steele and Geier-Hayes 1993). When fire is suppressed the interval between disturbances is effectively lengthened, allowing open, fire-maintained forests to shift to closed canopy, mesic stands of shade-tolerant and fire-intolerant species (e.g. younger age classes of Douglas fir). Fire-maintained stands that experience a natural disturbance regime are likely seral or in an intermediate stage of succession. In Rocky Mountain forests, shading caused by the invasion of conifer species (Douglas fir and lodgepole pine) as the stand moves toward climax, has favoured the invasion of mesophytic shrubs and herbs (Lunan and Habeck 1973).

One example of this is pinegrass (*Calamagrostis rubescens* Buckl.), a rhizomatous species that is well-adapted to dense fir canopies (Steele and Geier-Hayes 1993). Lack of light and increased competition from pinegrass may limit the existence and distribution of more desirable species such as native bunchgrasses. The loss of bunchgrasses is significant in the Trench, as native ungulates and livestock exhibit a high degree of preference for these species (rough fescue, Idaho fescue, bluebunch wheatgrass, Richardson's needlegrass) (Clark et al. 1998, Ross 2001). Animal preference for bunchgrasses is likely due to the low protein values of pinegrass relative to other bunchgrasses (Gayton 1997), as crude protein content of pinegrass decreases rapidly with advancing maturity (Freyman 1970). In contrast, bunchgrasses initiate early growth, become semi-dormant during summer drought, and show significant regrowth in the fall when soil moisture increases. Due to this growth pattern, bunchgrasses produce autumn forage with high nutritional levels (Hooper and Pitt 1998). Given that ungulate numbers (including livestock) tend to remain relatively constant in an area over time, the gradual loss of bunchgrasses has implications for the overgrazing of remaining vegetation (potentially causing further undesirable species composition shifts) (Gayton 1997).

Altered resource levels not only affects species composition and succession patterns but also affects plant diversity. Within dry, open forests, the highest levels of diversity occur at the lowest levels of canopy closure (Covington et al. 1997, Uresk and Severson 1998). Species diversity is important because it is recognized to be an important indicator of the resilience and resistance within a plant community (Schulz and Mooney 1993, Tilman and Downing 1994, Naumberg and DeWald 1999).

2.3.2. Wildlife Habitat

British Columbia's mountain valleys are well known for having high concentrations of wildlife. Low elevation habitats occupy a small portion of the province but are nonetheless very important (see Hudson et al. 1976). Three factors that influence the assemblage of wildlife species in NDT4 systems are short winters with low snowfall, a strategic location between the Great Basin to the south and boreal forests to the north, and a great diversity of vegetation types. A rich and varied collection of habitat niches results from the mosaic of grasslands and dry forest (Hope et al. 1991b). Several species of ungulates, including mule deer, white-tailed deer, bighorn sheep, and Rocky Mountain elk, migrate long distances to winter in the NDT4. Other species present include badger, western rattlesnake, gopher snake, and the Great Basin spadefoot toad (Hope et al. 1991a).

Forest ingrowth and encroachment affects wildlife habitat in primarily two ways. First, a large number of rare and endangered species depend exclusively on grassland or open forest habitats and are further threatened by the loss of these areas. Second, grassland habitats are the primary source of forage for many native ungulates, particularly in the winter, and the loss of primary range will result in overgrazing of remaining rangelands.

Variability in spatial and temporal components of disturbance regimes results in greater heterogeneity of habitats and resources for organisms. Heterogeneous habitats contribute to increased species diversity at scales from populations to regions (City of Boulder 1999). A large proportion of endangered and extirpated wildlife species rely on native grasslands and open forest habitat. In British Columbia, 25% of the provincial listed species are located in grassland habitats, with more endangered species found in open range and grassland than any other habitat (Pitt 2000). Although few studies have made a direct link between forest ingrowth and encroachment and the decline of wildlife populations, several studies have documented the importance of this habitat for endangered and threatened species (Newhouse and Kinley 2000, Cooper and Gillies 2000, Krannitz and Rohner 2000). Newhouse and Kinley (2000) studied the endangered British Columbia badger (*Taxidea taxus* ssp. *jeffersonii*) population. They observed that the small population numbers could be due to the declining number of Columbian ground squirrels (*Spermophilus columbianus*) (their primary food source) in the region. Columbian ground squirrels rely primarily on non-forested habitats and a decline in squirrel numbers will adversely impact several carnivore populations, including badgers.

Open grassland and forest habitats are critical winter habitat for native ungulates. Episodes of tree invasion have undoubtedly decreased carrying capacity for native ungulates in areas that have been historically open (Arno and Gruell 1986). In the Black Hills of South Dakota, livestock have a decided preference for forage grown in open meadows as opposed to forage grown under pine stands (Thompson and Gartner 1971). Shading seemed to reduce forage quality by lowering nitrogen-free extract, as well as plant sugar content (Thompson and Gartner 1971). Reduction of carrying capacity of open habitats will undoubtedly also arise with overgrazing of a shrinking area of primary ungulate and livestock range.

2.3.3 Forest Health and Production

Dry forest zones of North America have deteriorated in stand health as a result of fire suppression policies and selective logging of old-growth trees (Cooper 1960a, White 1985, Gayton 1996, Feeney et al. 1998, Arno et al. 2000b).

Dramatic increases in density have resulted in decreased tree growth rates and increased mortality of veteran trees (Feeney et al. 1998). A possible mechanism behind the decline in stand health could be the excessive competition between pre-settlement and post-settlement origin trees for limited resources, likely resulting in greater physiological stress (Arno et al. 2000b, Feeney et al. 1998).

Current conditions have also increased the risk of insect outbreaks (Feeney et al. 1998). In northern inland forests, the presence of old-growth trees killed by western pine beetle coupled with declining annual growth in previously fire-maintained stands indicate these dry forests are significantly overstocked (Smith and Arno 1999). In Oregon, more than 40 000 ha in the Blue Mountains now consist of dead and dying trees, primarily Douglas fir thickets, killed by insect and disease epidemics (Arno et al. 2000b). The primary concern in the Rocky Mountain Trench of BC is the apparent increase in Armillaria root disease, and fir beetle outbreaks. Forest health managers suggest that dry forest maintained at lower stocking levels ($\leq$250 stems/ha) will result in fewer Armillaria outbreaks (Begin, Forest Health Officer, Invermere Forest District, per. comm. 2001).

2.3.4 Fire Hazard

According to forest managers, the greatest negative impact of post-settlement fire suppression policies has been the increased risk of catastrophic wildfires (Anderson, Operations Manager, Invermere Forest District, per. comm. 2001). This is also the highest profile aspect of fire suppression. Wildfires that raged through Montana in the fire season of 2000 were a testament to the increasing dangers of massive fuel accumulation.

Development of a dense, shade-tolerant understory is a significant compositional and structural change in formerly open stands of old-growth dry forests (Weaver 1943, Keane et al. 1990). Dense fir thickets developed in the absence of fire have increased the risk of stand-replacing fires (Arno et al. 1995, Scott 1998, Feeney et al. 1998). Torching trees and crown fires are a significant source of fire-brands, which are considered to be a major ignition source of wildland homes and wilderness communities (Scott 1998).

Intense fires will also have adverse impacts on the vegetation community. Studies have shown that heavy fuel loads result in increased mortality to pre-settlement trees (i.e. veterans) (Scott 1998) along with severe disturbance of the understory community.

Intense understory disturbance enables exotic species invasion (Thomas et al.1999) and possibly, the development of temporary hydrophobic surface soils. Soil hydrophobicity would limit water infiltration and significantly impede vegetation recovery after fire (Wallis and Horne 1992).

2.4 Ecosystem Restoration

The Society of Ecological Restoration (1996) defines restoration as, "The process of assisting recovery and management of ecological integrity. Ecological integrity includes a critical range of variability in biodiversity, ecological processes and structures, regional and historical context, and sustainable cultural practices.". Although ecological integrity is a subjective term, in the context of fire-maintained ecosystems, restoration practices are being widely used to decrease the frequency of stand-replacing fires and overstory tree mortality while increasing herb production, diversity and water and nutrient availability (Fiedler and Carlson 1992, Covington et al. 1997, Kaye et al. 1999, Ritchie and Harksen 1999). The general goal of restoration in fire-maintained stands is to develop more open-stand structures consistent with historic disturbance regimes.

Within the Rocky Mountain Trench, prescribed fire and forest thinning are being used to restore open forests and grasslands. In cases where the plant community has degraded to a point where the desired plant community will not recover, intensive revegetation practices are being introduced (e.g. seeding and planting bunchgrass plugs).

2.4.1 The Restoration Process – Tree Thinning

Restoration usually begins in densely ingrown forested stands with thinning, which removes understory trees that may serve as ladder fuel. Seral, disturbance-dependent ponderosa pine and Douglas fir are favoured for retention, in order to maintain open stand structures and wildlife habitat. These trees are fire-resistant and long-lived, commonly surviving 400 – 600 years in conjunction with frequent underburns. Low-intensity prescribed fire is subsequently introduced to reduce fuel loadings, kill excessive saplings, rejuvenate undergrowth (herbaceous and shrub species), and recycle nutrients (Arno et al. 2000a).

2.4.1.1 Restoration Monitoring

An integral component of restoration management plans is a detailed monitoring plan. Monitoring will aid in the development of future plans, plans that contain an understanding of the ecological processes that link overstory management to understory dynamics and diversity (Naumberg and DeWald 1999). The objectives of restoration monitoring are to assess characteristics related to forest and ecosystem health, forage production, maintenance of open forest habitat and timber production (Ritchie and Harksen 1999). Although restoration treatments have been undertaken in many areas, treatment effects have rarely been monitored and evaluated. Furthermore, most research studies have examined only a narrow set of treatment effects. Prescribed burning has been studied widely, but seldom in combination with the logging treatments necessary to restore a more natural stand structure before burning (Smith and Arno 1999). Studies that examine detailed temporal vegetation responses to thinning are necessary to better evaluate both time lags and disturbance effects (Thomas et al. 1999).

The challenge in designing a monitoring strategy is finding an acceptable middle ground between practicality and statistically-sound research. Replication is necessary as it guarantees the validity of testing for significant treatment effects, based on the estimate of error (Fisher 1971). Replication is also the best insurance against chance events producing spurious treatment effects as it minimizes the effects of “natural noise”, increasing the precision of an estimate (Hurlbert 1984). Although the benefits of replication are well-known, the replication of treatments on a large-scale is often a luxury. Unreplicated trials should not prohibit the accumulation and publication of descriptive information gathered from monitoring efforts or other unreplicated trials (Brown and Waller 1986). When a monitoring strategy is based on unreplicated treatments, the monitoring plan should ensure that the experimental unit (the population of inference) is explicitly defined.

When monitoring strategies attempt to detect a significant local treatment effect and extrapolate the results to larger areas, they are labeled as pseudoreplicated, or, “the testing for treatment effects with an error term inappropriate to the hypothesis being considered” (Hurlbert 1984). More simply, if inferences are based on multiple sub-samples of a single experimental unit, then the experimental design is said to be pseudoreplicated. The basis of pseudoreplication is that sub-samples of an experimental unit are not independent (Hurlbert 1984). Experimental units or plots that are not independent could result in the misrepresentation of other similar experimental units, which may bias the significance of the treatment effect (Brown and Waller 1986). Hurlbert (1984) suggests that the solution to this dilemma is to utilize only a single datum for each experimental unit and to omit completely any formal analysis of the data for individual samples and sub-samples. Using only a single datum per unit would limit the amount of useful data that could potentially be gathered from a single treatment block at the landscape level. A more moderate approach would be to reduce the population of inference to a single, sub-sampled experimental unit and limit inference only to that treatment block (Stroup et al. 1986). Researchers may subsequently evaluate this information and individually decide that similar large-scale trials are justified. Therefore it is essential to define the population of inference when designing any monitoring strategy. Proper experimental design can be destroyed by failing to recognize what constitutes an experimental unit (Brown and Waller 1986).

Another problem associated with large-scale, long-term projects is the unpredictability of financial and administrative support. Without the long-term allocation of the land base and funds, research involving restoration monitoring has little assurance of success (Ritchie and Harksen 1999).

2.4.1.2 Restoration Efficacy

When monitoring efficacy, the goals of restoration must be clearly stated to ensure efficacy can be properly assessed. There are several variables that can be affected by restoration activities and thus, could be monitored.

2.4.1.2.1 Soil and Water Resources

Studies have shown that thinning increases soil moisture (Della-Bianca and Dils 1960, Riegel et al. 1992, Feeney et al. 1998, Kaye and Hart 1998b, Ffolliott et al. 2000, Stone et al. 2000). In Northeastern Oregon, Riegel et al. (1992) reported that increased soil water (in response to thinning) added two months to the growing season, leading to significantly greater understory biomass.

Although no clear pattern has emerged in the literature, increased soil water and temperature (a result of increased light) may increase rates of soil respiration (Kaye and Hart 1998b). Using soil respiration as an integrative measure of soil biological activity, Kaye and Hart (1998b) reported that respiration rates increased significantly in response to thinning, but only in a drought year. The authors hypothesized that in dry years, soil respiration may be limited by moisture availability.

Studies documenting the effects of thinning on available plant nutrients have been mixed with several finding increases in mineralizable nitrogen (Riegel et al. 1992, Kaye and Hart 1998a). Increased soil moisture has also been hypothesized to make root absorption of water and nutrients easier (Riegel et al. 1992). Additionally, high soil temperatures and increased pH greatly increase the rate of nitrification (Thain and Hickman 1980). Riegel et al. (1992) reported a significant increase in soil pH in response to thinning and Kaye and Hart (1998b) reported significantly greater soil temperatures. A combination of pH, moisture and soil temperature may improve plant and soil health by increasing nitrogen absorption and mineralization. Another study in Montana found that thinning had no significant influence on available soil nitrogen and microbial activity unless combined with prescribed fire (DeLuca and Zouhar 2000). In low production systems, nitrification may not change with thinning as nitrification may be impeded by carbon-rich secondary and structural compounds (lignins, phenolics) produced by vegetation (Vitousek and Matson 1985).

Due to reported increases in available nutrients and water, a common concern associated with restoration treatments is nutrient and water outflow following thinning. When quantified, however, the nutrient loss following thinning appears small (Kaye and Hart 1998b, Kaye et al. 1999). This is likely due to two reasons: (1) nutrient concentrations in the soil below the rooting zone are low in high density, pre-restoration stands, and (2) water outflow from deep percolation is small and temporally limited (Kaye et al. 1999). For losses to occur, increased nutrient availability must occur in concert with times of water movement (Kaye et al. 1999). Additionally, nitrogen losses are considerably lower from less productive systems as the C:N ratio of plant residue is relatively high and microorganisms that decompose residue can immobolize large amounts of nitrogen (Vitousek and Matson 1985). In general, the dryland systems being considered for forest restoration are relatively low production systems. Increased production by grasses (e.g. McConnell and Smith 1965, 1970, Riegel et al. 1992) and by the remaining trees after thinning (e.g. Della-Bianca and Dils 1960, Feeney et al. 1998) may also be major factors decreasing nutrient loss following restoration (Kaye et al. 1999). When plant regrowth is slow or experimentally inhibited following disturbance, nutrient losses are larger (Kaye et al. 1999). As a result, post-thinning disturbance (e.g. grazing) of recovering vegetation could have a significant impact on the recovery of a site during restoration.

2.4.1.2.2 Understory Vegetation Production

It is widely recognized that understory production increases when dense forest stands are thinned (McConnell and Smith 1965, 1970, Thompson and Gartner 1971, Riegel et al.

1992, Uresk and Severson 1998, Ffolliott et al. 2000, Ross 2001). However, recent studies show that the mechanism behind the increase in production is unclear. Although it is obvious that the most immediate effect of overstory opening is to increase light, the formation of root gaps in the soil may increase water and nutrient availability. Whether it is aboveground or belowground competition that dictates production within inland northern forests remains to be determined.

Many studies group production responses to thinning by vegetation groups, rather than by individual species, in particular, grasses and sedges, shrubs, and forbs, with the greatest overall biomass increase for grasses (McConnell and Smith 1965, 1970, Riegel et al. 1992). Studies in northern inland forests have found that a large proportion (13-44%) of the graminoid response could be attributed to an increase in pinegrass (McConnell and Smith 1965, 1970). At low levels of thinning, however, forbs tended to have the greatest increase in production relative to other growth forms. This may be because of the horizontal orientation of forb leaves, which enables these plants to achieve a fuller canopy of foliage (McConnell and Smith 1965). In the long-term, several studies indicate that shrub production (current annual growth) does not respond significantly to thinning (McConnell and Smith 1970, Riegel et al. 1992, Thomas 1999).

Despite generally significant increases in production over the long-term, there may be only minor initial increases in production immediately following thinning (e.g. McConnell and Smith 1965, Ross 2001). Increased resource availability is often countered by physical disturbance from thinning in the short-term, and may negate increased growth within the plant community. At very low thinning intensities, physical site disturbance may outweigh any minor benefits provided by increased resources

(Thomas et al. 1999, Thysell and Carey 2001). In the Rocky Mountain Trench, Ross (2001) observed that forage production did not increase significantly until 2 years after restoration treatments. Overall, the ability of the understory to respond to overstory removal is critical for maintaining a forage base for wild ungulates and livestock (Riegel et al. 1992).

2.4.1.2.3 Understory Species Composition and Diversity

By increasing available resources, thinning could allow a greater number of understory species to persist. Alternatively, at increased resource levels, one or two responsive species may monopolize resources (Tilman 1993, Thomas et al. 1999), resulting in decreased diversity (Alaback and Herman 1988). Ultimately, the outcome of thinning will depend on initial stand structure, the magnitude of change in the overstory, pre-treatment understory species composition, and on the specific management practices (i.e. type of thinning) imposed (Thomas et al. 1999). These complexities likely account for some of the variability in vegetation responses to thinning documented among studies.

Plant cover and species richness often increase at higher levels of thinning (Uresk and Severson 1998, Thomas et al. 1999). Uresk and Severson (1998) reported that uncommon species occurred only at low stand densities. Griffis et al. (2001) reported that although thinning failed to change species richness, it did increase the abundance of perennial graminoids (native and introduced). In the same study, introduced graminoid species richness increased at the highest levels of disturbance (Griffis et al. 2001). Opening of the overstory may favour early-successional species and possibly, exotics

(Thomas et al. 1999, Thysell and Carey 2001). Early germination, rapid growth, and allocation of resources to aboveground biomass enable weeds to preempt resource use by their competitors (Sheley et al. 1993, Herron et al. 2001).

Nutrient availability may be a driving force in plant community composition dynamics (Herron et al. 2001). Tilman and Wedin (1991) found that late seral species were superior competitors for N. Late seral species have high belowground biomass, and thereby create soils with high C:N ratios, and consequently, low N mineralization. The presence of late-seral species results from their ability to reduce quantities of extractable soil ammonium and nitrate. In examining the effect of nutrient availability on the interaction between bluebunch wheatgrass and spotted knapweed (*Centaurea maculosa auct. non* Lam.), Herron et al. (2001) concluded that bluebunch wheatgrass can outcompete the early successional spotted knapweed at low nutrient levels. This phenomenon may partially explain why early-successional native and exotic species initially increase after thinning.

In general, relatively little work has been done on species composition responses to restoration treatments, with most studies having examined production instead. High levels of variability within understory plant communities of dry forests (Uresk and Severson 1998) may prevent the gathering of useful information from monitoring trials. Responses of understory species to increased levels of light, water and nitrogen will vary depending upon their physiologic tolerances and competitive ability (Riegel et al. 1992). Plants that initiate growth early in the season have the potential to be more successful in competing for limited nutrients (Riegel et al. 1992), as exemplified by the early emerging pinegrass and introduced species such as spotted knapweed.

2.4.1.2.4 Forest Health and Production

Thinning is often used in dry forest restoration programs as it achieves a number of objectives. Partial or selective cutting can be used to reduce the proportional composition of shade-tolerant species (i.e. Douglas fir) and to increase the regeneration of shade intolerant trees such as ponderosa pine. Thinning can also be used to increase the vigor of remaining trees, making them less susceptible to insect attacks, to increase the distance between tree crowns, thereby decreasing the chance of a stand-replacing fire, and to promote the development of the large tree component (Smith and Arno 1999). Several studies have examined the effects of stand density, basal area and thinning on water potentials, leaf nutrient concentrations, leaf gas exchange (photosynthetic capability) and resistance to insect attacks (Della-Bianca and Dils 1960, Donner and Running 1986, Feeney et al. 1998, Kolb et al. 1998, Stone et al. 1999).

The principle behind thinning is that reducing the number of stems may improve site water relations by reducing both overall transpiration and live root density within the soil, increasing the available water for remaining trees, and reducing canopy interception to allow more rainfall to reach the soil (Donner and Running 1986). Studies have shown that thinning conifers will increase leaf water potentials proportional to the amount of basal area removed (Helvey 1975, Donner and Running 1986, Feeney et al. 1998, Kolb et al. 1998). In Montana, reduced evapotranspiration losses led to significantly greater late-summer water potentials (0.17 – 0.35 Mpa) in thinned lodgepole pine stands relative to the controls (Helvey 1975). Helvey (1975) also noted that significant differences in water potential were not permanent. After three growing seasons, increased growth by trees and the understory negated the effect of thinning. Not all studies have shown an increase in leaf water potential. For example, Mitchell et al. (1983) found that thinning lodgepole pine stands did not reduce water stress.

As a result of increased water potential in some thinning studies, computer simulations suggest that photosynthesis may be up to 21% greater in thinned stands (Della-Bianca and Dils 1960). Other thinning studies have found evidence of increased photosynthetic capability, as measured by leaf gas exchange and increased foliar nitrogen (Feeney et al. 1998, Kolb et al. 1998).

In some cases, favourable physiological responses to thinning has resulted in increased radial growth. Average basal area increment has been found to increase significantly following thinning (Della-Bianca and Dils 1960, Helvey 1975, Feeney et al. 1998). Conversely, several studies have reported poor growth responses of conifers to thinning (Staebler 1956, Yerkes 1960, Harrington and Reukema 1983). Staebler (1956) termed the response 'thinning shock', stating that if the foliar area of the tree is small in comparison to the cambial area, the supply of carbohydrates may not be enough to supply the increased rate of respiration. Thinning shock appears to be related to thinning intensity, site quality, tree species, as well as vigor and age, and is most common at intense thinning levels on poor to medium quality sites (Harrington and Reukema 1983). There have been several cases of thinning shock reported for Douglas fir and other conifer species, although there appear to be no cited cases for ponderosa pine.

Thinning has also been shown to increase resistance to insect attacks. For ponderosa pine and Douglas fir, important mechanisms of resistance include the development of tough foliage to limit defoliation and resin production to resist bark beetles (Kolb et al. 1998). Increased resin flow in thinned (Kolb et al. 1998) and thinned and burned ponderosa pine stands (Feeney et al. 1998), combined with increased leaf toughness in thinned pine (Feeney et al. 1998, Kolb et al. 1998), collectively result in an increased ability to resist attack by forest insects.

2.4.2 The Restoration Process – Revegetation Using Bunchgrass Plugs

Fire suppression has an impact on understory plant performance and abundance, but prolonged absence of fire may result in the loss of certain species from the plant community (Parker and Kelly 1989). Increasing canopy cover results in shade-intolerant species suppression and perhaps, species dormancy. Fire frequency can have a severe impact on seed bank renewal. If the interval is too long, seed longevity may be exceeded (Parker and Kelly 1989), resulting in elimination of certain species.

Revegetation of arid and semiarid sites that have been disturbed or manipulated is recently receiving increased attention (Grantz et al. 1998a). However, restoration of degraded rangeland ecosystems has not, to date, received the same amount of attention as other ecosystem restoration projects (e.g. wetlands). As a result, there is little empirical information to guide revegetation efforts (Pyke and Archer 1991, Grantz et al. 1998a). The bulk of revegetation work for the purpose of ecosystem restoration, has been done on direct seeding (e.g. Pyke and Archer 1991, Richards et al. 1998). Unfortunately, direct seeding restoration projects usually fail due to lack of moisture required for successful germination, especially in arid (and semi-arid) rangelands (Grantz et al. 1998a). For example, natural impediments (e.g. weather) and anthropogenic disturbance reduce the success of restoration plantings in the low deserts of California to about one success in every 10 years (Grantz et al. 1998b).

Transplants for rangeland restoration have been increasingly used since the 1980's and several techniques have been evaluated for successful establishment (e.g. Bainbridge and Virginia 1990, Bainbridge et al. 1995). The majority of this work has looked at transplanting native shrubs. Transplanting of native grass plugs has largely been done to test hypotheses related to interspecific competition and not to ecosystem restoration applications (e.g. Wilson 1994, Gerry and Wilson 1995, Peltzer et al. 1998, Peltzer and Wilson 2001).

To address the gradual loss of bunchgrasses from NDT4 stands in the Rocky Mountain Trench, the Invermere Forest District initiated a native seeding program in 1994 for the purposed of range rehabilitation, road reclamation and ecosystem restoration (Invermere Forest District 2000). Success of the seeding trials was low, so it was proposed that the native seed (local to the area) be grown into 'plugs' under greenhouse conditions for subsequent planting into degraded sites. The Inveremere Forest District first started using plugs in 1997. Species chosen for transplant were Richardson's needlegrass and bluebunch wheatgrass.

2.4.2.1 Factors Affecting Revegetation Success

The key to restoring native plant communities is often in identifying and overcoming factors that impede or restrict ecosystem development (Gerry and Wilson 1995). Gerry and Wilson (1995) found the factors affecting performance of six different transplanted grassland species were initial size, species type and competition. Factors that could affect establishment of bunchgrass plugs in the Rocky Mountain Trench include choice of species, season of planting and interspecific competition.

2.4.2.1.1 Species

2.4.2.1.1.1 Bluebunch Wheatgrass (*Pseudoregneria spicata* (Pursh)A.Löve)

Bluebunch wheatgrass is a perennial bunchgrass that begins growth in early spring and becomes dormant in the summer (Willms et al. 1979). It grows on dry, open sites in the steppe and montane zones of British Columbia. Wheatgrass species often have creeping rhizomes (Gould and Shaw 1983). Bluebunch wheatgrass has short rhizomes that may allow it to dominate under moist conditions (McLean 1979). The relatively tall growth habit of bluebunch wheatgrass (up to 1.5m tall) makes this grass susceptible to grazing as their apical meristems reach a height where they can be readily removed by grazing (Branson 1956). This species is one of the most important native forage bunchgrasses in BC (Stewart and Hebda 2000).

2.4.2.1.1.2 Richardson's Needlegrass (Stipa richardsonii Link*)*

Needlegrasses are cespitose perennials widely distributed in temperate and tropical regions of the world (Gould and Shaw 1983). Richardson's needlegrass grows in low-elevation grasslands and openings in montane forests, most often with pines or Douglas fir (Stewart and Hebda 2000). Richardson's needlegrass occupies moister habitats than bluebunch wheatgrass (McLean 1979). *Stipa* species are considered to be relatively resistant to grazing (Branson 1956, Peterson 1962, Wright 1971, Scagel and Maze 1984), largely because of their prostrate growth form, maintenance of carbohydrate reserves, and slow spring growth, which allows 'escape' from heavy grazing, as well as rapid regrowth after clipping (Peterson 1962). Few grass species have a higher drought tolerance than *Stipa* species (Gurevitch 1986).

2.4.2.1.1.3 Species Transplant Success

Preliminary field studies completed by range ecologists in the Invermere Forest District show that survivorship when transplanting Richardson's needlegrass was 94% without grazing and 50% with grazing. All plugs were planted in the spring (May 21, 1997) at the same location. Survivorship of bluebunch wheatgrass plugs was considerably lower (3.6%). These treatments were not replicated spatially, so it was not possible to extrapolate these results to the rest of the Trench. Species identity appeared to be an important determinant of competitive response in a transplant experiment in Saskatchewan (Gerry and Wilson 1995). One factor that may contribute to the variation in competitive ability of different species is growth rate (Grime 1979). Parsons et al. (1971) found bluebunch wheatgrass required 51 days for the completion of reproductive development, while needle and thread grass (*Stipa comata* Trin&Rupr) required only 18 days. Rapid growth of needle and thread grass appeared to be related to an increase in temperature while bluebunch wheatgrass appeared to be unaffected by temperature (Parsons et al. 1971).

2.4.2.1.2 Competition

Competition for limited resources may determine the presence, absence, or abundance of species in a community and determine their spatial arrangement (Pyke and Archer 1991). In the NDT4 context, a primary concern relating to the establishment of grass plugs is pinegrass competition. As a result of conifer ingrowth and encroachment, pinegrass, a shade tolerant species, has expanded into previously treeless openings once dominated by shade-intolerant bunchgrasses (Steele and Geier-Hayes 1995). Pinegrass is a rhizomatous species that initiates growth early in the spring (McLean 1979). Due to its shallow rooting habit, pinegrass is a very effective competitor in NDT4 stands. Competition for water, a key limiting factor for growth in arid environments, may be important during restoration activities (e.g. thinning and fire) (Melgoza et al. 1990). Ross and Harper (1972) found that physiologically earlier developing species (e.g. pinegrass) continually increase their ability to capture resources at the expense of later developers, and in doing so, increase their physical zone of influence. Peterson (1988) noted that interference from pinegrass reduced the weight of foliage, stemwood and roots of ponderosa pine seedlings. Several studies have shown that when transplants are grown in the presence of neighbours, their growth and establishment is suppressed (e.g. Ross and Harper 1972, Wilson 1994, Gerry and Wilson 1995, Peltzer and Wilson 2001).

2.4.2.1.3 Season

Late-spring and early-summer plantings of grasses often fail, due to dry soil conditions and competition from annual grass and broadleaf weeds (Smart and Moser 1997). Therefore, time of growth initiation or planting can also determine success of transplantation efforts. Native bunchgrasses in BC are classified as summer quiescent, as they become semi-dormant during summer drought, and show significant growth as soil moisture increases in the fall (Hooper and Pitt 1998). Most experiments initiate transplant seedlings in the spring. Success of transplant experiments may be improved by adding fall season plantings, as many native bunchgrass initiate growth very early in the spring (Willms et al. 1979, Stout et al. 1981, Hooper and Pitt 1998). Planting bunchgrasses in the fall may maximize survival by capitalizing on available moisture at that time.

2.5 Conclusions

Conifer ingrowth and encroachment is assumed to be due to human induced fire suppression, although there are likely other contributing factors, such as, long-term climate patterns, widespread overgrazing and the reduction in numbers of native amereicans. Although the cause of conifer ingrowth and encroachment is not clear it is apparent that fire-maintained ecosystems have undergone dramatic changes in structure and species composition since the advent of European settlement. These changes are viewed as undesirable due to lack of forage production, loss of diversity, increase in insect outbreaks and an increased risk of fire hazard in these areas. Restoration activities, such as, thinning and bunchgrass plant transplants are designed to return fire-maintained plant communities to a more desirable historical state. Across North America, the plant community response to restoration have been highly varied, likely due to initial site differences and the large-scale of these operations. It is necessary to monitor these activities at a site level to ensure the goals of restoration are being met. It is especially important in BC, where the largest, provincial terrestrial restoration program is currently underway, with no formal monitoring program in place.

3. MONITORING MECHANICAL RESTORATION EFFECTIVENESS IN MONTANE FORESTS OF THE EAST KOOTENAY

3.1 Introduction

Prior to 1900 (pre-settlement) ponderosa pine (*Pinus ponderosa* Douglas ex Lawson & Lawson var. *ponderosa*) and interior Douglas fir [*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco] forests characterized montane forests of the inland northwest. These forests were maintained largely by fire and other disturbances (insect outbreaks etc.) (Everett et al. 2000). In areas where fire-resistant ponderosa pine and Douglas fir were seral, low intensity surface fires limited shade-tolerant competitors such as lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. *ex*. S. Wats.) and young interior Douglas fir from establishing in the understory (Arno et al. 1995). In summarizing the literature on historic fire regimes in ponderosa pine/Douglas-fir types in Western Montana, Arno et al. (1995) concluded that prior to 1900 most stands had recurring surface fires at intervals of 26-50 years at high elevation sites (1500-1800m), and approximately 13 years at low elevations (800m-1500m).

In the late 1800's, selective logging preferentially removed most large ponderosa pine and Douglas fir. The simultaneous introduction of fire suppression policies in the early 1900's allowed for enhanced understory conifer regeneration in fire-maintained forests across North America (Kaye et al. 1999). This in combination with the absence of widespread native American burning, and unregulated grazing early in the 20th century resulted in a marked change in forest stand structure from relatively open, older-aged forests to closed, young forests (Veblen et al. 2001).

These changes also occurred in British Columbia (BC). With the establishment of the Forest Service in 1912. Regulations concerning fire management on forested land were greatly expanded (Dorey 1979). Policy was instituted that called for the suppression of fires in all areas to prevent the loss of valuable timber (Parminter 1978). Active suppression increased coincided with an increase in the fire return interval by as much as 60 years, accelerating forest ingrowth and encroachment within dry forest zones (Bai 2000), including those within the East Kootenay Trench of southeastern BC. Dry forest zones that historically experienced frequent, low intensity fires (5 – 50 years) that limited encroachment by woody species, are classified as Natural Disturbance Type 4 (NDT4) systems (Province of British Columbia 1995). There are approximately 250 000 ha of NDT4 in the Rocky Mountain Trench of BC. Gayton (1997) estimated that nearly 1% or 3000 ha of open NDT4 forest are lost each year in the Trench to ingrowth and encroachment, an estimate similar to those made in other parts of the province (Bai et al. 2001).

Within ingrown forests of North America, changes in forest structure and the associated understory have received considerable attention in the past because of reductions in forage availability for livestock and wildlife (Pase 1958, Cooper 1960, Ffolliott and Clary 1982, Bojorquez et al. 1990). In addition to forage declines, shading caused by the invasion of conifer species has favoured the invasion of mesophytic shrubs and herbs into historically dry stands (Lunan and Habeck 1973), resulting in species composition changes. For example, pinegrass (*Calamagrostis rubescens* Buckl.), a rhizomatous perennial that remains abundant under shade, is prevalent under dense fir canopies (Steele and Geier-Hayes 1993). Lack of light and increased competition from pinegrass may limit the existence and distribution of more desirable plant species including native bunchgrasses [e.g., rough fescue (*Festuca campestris* Rydb.), Idaho fescue (*Festuca idahoensis* Elmer), bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), Richardson's needlegrass (*Stipa richardsonii* Link.), needle-and-thread grass (*Stipa comata* Trin.&Rupr.) and stiff needlegrass (*Stipa occidentalis* Thurb. ex S. Wats. var. *pubescens* Maze, Tayor and MacBryde]. The loss of bunchgrass communities is significant within the Rocky Mountain Trench, as native ungulates and livestock exhibit a high degree of preference for these species (Clark et al. 1998, Ross 2001). Given that ungulate numbers (including livestock) tend to remain relatively constant in an area over time, the gradual loss of bunchgrasses can lead to overgrazing of remaining vegetation, causing further undesirable changes in species composition (Gayton 1997).

As land administrations shift away from management at the forest stand or single species level and towards ecosystem management, the effect of conifer encroachment on understory species composition and diversity is receiving increased attention (Thomas et al. 1999). The goal of ecosystem management is the maintenance of ecosystem integrity, including species composition (Naumberg and DeWald 1999). Therefore, an understanding of overstory-understory relationships is increasingly guiding efforts to mitigate negative impacts caused by ingrowth and encroachment through the use of ecosystem restoration (Fiedler and Carlson 1992, Covington et al. 1997, Kaye et al. 1999, Ritchie and Harksen 1999).

Although dry forest restoration treatments have been used in several areas of North America, including Arizona, Colorado, Montana and British Columbia, most research has examined a narrow set of treatment effects. Furthermore, while prescribed burning has been studied extensively, less attention has been given to the ecological impacts of the thinning treatments necessary to restore a more natural stand structure prior to the re-introduction of fire (Smith and Arno 1999). Restoration of dense stands begins with selective thinning to remove excess understory and weak overstory trees that cannot be safely killed in a prescribed fire (Arno et al. 2000).

Every phase of restoration should be monitored in isolation to ensure the goals and objectives of ecosystem restoration are being met, as well as obtain information that will guide future restoration efforts (Ritchie and Harksen 1999). This research project was designed to monitor thinning treatments conducted by the BC Ministry of Forests as part of their ecosystem restoration operations. Prescriptions were based on land use guidelines set by the Kootenay Boundary Land Use Plan (KBLUP) (Province of British Columbia 1997). The plan stipulates that existing grassland and open forest must be maintained in the region, and ingrown NDT4 forests restored to historical open forest (Province of British Columbia 1998) (Table 3.1). In response to the KBLUP, the Cranbrook and Invermere Forest Districts initiated a large-scale NDT4 restoration program. The Rocky Mountain Trench Restoration Program is the largest, longest-running terrestrial restoration initiative underway in the province (Machmer et al. 2002). According to current projections, an estimated 135 000 ha of land will be converted to grassland or open forest by the year 2030 (Rocky Mountain Trench Ecosystem Restoration Steering Committee 2000). This research project monitored the first of a three-phase rotational block prescription designed to restore ingrown forests in the Rocky Mountain Trench. In the first phase, ingrown forest stands were thinned to 20% - 70% of the original basal area.

Pre-treatment (i.e. pre-thinning) overstory-understory relationships were initially examined using a retrospective assessment at 2 sites in the dry forest zone of the Rocky Mountain Trench of British Columbia. This was followed by an examination of the initial 2-year understory response to tree thinning. Specific objectives were to (1) quantify various overstory-understory synecological relationships within NDT4 forests, and (2) determine the initial effect of forest thinning on changes in understory species composition, diversity and production. This project was also designed to represent the first step of an ongoing monitoring program implemented to evaluate the ongoing success of restoring ingrown ponderosa pine and Douglas fir forests in southeastern BC.

The following null hypotheses were tested:

?? There is no significant pre-treatment relationship between forest overstory characteristics [canopy closure (understory light, timber volume (m^3/ha), or merchantable stem density (stems/ha)] and the understory plant community, including species richness and diversity, species cover, and forage production.

?? ? reduction of forest overstory will not significantly increase the density of important forage and browse species.

3.2 Methods

3.2.1 Study Area

This research was conducted in the southeastern corner of British Columbia in the Rocky Mountain Trench, within the Invermere Forest District (Fig. 3.1), an area more commonly known as the East Kootenay Valley. This region is strongly influenced by maritime polar air masses that are drier after being lifted over the Coast, Monashee, and Selkirk Mountains. The southern valley has an upland continental climate with well-defined seasons. Summers are characterized as warm and dry while winters are typically cold with deep valley inversions (Marsh 1986), which often causes warmer temperatures at low elevation sites (McLean and Holland 1957). Mean monthly air temperatures vary from –8.3? to 18.2?C (Table 3.2). Average annual precipitation is 384.5mm, with May and June being the wettest months (Table 3.2). There is an average of 147.9cm of snow during the winter months. This project was initiated in 1999 when precipitation and temperatures were near average. However, 2000 and 2001 were dry with approximately 45% and 35% of average precipitation during the growing season (May-September), respectively (Table 3.2).

Two blocks (i.e. treatment areas) were selected for this project, and included the Sheep Creek North Range Unit (RU5041) and the Wolf/Sheep Creek (also known as Premier Ridge) Range Unit (RU5015). Both sites were approximately 75km south of Invermere and 20km apart (49?58’N 115?43’W) (Fig. 3.1). These areas will hereafter be referred to as the Sheep Creek and Wolf Creek areas, respectively. Both Sheep Creek and Wolf Creek were highlighted in the Kootenay Boundary Land Use Plan (Province of British Columbia 1997) as “important ungulate winter range”. In addition, Wolf Creek is a zoned as a Special Resource Management Zone (SRMZ), largely due to a high concentration of regionally significant and sensitive resource values, including critical wildlife habitat (e.g. Hudson et al. 1976).

Current commercial uses of these areas include cattle grazing. The earliest recorded grazing at Sheep Creek was 1937 and at Wolf Creek, 1941. Large ponderosa pine and Douglas fir were likely selectively logged in these areas during the 1930’s, as many forest stands were ‘unofficially’ logged to support railway tie production (Phil Burke, Range Officer, Invermere Forest District, per comm., 2002). Decomposing, large Douglas-fir and ponderosa pine stumps were observed in both stands.

Both blocks were situated within NDT4 forests. The Sheep Creek block is situated in the IDFdm2 (Kootenay dry mild interior Douglas-fir variant) vegetation zone, while Wolf Creek is located in the PPdh2 (Kootenay dry hot ponderosa pine variant) vegetation zone (Braumandl and Curran 1992). Zonal PPdh2 sites have open stands of Douglas fir and ponderosa pine with an understory of predominantly bluebunch wheatgrass. Zonal IDFdm2 sites have climax stands of Douglas fir with an understory dominated by pinegrass and shrubs such as birch-leaved spiraea (*Spiraea betulifolia* Pall. ssp. *lucida* (Dougl. ex Greene) Taylor & MacBryde), common juniper (*Juniperus communis* L.), soopolallie (*Shepherdia canadensis* (L.) Nutt.), Saskatoon (*Amelanchier alnifolia* Nutt.), and common snowberry (*Symphoricarpos albus* (L.) Blake) (Braumandl and Curran 1992). Site specific characteristics for each block are contained in Appendix 1.

Soils at Sheep and Wolf Creek are classified as Orthic Eutric Brunisols (Lacelle 1990). Eutric Brunisols are characterized as strongly calcareous and low in organic matter (National Research Council of Canada 1998). The dominant soil association at both sites is Fishertown, a gravelly, sandy loam derived from fluvioglacial parent material (Lacelle 1990). The soil is rapidly drained and located on moderately to strongly rolling sites (Lacelle 1990). A less common soil association found at both sites is the Wycliffe association, consisting of Brunisolic soils derived from morainal parent material containing limestone (Lacelle 1990). There are minor occurrences of the Elko soil association at Wolf Creek, an Orthic Eutric Brunisol on glaciofluvial parent material. These soils are not as gravelly as the Wycliffe and Fishertown associations, but are still relatively well-drained (Lacelle 1990).

3.2.2 Experimental Design

An identical experimental design was used at each block. To facilitate objective and representative data collection across blocks, sampling was superimposed on existing timber cruise plots, from which comprehensive overstory information had previously been collected by Invermere Forest District staff. An added benefit of this approach was that cruise plots were systematically distributed (on a 100m*100m grid) throughout each block and were therefore representative of a wide range of initial forest structure conditions. Prior to sampling, however, plots were stratified by biogeoclimatic zone using methods outlined in Braumandl and Curran (1992) and slope. All timber cruise plots identified as being in the IDFdm2 and PPdh2 and having a slope less than 5% were selected for subsequent monitoring. Slopes greater than 5%

were excluded to remove strong moisture gradients as a confounding factor in the analysis. Final plot numbers in the Sheep Creek and Wolf Creek blocks were 15 and 18, respectively.

Within each sampled timber cruise plot, three parallel 10 m transects were established oriented south to north. The 2 outer transects were equidistant off the centre of the plot (4m in either direction), while the middle transect intersected the plot centre (Fig. 3.2). Transect ends and plot centres were permanently marked with rebar pins to facilitate relocation, and all plots located using a GPS. Sampling occurred for three consecutive years beginning in 1999, the year thinning activities were initiated.

Pre-thinning sampling for understory herb and shrub cover, as well as understory light and duff were completed in 1999 at both bocks. However, forage production data were collected in 1999 at Wolf Creek only because Sheep Creek was thinned later that summer (i.e. June 1999). Thinning of Wolf Creek occurred a year later during June-July 2000. Timber cruise data were made available by the Invermere Forest District and summarized by plot.

In 2000, all first-year post-thinning sampling was completed at Sheep Creek. At Wolf Creek, only forage production was sampled in 2000 due to the timing of harvest relative to plant growth. A year later, comprehensive second-year post-harvest sampling was completed at both blocks in 2001. All thinning treatments were consistent with the KBLUP (Province of British Columbia 1998) (Table 3.1) and were intended to promote winter forage availability and create the open forest habitat required for many threatened or endangered species (e.g., badger, Lewis woodpecker, and sharp-tailed grouse).

3.2.3 Vegetation Sampling

Vegetation sampling was modified from the methods provided in the document, 'Monitoring Restoration of Fire-Maintained Ecosystems in the Invermere Forest District' (Powell et al. 1998). Within each plot, the 2 outer 10 m transects were sampled for vegetation cover by species (Daubenmire 1959) and key species density. Percent canopy cover by plant species was estimated in 0.1m^2 quadrats positioned every metre (n=20). In addition, the density of key native bunchgrasses (rough fescue, Idaho fescue, bluebunch wheatgrass, Richardson's needlegrass, needle-and-thread grass and stiff needlegrass) were counted in 10m^2 (1m * 10m) belted transects established along the interior of each outside transect. Density was averaged across the 2 transects (x/m^2). Density counts were restricted to those bunchgrass species historically common within NDT4 plant communities, and were also considered important forage species for wildlife.

Species richness was determined by counting the total number of species found in all quadrats (x/2m^2). Species diversity was determined using the Shannon-Weiner diversity index ($H=-P_i\log[P_I]$) (Bonham 1983).

Ocular estimates of shrub canopy cover (Daubenmire 1959) were obtained within 20, 2m^2 (1m * 2m) quadrats nested overtop the 0.1 m^2 quadrats at each metre mark of the outside transects. Shrub quadrats were contiguous along the outer transects, oriented with the narrow side on the transect (Fig. 3.2). The density of key shrubs and tree saplings (<1.5m tall) was assessed in 2, 20m^2 (2m * 10m) belted transects, each centred on the outside line transects. Key shrubs included common browse species [Saskatoon, antelope-brush (*Purshia tridentata*

(Pursh) DC.)] and woody species considered to be 'encroaching' (e.g. Douglas fir and lodgepole pine). Density was averaged across the 2 transects (x/m^2). Depth of the Ah horizon was also assessed at each plot to assess the effect of conifer ingrowth and encroachment on soil organic matter.

Forage production was quantified within 4, 0.5 m^2 quadrats (0.5m*1m) systematically located on the centre transect. Quadrats were located, *a-priori*, at different locations in each of the 3 years of the study (Fig. 3.2) to avoid confounding effects of sampling during subsequent years. Current annual production in all quadrats was clipped to ground level in early September after peak growth was reached. All samples were sorted by descriptive group for analysis and included bunchgrasses, pinegrass, other grass, sedges (*Carex* spp.), forbs and shrubs. Kinnikinnick (*Arctostaphylos uva-ursi* (L.) Spreng.) was not clipped as it was not a species of direct interest for monitoring habitat changes, as it is not a desirable forage species for domestic and wild ungulates. Descriptive groups were assessed instead of individual species due to suspected difficulty in detecting statistically significant changes at the species level. To assess the practical importance of ungulate (wild and domestic) herbivory in each block, two $(1.5m)^2$ range cages were randomly situated in each of 5 randomly selected plots per block. In each cage, vegetation was sampled the same as the adjacent production quadrats within the plot, with caged-uncaged comparisons (i.e. the paired-plot method) used to quantify the level of herbivory (Bonham 1983). All vegetation samples were stored in a paper bag and air-dried, and subsequently oven-dried at 60 ?C to constant mass and weighed.

Understory light is a direct measurement of the overstory influence on understory growing conditions. The amount of diffuse non-interceptence light (or understory light) was measured using a LI-COR®LAI-2000 Plant Canopy Analyzer (Welles and Norman 1991). This value is the ratio of diffuse light measured at the top of each 0.1 m^2 Daubenmire frame (i.e. 30cm high) used for plant canopy cover measurement, as a proportion of the diffuse light measurement simultaneously taken from a vantage point with an unobstructed sky view. Light measurements could only be taken on days when the sky was uniformly overcast. BC Ministry of Forests staff took all light measurements.

Tree volume (m3/ha), and tree stem density (stems/ha) pre-thinning values were obtained from Ministry of Forests timber cruise data. Standard cruising methodology was used using variable plot methods (Province of British Columbia 2002). Variable plot sampling , also known as prism sampling, was used since the probability of tree selection was proportional to basal area, therefore, the large diameter trees are sampled with the same intensity as the small diameter trees (Province of British Columbia 2002). Post-thinning measurements were taken by Ministry of Forests Staff. Post-thinning values were obtained by subtracting the basal area and volume of trees present in the pre-thinning plots that were not present in the post-thinning plots. The problem with this method is that not all trees were accounted in the pre-thinning variable plot sampling (e.g. some smaller trees may be excluded from the variable plot), therefore it is not a precise measurement of the amount of material removed. Post-thinning measurements can be viewed more as a measure of the intensity of thinning that occurred at each plot.

3.2.4 Statistical Analyses

In the investigation of pre-thinning plant synecological relationships, treatments were considered to be the varying levels of forest ingrowth (overstory characteristics) among plots. Differences between the 2 blocks were initially assessed using an analysis of variance (ANOVA) for a completely random design with subsampling. Preliminary analysis indicated there were no significant block by treatment effects ($p>0.1$). However, between plot variance was also high within each block (Table 3.3) and may have prevented the identification of significant interactions. Additionally, most pre-thinning descriptive group canopy cover values were significantly different ($p<0.05$) between blocks, as was understory light and timber volume ($p<0.05$) (Table 3.3). Due to the noted abiotic and biotic differences between blocks, they were examined separately in all subsequent analysis.

Pre-thinning relationships of understory light, merchantable stem density and overstory tree volume with the understory characteristics were examined using regression techniques (Steel et al. 1997). Within a block, treatment averages of each response variable were calculated for each plot and regressed against the independent variables, with each plot forming one point in the regression. All regressions were checked for non-linear relationships, there were none found. It is possible that non-linear relationships would be detected at higher sample sizes.

All data were checked for normality prior to analysis. Non-normal data were transformed using a square root (tree volume, tree density, and pinegrass, bunchgrass shrub, sedge, and forb production, as well as bunchgrass, sedge and bryophyte canopy cover) or a log+1 (Saskatoon canopy cover and density) transformation. Where transformations were necessary, negative values were made positive by adding the lowest value in the data set to each observation. Additionally, data were always uniformly transformed within a response variable across both blocks and years. All differences were considered significant at $p<0.10$, unless indicated otherwise.

To evaluate changes in vegetation following thinning, understory canopy cover and density values within a descriptive group in the pre-thinning year were subtracted from values in years 1 and 2 post-thinning (i.e. 2000-1999, 2001-2000 and 2001-1999). Changes in independent variables by plot were also quantified during the same time periods. Responses to thinning were then determined by regressing the change in independent variables against the change in canopy cover, density and production of each descriptive group.

3.3 Results

3.3.1 Pre-Thinning Relationships

Understory light at Sheep Creek displayed a significant ($p<0.10$) positive association to 3 understory variables (Table 3.4) including Saskatoon canopy cover (Fig. 3.3) and density, as well as total live herb canopy cover (Fig 3.4). Several other variables also approached significance ($p<0.20$) (Table 3.4), displaying weak positive relationships. Only spiraea canopy cover was significantly ($p<0.05$) associated with overstory tree density, also displaying a positive relationship.

At Wolf Creek, understory light was positively ($p<0.10$) related to 9 understory variables (Table 3.4), including species diversity (Fig. 3.5) and richness along with shrub, forb, sedge, and bunchgrass canopy cover (Fig 3.6), but was negatively ($p<0.01$) related to sedge production (Table 3.4). Similar to the other block, Saskatoon canopy cover (Fig. 3.3) and density were positively related to light at Wolf Creek.

At both blocks, depth of the Ah layer had too little variance between plots to analyze (1-2cm), although it was observed that the thicker Ah layers were found under relatively open canopies.

3.3.2 Overstory Changes With Thinning

At Sheep Creek, thinning removed an average of 68m^3/ha of timber, leaving 59m^3/ha. Tree stem density decreased by 261, leaving 243 stems/ha. In contrast, understory light increased ($p<0.001$) an average of 27% across all plots following thinning.

Thinning treatments at Wolf Creek removed an average of 48 m^3/ha, leaving 27 m^3/ha. Stem density decreased by 513, leaving 192 stems/ha. Understory light subsequently increased ($p<0.001$) by 30% following thinning.

3.3.3 Post-Thinning Relationships

3.3.3.1 Sheep Creek

Over the two years of the study, the overall canopy cover of pinegrass, birch-leaved spiraea, total shrubs and bryophytes all declined significantly ($p<0.05$) at Sheep Creek (Table 3.5). The only understory characteristic that increased ($p<0.10$) over that period was bunchgrass density (Table 3.5). Note that changes in production could not be assessed due to the lack of data from 1999 prior to thinning.

Following thinning, there were several positive responses in the understory of plots at Sheep Creek that were associated with overstory changes (Table 3.6). A total of 8 understory variables responded positively ($p<0.10$) at increased levels of thinning, 8 between 2000-2001 and 1 between 1999-2001 (Table 3.6). In general, positive changes in the understory were more closely associated to changes in overstory tree density (5 relationships) rather than timber volume (2 relationships) or understory light (1 relationship) (Table 3.6). Understory characteristics demonstrating positive responses included species richness (Fig. 3.7) and diversity, as well as forb and bryophyte canopy cover.

There were also 3 negative ($p<0.10$) associated with increased thinning intensity within the understory (Table 3.6). Changes in Saskatoon density varied inversely with greater reductions in tree density and timber volume in the first year after harvest (Table 3.6). Notably, bunchgrass density demonstrated a negative response with understory light increases following thinning over the period from 1999 to 2001 (Table 3.6, Fig. 3.8). This was despite an overall increase in bunchgrass density during the study (Table 3.5).

3.3.3.2 Wolf Creek

Thinning treatments at Wolf Creek were not completed until June 2000. As a result, changes recorded in August 2001 were equivalent to 1 full year of recovery. Thinning reduced ($p<0.10$) the canopy cover of pinegrass, sedge, and total live canopy cover, as well as bunchgrass and forb production (Table 3.5).

There were several negative associations found between the amount of overstory thinning and the understory at Wolf Creek (Table 3.7). Of the 10 significant ($p<0.10$) relationships found, 9 indicated the understory responded negatively at greater levels of thinning. Of these, 8 occurred between 1999-2001 while only 1 occurred between 2000-2001 (Table 3.7). Although forage production was negatively affected between 2000 and 2001 (Fig. 3.10), this same relationship did not materialize over the longer three year period from 1999 to 2001. Other negatively affected characteristics included species richness (Fig. 3.9) and the canopy cover of pinegrass (Fig. 3.11), bryophytes, and total live canopy cover, which were all associated with changes in both understory light and timber volume (Table 3.7). The single positive understory response was in shrub production (Table 3.7).

3.4 Discussion

3.4.1. Pre-Thinning Overstory-Understory Relationships

Initial differences in understory plant communities between blocks were likely the result of varied overstory and/or ecosite conditions. Wolf Creek is located in the PPdh2 biogeoclimatic zone and Sheep Creek in the IDFdm2. Ponderosa pine sites are historically more open, drier and thus, better suited to support shade intolerant bunchgrass communities. In contrast, Douglas fir sites are relatively closed, moister and therefore capable of supporting more shrubs adapted to mesic conditions, such as, Saskatoon and birch-leaved spiraea. Given that the outcome of thinning will depend on initial plant community structure and composition (Thomas et al. 1999), differences between the 2 sites, as determined by landscape-scale variation, likely accounts for much of the differential responses to thinning.

Among the overstory variables examined in the pre-thinning data, understory light levels clearly was most closely associated with understory plant characteristics at both blocks. Tree density was associated with only a single understory characteristic, while basal area had no significant association. Studies documenting overstory effects on the understory have found that understory light is significantly associated with species presence and abundance as measured by density and canopy cover (Lieffers and Stadt 1993, Naumberg and DeWald 1999), while overstory tree characteristics (e.g. stem density , stand volume) are more closely associated with understory biomass (Riegel et al. 1995, Naumberg and DeWald 1999). Neither tree density nor timber volume are thought to relate well to the spatial distribution of trees (i.e. clumped vs. uniformly dispersed) in a stand or the influence of crown cover on the understory (DeMaere, Range Research Technician, BC Ministry of Forests, per. comm. 2002), which may explain why understory light is generally the best predictor of pre-thinning understory plant characteristics, particularly species canopy cover and presence. Furthermore, light measurements are a direct indication of the above-ground competitive influences within ingrown

NDT4 stands, as it is the only measurement that directly accounts for the overstory influence of trees and shrubs because it is taken 30cm off the ground.

The adverse impact of declining understory light levels is apparent at both blocks but is better expressed at Wolf Creek, possibly due to the larger sample size at this site. Declining diversity (Fig. 3.5) and species richness at low light levels has implications for the health of a plant community. Less diverse plant communities are less 'resilient' and less likely to recover from disturbances such as grazing or fire (Schulz and Mooney 1993, Tilman and Downing 1994, Naumberg and DeWald 1999). This finding is consistent with other studies completed in North American fire-maintained ecosystems (Covington et al. 1997, Uresk and Severson 1998).

Additionally, results at Wolf and Sheep Creek reflect the association between the bunchgrass and the palatable shrub community (e.g., Saskatoon) to increased tree crown cover or decreased light levels (Fig. 3.3 and 3.6). As light declines over time the more productive and light demanding species disappear and create room for the establishment and growth of other species better suited to the changing conditions (Knowles et al. 1999). There was no significant relationship, positive or negative, found between pinegrass canopy cover and light reinforcing the notion that this species is tolerant of the loss of light (Lunan and Habeck 1973, Steele and Geier-Hayes 1993). Furthermore, birch-leaved spiraea cover was positively associated with increasing tree density (Table 3.4). Replacement of desirable forage species with less palatable species may have implications for grazing management at both locations.

The general lack of significant pre-thinning relationships between the overstory and forage production at Wolf Creek is contrary to several studies that documented a strong negative relationship between forage production and crown closure (Pase 1958, Cooper 1960, Moir 1966, Ffolliott and Clary 1982, Borjoquez et al. 1989, Knowles et al. 1999). The only significant relationship found was a negative association between sedge production and light (Table 3.4), which was somewhat unusual as there was a positive relationship between sedge canopy cover and light (Table 3.4). These seemingly contradictory patterns may be attributed to a greater number of sedge plants at increased light levels, the size and biomass of which may be limited by intense competition from other graminoids under these conditions, leading to lower production. In any case, sedge production contributed relatively little to total production (7%), and thus, has limited implications for ungulate management. Pre-thinning forage production versus light relationships may also need to be examined in more detail due to the small sample sizes employed here and the fact that only one year of pre-thinning data was collected at the Wolf Creek site only.

3.4.2. Post-Thinning Overstory-Understory Relationships

The thinning treatments resulted in significant negative changes in the understory at both locations. This finding was likely due, at least in part, to the disturbance associated with thinning (i.e. selective logging) itself and its direct impact on the understory such as the destruction of Saskatoon shrubs at Sheep Creek (Table 3.6). The greater number of trees removed at Wolf Creek, and the fact that thinning occurred during the growing season (i.e. July), also suggests the overwhelmingly negative understory responses at this location are due to physical disturbance. Results of this research indicate that at low thinning intensities, physical site disturbance may outweigh any benefits to plants provided by increased resources such as light, particularly in the short-term. This has been found in other thinning projects as well (Thomas et al. 1999, Thysell and Carey 2001). The observed absence of an increase in herb canopy cover (Table 3.5) is consistent with other studies that have found plant canopy cover by life form failed to increase 2 years post-thinning (Riegel et al. 1995, Ross 2001). Ross (2001) observed that forage production in the East Kootenay did not increase significantly until 2 years after restoration treatments were initiated.

The results found here should also be tempered by the unusually dry conditions of 2000 and 2001 (Table 3.2), which may be one of the leading cause in the reduction of plant canopy cover and production after thinning. Severe drought accompanying thinning would have limited the potential for plant regrowth, and may have further amplified the impact of mechanical disturbance associated with logging**.** In addition, significant grazing effects ($p<0.10$), particularly at Sheep Creek (Table 3.8), may have placed further stress on the understory, leading to poor plant community responses over a 1 or 2 year period. The return of average precipitation, coupled with continued recovery of vegetation, will likely result in greater recovery of the herbaceous understory. Monitoring the duration it takes a plant community to positively respond is important as the ability of the understory to recover from mechanical operations is critical for maintaining a stable forage supply for wild ungulates and livestock (Riegel et al. 1992), as well as preventing the over utilization of NDT4 rangelands.

Despite severe summer drought, the presence of grazing and mechanical disturbance, bunchgrass density did increase at Sheep Creek over the 3 years of monitoring. This change coincided with a general reduction in species such as pinegrass. Relative to other grasses, bunchgrasses are less limited by water and are adapted to low-nutrient environments (Herron et al. 2001), which likely gives these species a competitive advantage at the sites examined, especially shallow-rooted pinegrass. It should also be noted, however, that the observed increase in bunchgrass density may not be due to actual recruitment over the limited time period examined here. Rather, the change could be due to an increase in the size of heavily suppressed (and thus, undetected) bunchgrasses during pre-thinning sampling. Regardless of the mechanism, the observed increase suggests that this key understory component is poised to recover following the return of average growing conditions. Although bunchgrass density generally increased, there was a negative relationship between the change in bunchgrass density and light transmittance (Table 3.6, Fig. 3.8), likely an artifact of their susceptibility to mechanical disturbance, similar to that of other species.

Despite the short-term negative effects of thinning, it appears that in the second year of recovery, more positive changes were evident at greater thinning intensities within the plant communities examined at Sheep Creek (Table 3.6). These results are consistent with other studies that have found higher levels of plant canopy cover and species richness at greater levels of thinning (Uresk and Severson 1998, Thomas et al. 1999). This same trend did not materialize at Wolf Creek (Table 3.7), in part due to the shorter elapsed response time of that block following thinning (i.e. only 1 year of recovery).

Unlike the pre-thinning relationships, positive changes in the understory at Sheep Creek were often related to stem density and stand volume rather than light (Table 3.6). This trend suggests the plant communities examined may be associated with changes in belowground resource availability such as nutrients and water rather than light itself. Studies have shown that thinning increases soil moisture (Della-Bianca and Dils 1960, Riegel et al. 1992, Feeney et al. 1998, Kaye and Hart 1998b). In Northeastern Oregon, Riegel et al. (1992) reported that increased soil water (in response to thinning) added two months to the growing season, leading to significantly greater understory biomass. Studies documenting the effects of thinning on available plant nutrients have also found increases in mineralizable nitrogen (Riegel et al. 1992, Kaye and Hart 1998a).

Negative relationships between key shrub density (Table 3.6) and overstory thinning intensity at Sheep Creek were likely due to the mechanical disturbance caused by thinning operations. Shrubs maintain aboveground biomass and are therefore susceptible to damage and may account for other studies indicating that shrub production (i.e. current annual growth) does not respond significantly to thinning (McConnell and Smith 1965, Riegel et al. 1992, Thomas et al. 1999). This finding, however, contrasts with the positive change in shrub production observed in relation to changes in tree volume at Wolf Creek (Table 3.7), and is somewhat surprising given the high level of disturbance at the latter block (512 stems/ha removed). A potential explanation involves the initial abundance of shrubs at each location, as Wolf Creek had a less extensive shrub community compared to Sheep Creek (Table 3.3). This would both reduce the potential for a negative impact from the physical disturbance of thinning at Wolf

Creek, as well as minimize competition among recovering shrubs in the post-thinned environment, facilitating their growth. This is reinforced by the observation that shrub cover was not significantly impacted at Wolf Creek by thinning as is was at Sheep Creek (Table 3.5).

A consideration when using thinning as a restoration tool is that opening of the overstory may favor early-successional species and possibly, exotic species (Thomas et al. 1999, Thysell and Carey 2001). Early germination, rapid growth, and allocation of resources to aboveground biomass enable weeds to preempt resource use by their competitors (Sheley et al. 1993, Herron et al. 2001). Although there were no noxious weeds found in the pre-thinning plant communities, 2 plots had Canada thistle (*Cirsium arvense* (L.) Scop. var. *horridium* Wimm. & Grab.) at the Sheep Creek site in 2001, a noxious weed in the East Kootenay (B.C. MoF/MoAFF Noxious/Nuisance Weed List). Occurrences were located on highly compacted soils, and did not appear to be restricting the range or growth of native species. Thysell and Carey (2001) observed a 280% initial increase in exotic species but recorded a decline in the first year post-thinning to the third. The initial increase in exotic species may be temporary as weed species may have 'transient occupancy' (Thysell and Carey 2001) at Sheep Creek.

3.5 Conclusions and Recommendations

It is apparent that increased tree ingrowth within the NDT4 stands investigated is associated with negative changes in the understory plant community (cover, production and diversity) at both Sheep and Wolf Creek, indicating that restoration activities were warranted.

Although initial short-term changes in the thinned communities examined here appear to be negative, this seems to be largely due to mechanical disturbance and drought, potentially compounded by grazing. Greater thinning intensities were associated with larger positive changes in the understory in species richness and diversity. There was also some evidence for the recovery of the bunchgrass community. However, these results were not consistent across both study locations. It is therefore evident that individual blocks slated for restoration will need to be monitored in the future to see if recovery occurs and/or continues in each. Additionally, blocks should be monitored for the longevity of damage due to thinning activities (e.g. weed invasion, soil compaction leading to poor herb re-establishment or forage production).

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Table 3.1 Restoration targets[1] for various habitat components on crown land in the East Kootenay Trench of BC at the end of 30 years (2030).

Habitat component	**Current distribution (% of Trench)**	**Final distribution target (% and ha of Trench)**	**Tree density target (stems/ha)**
Shrubland	5 %	5 % (12,500 ha)	0
Grassland	10 %	23 % (57,500 ha)	? 75
Open forest	Open & managed forest is 85 % combined	31 % (77,500 ha)	76 – 400
Managed forest		41 % (102,500 ha)	400 – 5,000

1 Targets are achieved within the Crown NDT4 land base at the forest district level (Machmer et al. 2001).

Table 3.2 Temperature and precipitation at Johnson Lake weather station, 1999-2001[1] compared to the long-term averages recorded at the Cranbrook[2] airport (1968 - 1990).

	1999		**2000**		**2001**		**Long-Term Avg.**	
	Temp. (?C)	Ppt. (mm)	Temp. (?C)	Ppt. (mm)	Temp. (?C)	Ppt. (mm)	Temp. (?C)	Ppt. (mm)
May	10.1	22.6	10.6	22	12.8	7.7	11.2	43.6
June	14	50.1	14.8	16.9	14.3	29.2	14.9	50.5
July	16.5	44.1	19.5	13.8	19.4	16	18.3	31.6
Aug	18.4	47.5	18.2	12.1	20.6	4.2	17.8	34.4
Sept	12	6.5	11	21.9	14.1	13.8	12.4	32.6

1 49?55'N 115?44'W, Elev.-853m.

2 49?37'N 115?47'W, Elev.-939m.

Table 3.3 Comparison of overstory and understory (i.e. canopy cover) characteristics between the Sheep and Wolf Creek blocks, as sampled prior to thinning (1999).

		Sheep Creek (IDF)		**Wolf Creek (PP)**		
Strata	**Variable**	**Mean**	**StDev**	**Mean**	**StDev**	**p-value**
Understory						
	Bunchgrass canopy cover[1,2] (%)	1.62	1.62	7.22	1.42	**<0.001**
	Pinegrass canopy cover (%)	9.93	5.57	16.67	11.61	**0.05**
	Shrub canopy cover (%)	14.69	5.12	7.07	3.04	**<0.001**
	Carex canopy cover[2] (%)	0.61	0.18	4.9	5.06	**<0.001**
	Forb canopy cover (%)	8.38	5.75	8.72	5.34	0.86
Overstory						
	Volume (m^3/ha)[2]	126.77	63.5	75.25	44.39	**0.008**
	Density (stems/ha)[2]	503.62	367.36	705.28	457.50	0.23
	Understory light (%)	27.3	7	33.5	10	**0.05**

[1]Native bunchgrasses considered historically common as listed on pp.51.
[2]p-values are reported based on analysis using transformed data. Means and standard deviations of original data are presented.

Table 3.4 Summary of pre-thinning regressions of the understory variables on understory light and overstory tree density. Only regressions with p<0.20 are reported.

Block	Independent variable	Dependent variable	r^2 value	Root MSE	P-value	Regression equation
Sheep Creek (n=15) [4]						
	Light (% of full)	Species richness (x/2m^2)	0.18	5.65	0.11	y=8.42+34.80x
		Species diversity	0.18	0.17	0.13	y=0.57+0.97x
		Bunchgrass density[1,2] (x/10m^2)	0.16	1.13	0.13	y=0.26+6.48x
		Shrub canopy cover(%)	0.14	5.02	0.17	y=7.95+0.31x
		Saskatoon canopy cover(%)[2]	0.22	0.31	**0.08**	y=-0.16+2.41x
		Saskatoon density (x/20m^2)[2]	0.30	0.45	**0.03**	y=-0.14+3.88x
		Total herb canopy cover (%)	0.32	12.37	**0.03**	y=1.99+110.61x
	Tree density (stems/ha)	Spiraea canopy cover (%)	0.30	5.48	**0.04**	y=-3.49+0.48x
Wolf Creek (n=18)						
	Light (% of full)	Species richness (x/80m^2)	0.18	6.01	**0.07**	y=10.81+25.92x
		Species diversity	0.26	0.19	**0.03**	y=0.47+0.97x
		Bunchgrass canopy cover (%)[1,2]	0.29	8.9	**0.02**	y=6.68+0.11x
		Forb canopy cover (%)	0.67	4.80	**<0.001**	y=-4.39+39.93x
		Total canopy cover (%)	0.44	17.65	**0.002**	y=17.76+144.71x
		Shrub canopy cover (%)	0.20	2.96	**0.06**	y=2.37+13.60x
		Saskatoon canopy cover (%)[2]	0.33	0.21	**0.01**	y=0.20+1.39x
		Saskatoon density (x/20m^2)[2]	0.49	0.27	**0.02**	y=0.74+2.33x
		Sedge canopy cover (%)[2]	0.35	0.92	**0.008**	y=-0.14+6.25x
		Sedge production (kg/ha)[2,3]	0.63	0.31	**0.006**	y=1.95-0.5x

[1] Native bunchgrasses considered historically common as listed on pp.51.

[2] p-values are reported based on analysis using transformed data. Means and standard deviations of original data are presented.

[3] These data only use 10 plots as production data was not collected at all 18 plots in 1999.

[4] No production available from Sheep Creek in 1999.

Table 3.5 Understory variables undergoing significant (p<0.1) changes during 3 consecutive years of sampling from 1999-2001 at each block.

Block	Time	Response Variable	? mean	SD? x	Pr<F
Sheep	2000 – 2001	? Pinegrass canopy cover (%)[1]	-3.15	3.66	0.03
	1999 – 2001	? Bunchgrass density (x/$10m^2$) [1]	6.1	7.30	0.07
		? Spiraea canopy cover (%)	-4.03	1.09	0.02
		? Shrub canopy cover (%)[1]	-6.37	6.25	0.002
		? Bryophyte canopy cover (%)[1]	-6.75	8.31	0.02
Wolf	1999 – 2001	? Pinegrass canopy cover (%)	-7.36	8.24	0.03
		? Sedge canopy cover (%)	-2.46	3.61	0.09
		? Total herb canopy cover (%)	-28.68	17.41	<0.001
		? Bunchgrass production (kg/ha)[1,2]	-21.7	36.4	0.03
		? Forb Production (kg/ha) [1,2]	-25.2	33.9	0.03

[1] p-values are reported based on analysis using transformed data. Means and standard deviations of original data are reported.

[2] These data only use 10 plots as production data were not collected at all 18 plots in 1999.

Table 3.6 Relationship of changes in the tree overstory following thinning to subsequent understory changes from 1999-2001 at Sheep Creek (Interior Douglas Fir zone). Only regressions with p<0.1 are reported (n=15).

Time Period	Independent Variable	Dependent Variable	r^2 Value	Root MSE	Pr<F	Regression Equation
1999-2000						
	? Tree density (stems/ha)	? Saskatoon density (x/20m^2)[1]	0.43	3.45	0.07	y=3.78-0.27x
	? Volume (m^3/ha)	? Saskatoon density (x/20m^2)[1]	0.36	3.82	0.07	y=3.39-0.50x
2000-2001						
	? Tree density (stems/ha)	? Species diversity	0.25	0.11	0.07	y=-0.08+0.01x
		? Species richness (x/80m^2)	0.43	2.84	0.01	y=-5.59+0.25x
		? Total cover (%)	0.33	9.44	0.03	y=-14.36+0.68x
		? Forb cover (%)	0.22	4.51	0.09	y=-4.87+0.25x
		? Bryophyte cover (%)[1]	0.29	4.40	0.05	y=-6.43+0.29x
	? Volume (m^3/ha)	? Species richness (#spp/80m^2)	0.44	2.72	0.01	y=-5.25+0.49x
		? Bryophyte cover (%)[1]	0.42	3.62	0.02	y=-7.52+0.61x
	? Light (%)	? Bryophyte cover (%)[1]	0.27	5.29	0.05	y=-5.72+25.1x
1999-2001						
	? Light (%)	? Bunchgrass density (x/10m^2)[1]	0.22	0.68	0.08	y=5.09-2.83x

[1] p-values are reported based on analysis using transformed data. Means and standard deviations of original data are reported.

Table 3.7 Relationship of changes in the tree overstory following thinning to subsequent understory changes at Wolf Creek from 1999-2001 (Ponderosa Pine zone). Only regressions with p<0.1 are reported (n=18).

Time Period	Independent Variable	Dependent Variable	r^2 Value	Root MSE	Pr<F	Regression Equation
1999-2000						
	? Volume (m^3/ha)	? Shrub Production (kg/ha)	0.45	0.50	0.07	y=-0.14+0.12x
2000-2001						
	? Light (%)	? Total Production (kg/ha)	0.22	3.72	0.06	y=3.08-11.3x
1999-2001						
	? Light (%)	? Species Richness (#spp/80m^2)	0.17	4.29	0.09	y=2.15-10.5x
		? Total Cover (%)	0.16	16.61	0.1	y=39.7-15.6x
		? Pinegrass Cover (%)	0.30	6.92	0.02	y=0.33-24.9x
		? Bryophyte Cover	0.25	1.54	0.05	y=1.14-3.87x
	? Tree Density (stems/ha)	? Pinegrass Cover (%)	0.30	7.10	0.02	y=-0.92-0.34x
		? Bryophyte Cover	0.20	1.38	0.1	y=0.93-0.05x
	? Volume (m^3/ha)	? Pinegrass Cover (%)	0.24	7.40	0.04	y=-1.17-1.08x
		? Bryophyte Cover	0.20	2.34	0.09	y=0.95-0.17x

Table 3.8 Comparison of forage removed for all significant (p<0.10) variables, as determined by caged and uncaged production (kg/ha) data at both blocks in 2000 and 2001.

			Caged		Uncaged		Pr<F
Location	**Time Period**	**Functional Group**	**Mean**	**StDev**	**Mean**	**StDev**	
Sheep	2000	Shrubs[1]	27.3	11.3	7.7	7.4	0.01
		Total	131.4	53.3	45.7	34.6	0.02
	2001	Pinegrass	44.2	18.9	9.3	8.7	0.01
		Shrubs[1]	50.5	33.7	4.0	3.2	0.02
		Total	124.5	20.4	37.9	38.1	0.002
Wolf	2001	Forbs	38.9	25.0	13.0	9.1	0.06

[1] p-values are reported based on analysis using transformed data. Means and standard deviations of original data are presented.

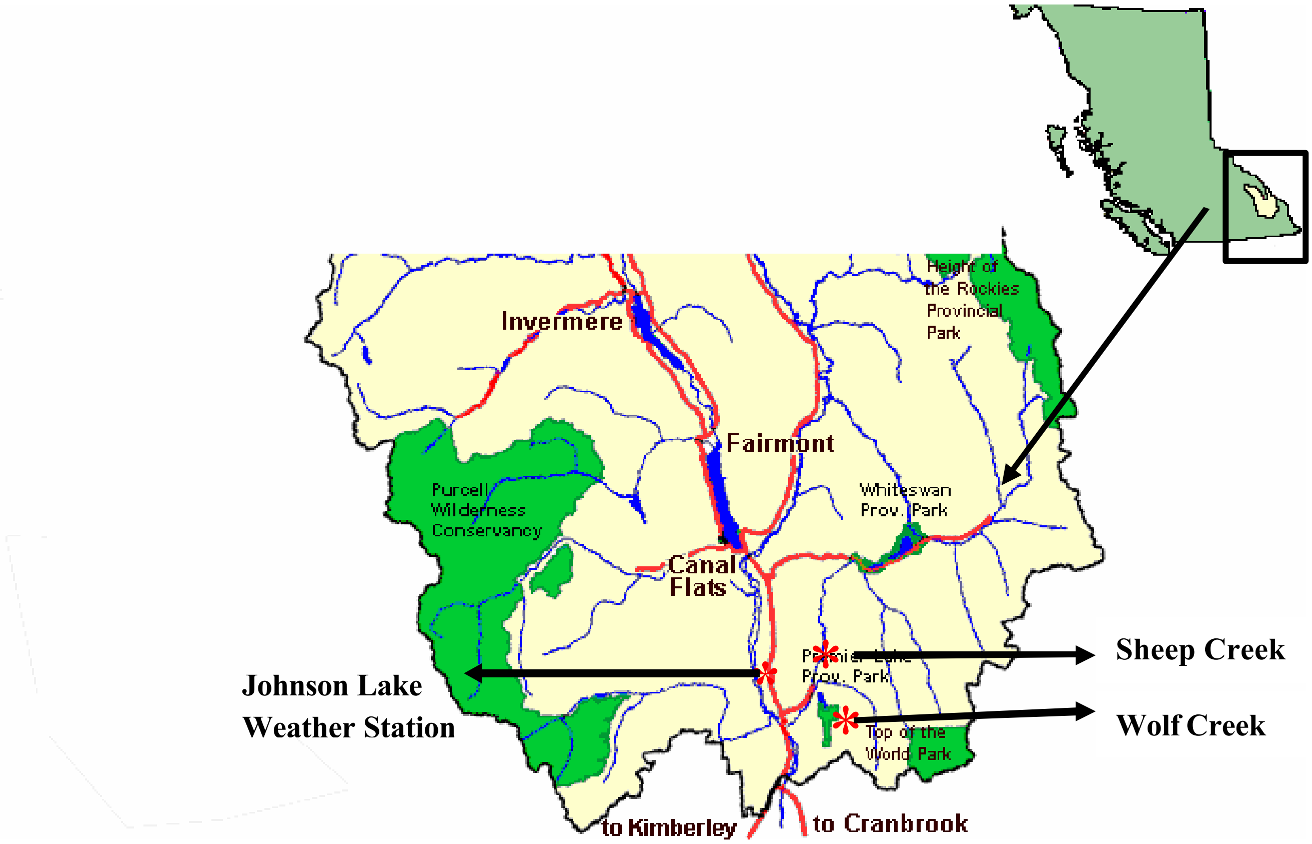

Fig. 3.1. Invermere Forest District. An (*) indicates the location of the two monitoring blocks (Sheep and Wolf Creek).

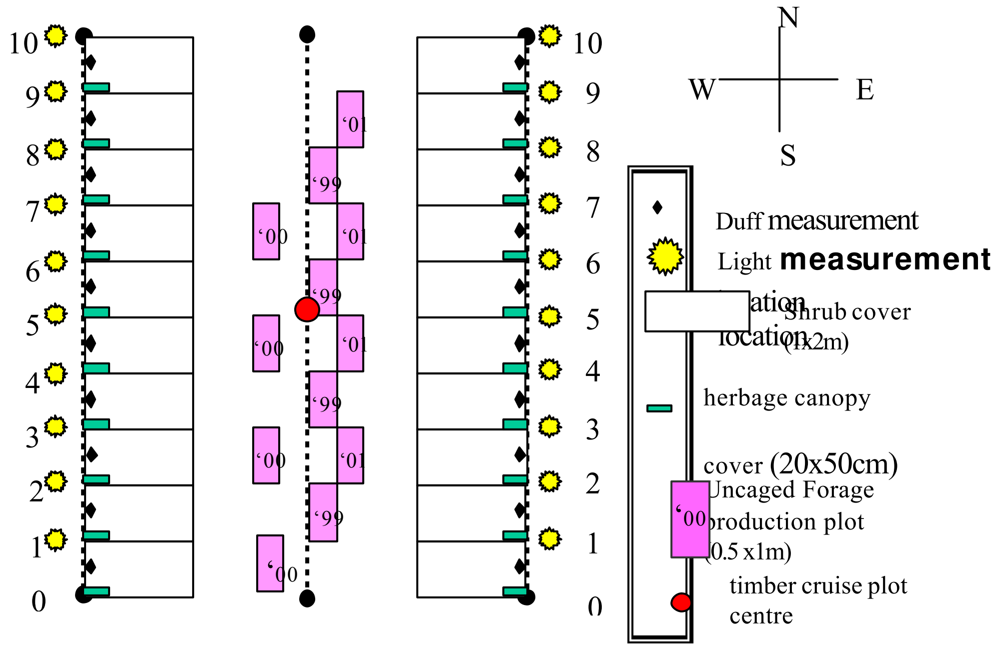

Fig. 3.2. Example of a plot used for sampling the understory (DeMaere 2001). Adapted from Powell et al. (1998).

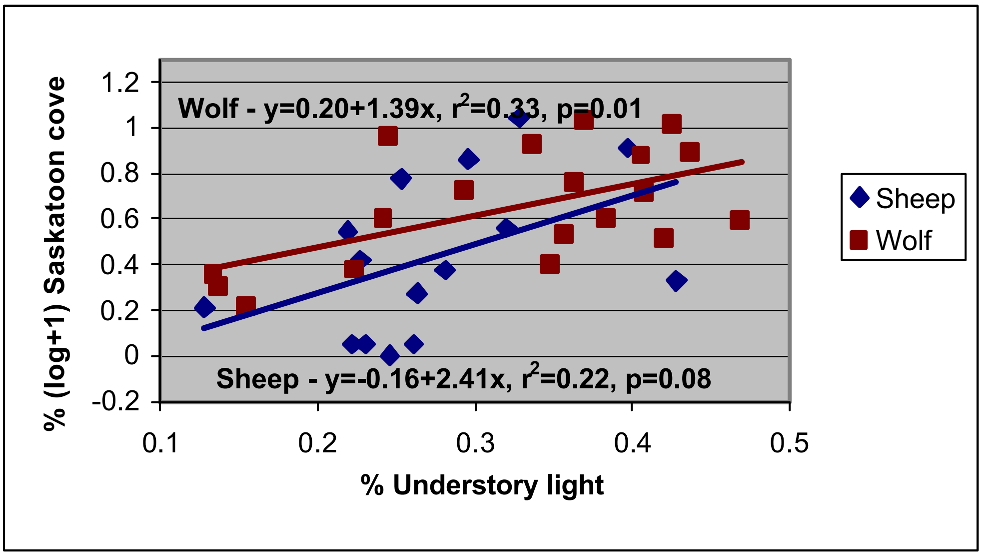

Fig. 3.3. Pre-thinning Saskatoon cover (%) regressed against understory light for both blocks in 1999.

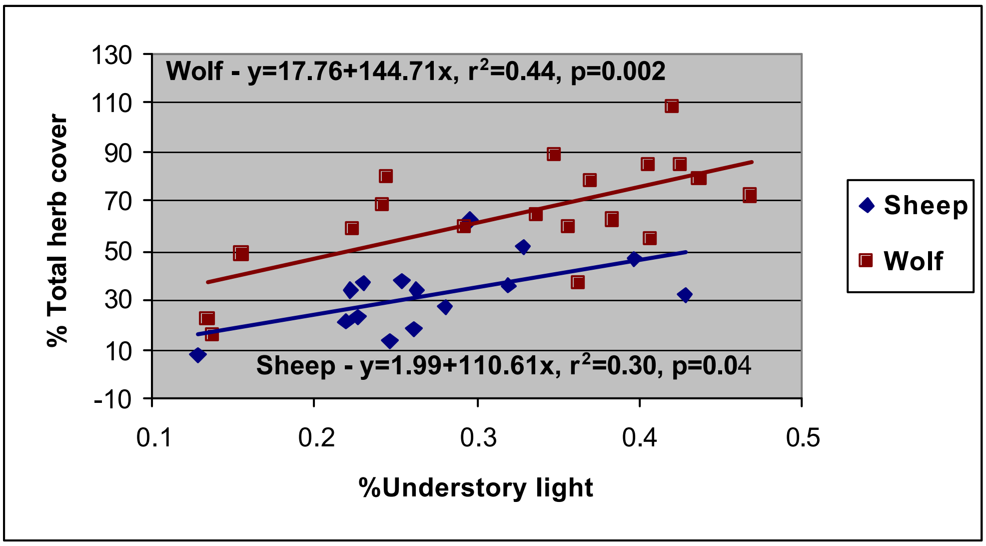

Fig. 3.4. Pre-thinning total herb cover (%) regressed against understory light for both blocks in 1999.

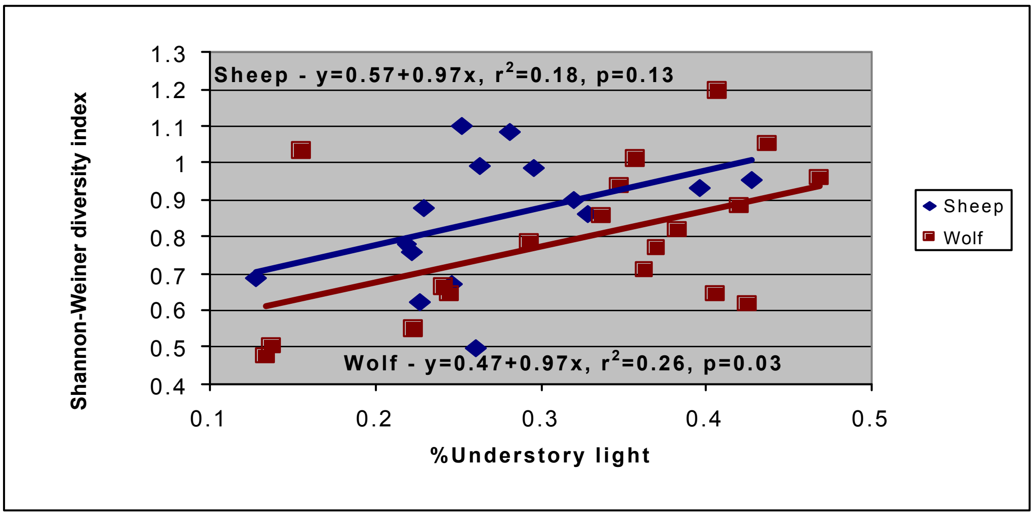

Fig. 3.5. Pre-thinning species diversity (Shannon-Weiner index) regressed against understory light for both blocks in 1999.

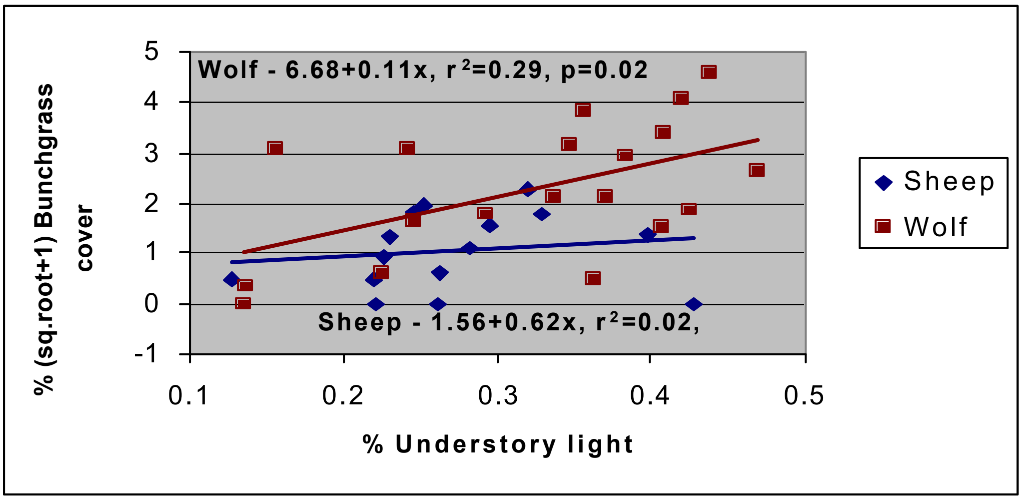

Fig. 3.6. Pre-thinning bunchgrass cover regressed against understory light for both blocks in 1999.

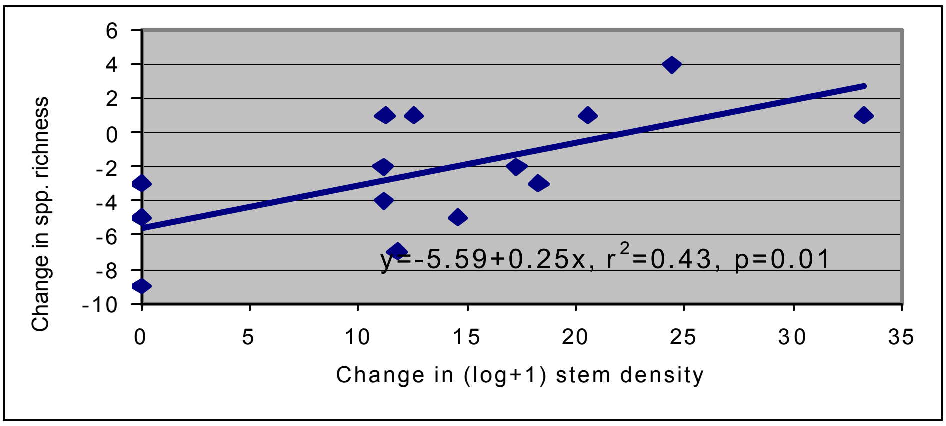

Fig. 3.7. Change in (?) species richness regressed against ? stem density (log+1 transformation) at Sheep Creek between 2000 and 2001.

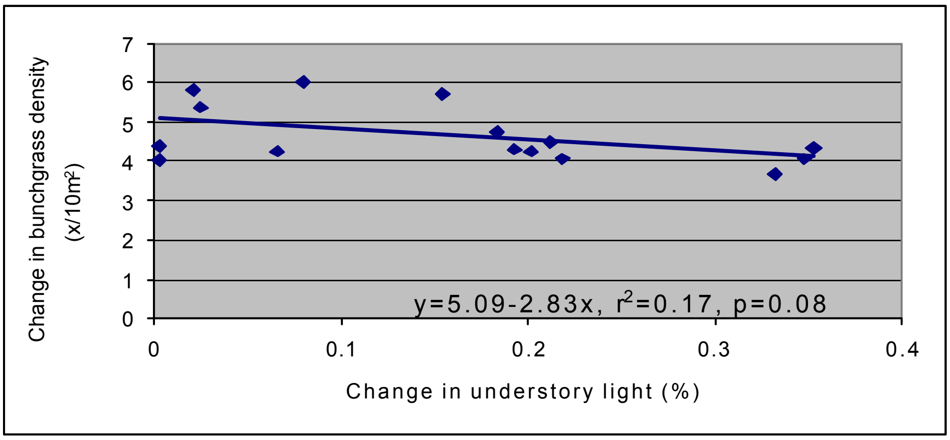

Fig. 3.8. Change in (?) bunchgrass density (x/10m^2) regressed against ? understory light at Sheep Creek between 1999 and 2001.

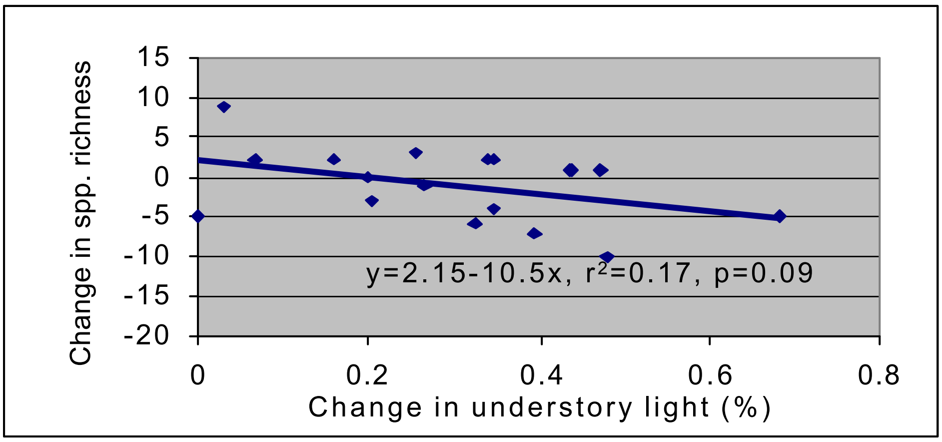

Fig. 3.9. Change in (?) species richness regressed against ? understory light at Wolf Creek between 1999 and 2001.

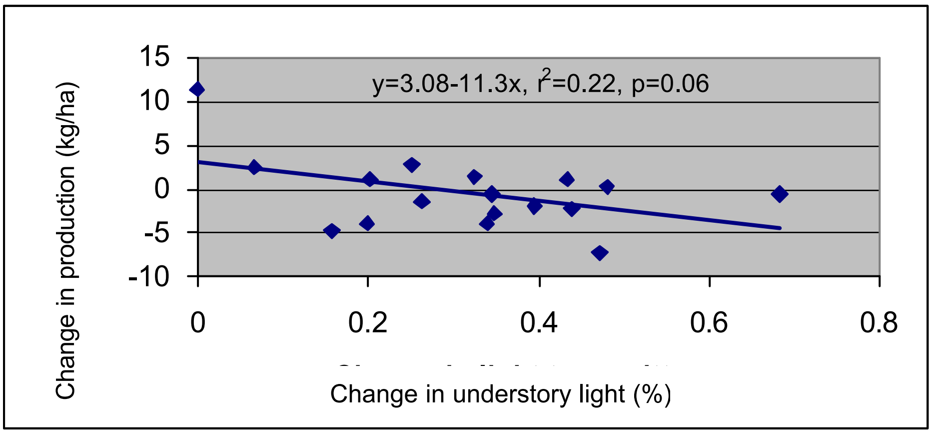

Fig. 3.10. Change in (?) production (kg/ha) regressed against ? understory light at Wolf Creek between 2000 and 2001.

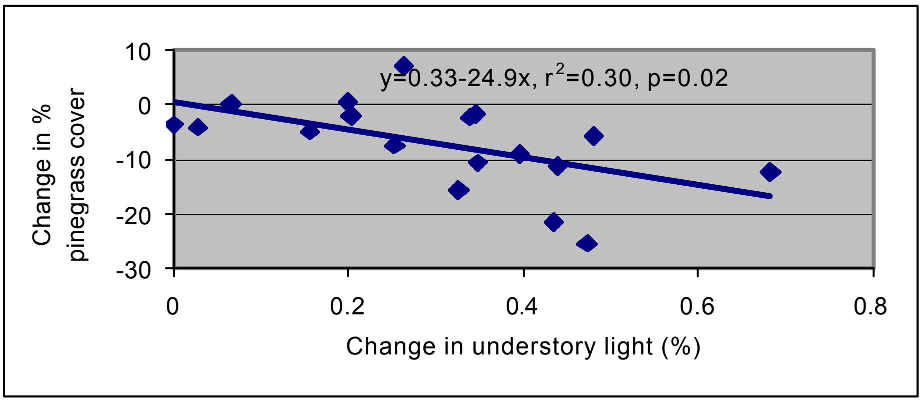

Fig. 3.11. Change in (?) pinegrass cover regressed against ? understory light at Wolf Creek between 1999 and 2001.

4. USING BUNCHGRASS PLUGS TO RESTORE DEGRADED RANGELAND IN THE EAST KOOTENAY.

4.1 Introduction

Forest understory species are comprised of a variety of growth forms with diverse physiologic tolerances and reproductive strategies that allow them to survive under the existing abiotic and biotic conditions (Riegel et al. 1995). As a forest overstory grows, understory species composition responds to changes in the quantity and quality of light, as well as the availability of soil water and nutrients (Riegel et al. 1992, 1995). Presumably, each individual understory species has an extinction point along a resource gradient. Over time, as resources such as light decline, the more productive and light demanding species disappear, creating room for the establishment and growth of other species more suited to the new conditions (Knowles et al. 1999).

In fire-maintained forests of North America, fire suppression has facilitated a change from open, dry stands to closed canopy, mesic stands of shade-tolerant and fire sensitive species (Cooper 1960, Arno and Gruell 1986, Lunan and Habeck 1973, Habeck 1990, Arno et al. 1995, Gayton 1997, Smith and Arno 1999, Arno et al. 2000). In the Rocky Mountain Trench of British Columbia (BC), this trend is evidenced by the ingrowth and encroachment of low- density veteran forests by younger age classes of interior conifer species, including Douglas- fir [*Pseudotsuga menziesii* var. *glauca* (Beissn.) Franco] and lodgepole pine (*Pinus contorta* var. *latifolia* Engelm. *ex*. S. Wats.) (Gayton 1997). Although open forests of the Rocky Mountain Trench are thought to be historically stable, these fire- maintained plant communities were maintained in a mid-seral state. Within these stands, shading caused by the invasion of conifer species as the stand moves toward a late-seral or climax state, has favored the invasion of mesophytic shrubs and herbs (Lunan and Habeck 1973). These late seral species can successfully out-compete desirable mid-seral species (including native bunchgrasses) that are intolerant of the new conditions, including low light (Tilman 1988).

For example, pinegrass (*Calamagrostis rubescens* Buckl.), a dominant rhizomatous species of northern inland forests (Franklin and Dyrness 1973), is often abundant under dense conifer canopies (Steele and Geier-Hayes 1993). In combination, decreased light and increased competition from pinegrass may limit the existence and distribution of native bunchgrasses that were once common prior to ingrowth.

Loss of the bunchgrass community is significant, particularly in the Rocky Mountain Trench, as native ungulates and livestock exhibit a high degree of preference for native bunchgrasses rough fescue (*Festuca campestris* Rydb.), Idaho fescue (*Festuca idahoensis* Elmer), bluebunch wheatgrass [*Pseudoroegneria spicata* (Pursh) A.Löve], Richardson's needlegrass (*Stipa richardsonii* Link.), needle-and-thread grass (*Stipa comata* Trin.&Rupr.) and stiff needlegrass (*Stipa occidentalis* Thurb. ex S. Wats. var. *pubescens* Maze, Tayor and MacBryde) (Jalkotzky, wildlife biologist, Arc Wildlife Services Ltd. per comm. 2001, Ross 2001). This could be attributed to the fact that pinegrass rapidly loses protein at advancing stages of maturity (Freyman 1970), while bunchgrasses retain fairly high protein values into the fall (Hooper and Pitt 1998). As a result, animal preference for pinegrass declines relative to other bunchgrasses (Gayton 1997), especially in the fall. Given that ponderosa pine and interior Douglas fir forests of the southern interior of BC are particularly important for wildlife as fall and winter range (e.g. Hudson et al. 1976), this loss of forage quality and quantity is a concern for wildlife biologists and rangeland managers (Gayton 1997, Jalkotzky, wildlife biologist, Arc Wildlife Services Ltd. per comm. 2001). Additionally, because ungulate numbers (including livestock) tend to remain relatively constant or even increase within the Rocky Mountain Trench (Gayton 1997), the gradual loss of bunchgrasses and their replacement with less palatable pinegrass may have implications for the overgrazing of remaining vegetation, resulting in further undesirable species composition shifts.

In response to the loss of open grassland (i.e., bunchgrass) dominated plant communities, the Invermere Forest District has initiated an extensive open forest and range restoration program using prescribed burning and silvicultural thinning (see Chapter#3). However, it is possible that overstory removal in these restoration programs may enhance pinegrass vigor and production, rather than restore displaced bunchgrass communities. Concurrent research is monitoring these understory responses, including that of pinegrass, to restoration activities (see Chapter#3).

The Invermere Forest District also initiated an intensive native seeding program in 1994 for the purpose of facilitating rangeland rehabilitation, road reclamation and ecosystem restoration (Invermere Forest District 2000). Success of the seeding trials was low due to poor germination. This is common of restoration seeding projects in arid and semi-arid rangeland, where projects fail because of the lack of moisture required for successful germination (Grantz et al. 1998). Due to the limited success of seeding, the use of transplants has been increasing since the 1980's for rangeland restoration and several techniques have been evaluated for successful establishment, in both a research and applied land management context (Bainbridge et al. 1995, Grantz et al. 1998).

The majority of transplantations for restoration work to date have been done using shrubs (Bainbridge et al. 1995, Grantz et al. 1998). In contrast, bunchgrass transplantation efforts have been used primarily to test hypotheses related to interspecific competition (Wilson 1994, Gerry and Wilson 1995, Peltzer and Wilson 2001) rather than applications in ecosystem restoration. Furthermore, the BC provincial government (BC Ministry of Forests and the BC Ministry of Water, Land and Air Protection) is one of the few public land administrations known to be currently investigating the use of bunchgrass plugs for widespread restoration. The Invermere Forest District proposed the use of bunchgrass transplants for the purpose of restoration in 1997. Bypassing the seeding, germination and emergence phases of plant establishment was thought to increase the opportunity for successful bunchgrass restoration. Native seed (local to the area), including rough fescue, bluebunch wheatgrass and Richardson's needlegrass, was grown into 'bunchgrass plugs' under greenhouse conditions for subsequent planting on degraded sites.

This research was designed to assess the feasibility of using two species (bluebunch wheatgrass and Richardson's needlegrass) for use in restoring ingrown forests. In this context, bunchgrass plugs are specifically being used to restore bunchgrasses to thinned ingrown forests where they were once common in grasslands and open forests. The goal of this experiment was to assess the success of transplanting native bunchgrass plugs.

The following null hypotheses were tested:

? There is no difference in survival and vigor between the species,

? There is no competition effect of pinegrass on survival and vigor of the bunchgrass transplants,

? There is no difference in survival and vigor when bunchgrass plugs are planted in different spring or fall.

4.2 Methods

4.2.1 Study Area

Planting trials were conducted at 3 blocks, all located within 100 km of each other within the Rocky Mountain Trench of BC (Fig. 4.1). The region is strongly influenced by maritime polar air masses that are drier after being lifted over the Coast, Monashee, and Selkirk Mountains of BC (Marsh 1986). The southern valley has an upland continental climate with well-defined seasons (Marsh 1986). Summers are characterized as warm and dry while winters are cold with deep valley inversions (Marsh 1986), which makes the winters relatively warm at low elevation sites (McClean and Holland 1957). Mean monthly air temperatures vary from –8.3? to 18.2?C, while average annual precipitation is 384.5 mm, with May and June being the wettest months (Table 4.1). There is an average of 147.9 cm of snow during the winter months. This project was initiated in 2001 when summer rainfall was 35% of the long-term average during the growing season (May-September) (Table 4.1). Precipitation and temperature values were obtained from the Johnson Lake weather station in close proximity to the 2 southernmost blocks (Fig. 4.1). Precipitation values for the northernmost block (Zehnder) were obtained from a rain gauge set up at the block, although there is no data available for the year prior to planting.

All blocks were uniform in vegetation type (interior Douglas-fir), abundant in pinegrass, and lacking ingrowth to give full light conditions (i.e. were recently thinned stands). All blocks were selected to avoid confounding effects of moisture gradients and shading. One site (northernmost site) was located in the IDFun (undifferentiated interior Douglas-fir (Windermere Lake) unit) and the other 2 sites within the IDFdm2 (Kootenay dry mild interior Douglas-fir variant). Zonal IDFdm2 sites have climax stands of Douglas-fir with an understory dominated by pinegrass and a high cover of shrubs such as birch-leaved spirea [*Spiraea betulifolia* Pall. ssp. *lucida* (Dougl. ex Greene) Taylor & MacBryde)], common juniper (*Juniperus communis* L.), soopolallie [*Shepherdia canadensis* (L.) Nutt.], Saskatoon (*Amelanchier alnifolia* Nutt.), and common snowberry [*Symphoricarpos albus* (L.) Blake] (Braumandl and Curran 1992). Zonal IDFun sites have open stands of Douglas fir, with no other dominant tree species. Dominant understory species are bluebunch wheatgrass [*Pseudoroegneria spicata* (Pursh) A. Löve] and junegrass (*Koeleria macrantha* Ledeb.) (Braumandl and Curran 1992). Current commercial uses of these areas include cattle grazing. To test the operational success of transplanting plugs, sites were not protected from grazing during the establishment trials.

Soils at all 3 blocks were characterized by Orthic Eutric Brunisols (Lacelle 1990). The dominant soil association is Fishertown, a gravelly, sandy loam derived from fluvioglacial parent material, with rapid drainage (Lacelle 1990). There are also minor occurrences at all sites of the Wycliffe association, a Brunisolic soil derived from morainal parent material containing limestone (Lacelle 1990).

4.2.2 Experimental Design

Each experimental block was 6.0 by 7.2 m in dimension ($43.2m^2$). Bunchgrass plugs were systematically planted 60 cm apart in the fall (8-9 October, 2000) and in the spring (8-11 May, 2001). The treatments were *S. richardsonii*-pinegrass removal, *S. richardsonii*-no pinegrass removal, *P. spicatum*-pinegrass removal and *P. spicatum*-no pinegrass removal. Pinegrass was removed using glyphosate, a systemic, translocated, non-residual herbicide. Glyphosate was applied directly to pinegrass plants only by wiping the herbicide (7g/L concentration) onto leaves with a cloth. During treatment, transplants had a glass jar placed over them to prevent exposure to glyphosate. Herbicide treatments had a significant ($p<0.001$) effect on the cover of pinegrass at all blocks in both seasons, reducing pinegrass by an average of 8% (13% - 3%) (Table 4.2). Additionally, the South block initially had a significantly lower percent cover of pinegrass compared to the other 2 blocks ($p<0.001$). There was no supplementary watering provided to plugs to ensure natural field conditions.

Eighty plugs of each species were planted at each of the 3 blocks in October 2000, with 20 randomly assigned to each of the 4 treatments. In the spring, 32 additional plugs were planted at each of 2 blocks, with 8 plugs randomly assigned to each treatment. Samples sizes were reduced in the spring due to a limited supply of plugs. Plant height, tiller numbers, and basal area were assessed for each plug at the time of establishment. All plugs were grown from seed at the Skimikin Nursery in Tappen, BC. Plugs planted in the fall were subsequently examined on 8 May, 2001 for overwinter survival Both spring and fall planted plugs were monitored for survival, height, basal area, number of tillers and number of inflorescences during September 2001.

4.2.3 Statistical Analyses

To meet the assumptions of analysis of variance, all data were tested for normality using univariate procedures in SAS (SAS 1999), with no transformations necessary. The effect of season, species and pinegrass removal on plug survival was tested using a split-plot design, with season of planting as the main plot factor. Survival percentages for each treatment combination were used as observations in the model (i.e. 16 observations with no subsampling). The effect of bunchgrass size (i.e. initial tiller number, height and basal area) on survival was also tested using a one-way ANOVA. For this analysis, each bunchgrass species was examined separately, however, as the two species have inherently different tiller numbers (i.e. *S. richardsonii* generally has a greater number of tillers than *P. spicatum*).

Survival was lower than expected for *S. richardsonii* plugs planted in the fall, and also variable across blocks, seasons and species (Table 4.3), resulting in an unbalanced number of remaining plugs among treatment groups. An ANOVA using a split-plot design, identical to the survival analysis, was conducted on the surviving plugs to determine the effect of planting season, pinegrass removal, and plug species on growth characteristics (i.e. tiller numbers). Where higher level block interactions were detected (2-way or 3-way), blocks that were responding differently were isolated and analyzed separately.

Due to unequal sample sizes among treatment combinations based on variable survival, treatment effects were further analyzed by season using a fully randomized block design (i.e. seasons were analyzed separately). This was done to isolate possible pinegrass effects within each season.

Although basal area and plant height were measured on each transplant, there was no evidence of a change in basal area (or the change was too small to analyze) and grazing effects confounded any changes in height. As a result, these latter variables were dropped from the analysis. All results were considered significant at $p<0.10$, unless noted otherwise.

4.3 Results

4.3.1 Survival

Pinegrass removal effected ($p=0.02$) the survival of bunchgrass plugs (Table 4.4). Plugs with the surrounding pinegrass removed had a significantly greater survival rate than those plugs with no pinegrass removed (Fig. 4.2). There were also significant species and season by species effects ($p<0.0001$) (Table 4.4), as a greater percentage of *P. spicatum* plugs survived overall compared to *S. richardsonii* plugs (Table 4.3). The season by species interaction was due to greater survival of fall-planted *P. spicatum* plugs compared to *S. richardsonii*. Conversely, a greater number of *S. richardsonii* plugs survived when planted in the spring than *P. spicatum* (Table 4.3, Fig. 4.3).

Additional analysis indicated the size of the bunchgrass plug, as determined by initial tiller number, basal area and height, generally had a significant effect (minimum $p<0.10$) on the survival of both species. These results indicate larger plugs had a greater likelihood of survival during the first year after establishment under the conditions of this investigation (Table 4.5).

4.3.2 Bunchgrass Plug Growth

Changes in tiller number within the remaining live plants were also effected by the treatments. A significant season by species interaction was again evident (p=0.0003) (Table 4.6). *S. richardsonii* plugs that managed to survive fall planting lost fewer tillers compared to those planted in the spring (p=0.04). Similarly, *P. spicatum* plugs that survived spring planting lost fewer tillers than those surviving from the fall (p=0.002). These results are in contrast to the survival data outlined earlier.

When the fall and spring planted plugs were analyzed separately, significant interactions between block and pinegrass, as well as block, pinegrass, and species, were found (p<0.01). Further examination indicated the south block behaved significantly different than either the North or Zehnder blocks. The inconsistent results are likely due to initial differences in pinegrass cover among blocks (6% at the South block compared to 23.5% and 15% at the Zehnder and North blocks, respectively), and a greater level of grazing at the South block (15% of plugs were grazed, versus 2% at the North and Zehnder block). As a result, the South block was analyzed independently of the others.

When the North and Zehnder block were combined there was a significant pinegrass effect (p=0.02) and a significant species effect (p=0.05) in the fall planted plugs (Table 4.7). Bunchgrass plugs in the pinegrass removal treatment lost significantly fewer tillers than those with no pinegrass removal. Between species, *P. spicatum* plugs lost a greater number of tillers overall (63%) versus *S. richardsonii* plugs (10%) (p=0.05). There was also a pinegrass by species interaction (p=0.07) (Table 4.7), which was largely due to the positive effect of pinegrass removal on tiller numbers in *S. richardsonii* plugs (Fig. 4.4).

When the fall planted South block was analyzed in isolation, there was a significant species by pinegrass interaction ($p<0.10$; Table 4.8), with *S. richardsonii* plugs losing a larger number of tillers when the adjacent pinegrass was removed (Fig. 4.5).

Among the spring planting treatments, there was a significant block ($p=0.08$) and species ($p=0.03$) effect (Table 4.9). Plugs in the Zehnder block lost fewer tillers (10%) across both species than those in the North block (44%). Additionally, *S. richardsonii* plugs in both blocks lost a greater number of tillers than plugs of *P. spicatum* (Fig. 4.6). The absence of higher level interactions (e.g. block by main treatment effects) indicated treatments within these 2 blocks behaved similarly. There were no significant pinegrass treatment effects with spring planting.

Although inflorescence data were too variable to detect differences, all the plugs that did produce seedheads were planted in the fall (32 of 240) rather than spring. Two of these were *S. richardsonii* plugs while 30 were *P. spicatum*.

4.4 Discussion

Fall planted plugs were generally more likely to survive under the conditions of this study. Despite this, the favorable survival of *S. richardsonii* with spring planting complements successful spring planting trials completed by the Invermere Forest District using this species (Invermere Forest District 2000). Preliminary field studies completed by range ecologists in the Invermere Forest District showed that survivorship when transplanting *S. richardsonii* was 94% without grazing and 50% with grazing. All plugs were planted in the spring (May 21) at the same location. In that same trial, however, survivorship of *P. spicatum* plugs was considerably lower (3.6%), reinforcing the results found here that this species is not adapted to spring planting. The current study also found survival of *S. richardsonii* to be much lower than *P. spicatum*, particularly in the fall planting treatment (Table 4.3). Overall, the results observed here indicate survival can be optimized by planting *P. spicatum* plugs in the fall and *S. richardsonii* in the spring. Abnormally dry weather conditions combined with other stresses may have contributed to the seasonal and intraspecific variation found in the 2001 growing season. Separate trials and continued monitoring of transplanted plugs at these 3 blocks are needed to broaden the temporal scope of inference.

Differential survival rates for spring and fall planting may be related to the biology of the 2 species. *P. spicatum* initiates growth in the early spring, as early as the third week of February (Willms et al. 1980). Thus, planting this species in the spring (i.e. May) will shorten its growing season considerably. Parsons et al. (1971) found *P. spicatum* required 51 days for the completion of reproductive development, while needle and thread grass (*S. comata*) required only 18 days. Rapid growth of needle and thread grass appears to be related to an increase in temperature (Parsons et al. 1971), which implies this species behaves similar to a C_4 (warm season) rather than a C_3 (cool season) species. *Stipa* species have been reported to behave similar to C_4 species, growing well in relatively hot and dry climates (Gurevitch 1986). This may be related to anatomical and morphological characteristics associated with drought tolerance. Rolled leaves, and prolonged metabolic activity after the onset of dry conditions both contribute to superior drought tolerance of this C_3 species (Gurevitch 1986) relative to other C_3 species. The growth of *P. spicatum*, a C_3 species, appears to be unaffected by temperature (Willms et al. 1980). Planting *S. richardsonii* in the fall may decrease its chance of survival due to low temperatures, while planting *P. spicatum* in the spring may decrease its chance of survival due to a shortened growing season. Although season of planting had a clear effect on survival, the effect on growth is not as clear. While *S. richardsonii* plugs planted in the fall grew better than *P. spicatum* plugs, the opposite relationship was true with spring planting. This is likely a result of selection for strong growth traits, as plugs that survived in sub-optimal planting conditions will have traits that predispose them to superior growth, resulting in a bias within the surviving plugs towards greater tiller increases (or fewer tiller losses).

These results are further supported by the established selection bias within plugs that survived planting trials in favor of larger individuals (Table 4.5). Studies examining inter-specific competitive responses of grass plugs in North American grasslands have found that initial plant size confers a competitive advantage over other species (Wilson 1994, Gerry and Wilson 1995). These findings are therefore consistent with the observations in this study and suggest that larger plugs should be used in restoration projects to maximize the potential for bunchgrass establishment and growth.

Another factor that affected the survival and vigor of plugs was pinegrass removal. Competition for limited resources may determine the presence, absence, or abundance of species in a community and determine their spatial arrangement (Pyke and Archer 1991). Pinegrass competition had an adverse impact on plug survival in both planting seasons (Fig. 4.2). Pinegrass is a rhizomatous species that initiates growth early in the spring (McLean 1979). Due to its shallow rooting habit and early emergence, pinegrass is a very effective competitor for the limited moisture found in NDT4 stands. For example, Peterson (1988) noted that pinegrass competition had a negative impact on ponderosa pine seedling stemwood, foliage and root weight. Studies have shown early emerging species continually increase their ability to capture resources at the expense of later emergers, and in doing so, increase their physical zone of influence (Ross and Harper 1972). Therefore, when transplants are grown in the presence of early growing neighbours, their growth and establishment is compromised (e.g. Ross and Harper 1972, Wilson 1994, Gerry and Wilson 1995, Peltzer and Wilson 2001).

It appears that although pinegrass competition affected plug survival in both seasons it had a greater impact on *S. richardsonii* rather than *P. spicatum* growth (Fig. 4.4). *P. spicatum* is an early-emerging species and slightly rhizomatous, and may be a more effective competitor against pinegrass for moisture. The lack of an effect of pinegrass removal on change in tiller numbers in the spring planting treatment (Fig. 4.6) could be due to abiotic conditions at the time of planting. Transplant shock and the lack of moisture may have limited spring growth rather than pinegrass competition.

Drought during 2000 and 2001 (Table 4.1) may also have affected plug survival and growth in this investigation. The year prior to planting (2000) was unusually dry (~45% of normal, May-September) as was the year of planting (~45% of normal, May-September) (Table 4.1). Precipitation was greater at the Zehnder block (Table 4.1) during the growing season, however, and may be responsible for the better plug growth at this site within the spring planting treatment (Table 4.9).

Although inflorescence production was limited in this study, plugs of *P. spicatum*, particularly those planted in the fall, did exhibit considerable seedhead production. This response is important as it represents an important recovery mechanism (rebuilding the soil seedbank) for this key bunchgrass, thereby increasing the likelihood for additional increases in this species.

4.5 Conclusions and Recommendations

S. richardsonii and *P. spicatum* plugs are both good candidates for restoration of recently restored forests, if planted in the proper season, and are able to establish even during drought conditions. Using larger plugs will also increase the chance of survival. Agencies practicing restoration should consider transplanting *P. spicatum* plugs in the fall and *S. richardsonii* in the spring to maximize the survival of plugs. Fall planting trials need to be examined further for *S. richardsonii*.

Although both species appeared to be negatively affected by pinegrass competition, they each maintained moderate survival across both planting seasons. Factors such as initiation of growth, tolerance of drought, grazing, initial plug size and time needed for development all need to be considered when planning a restoration strategy.

It is recommended that plugs be monitored for several years to ensure these results are not anomalous. This will allow for a more comprehensive evaluation of the effect of pinegrass competition on plug survival and growth.

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Table 4.1. Temperature and precipitation at Johnson Lake weather station[1], 2000-2001, precipitation data at Zehnder Block[2], 2000 and long-term average[3] temperature and precipitation.

	Zehnder Block		Johnson Lake				Long-Term Avg.	
	2000		2000		2001			
	Temp. (?C)	Ppt. (mm)	Temp. (?C)	Ppt. (mm)	Temp. (?C)	Ppt. (mm)	Temp. (?C)	Ppt. (mm)
May	n/a	4	10.6	22	12.8	7.7	11.2	43.6
June	n/a	18	14.8	16.9	14.3	29.2	14.9	50.5
July	n/a	35	19.5	13.8	19.4	16	18.3	31.6
August	n/a	35	18.2	12.1	20.6	4.2	17.8	34.4
September	n/a	17	11	21.9	14.1	13.8	12.4	32.6

[1]Location-49?55'N 115?44'W, Elev.-853m.

[2]Data based on rain gauge data established by researcher, no temperature data available.

[3] Cranbrook Airport Weather Station. 49?37'N 115?47'W, Elev.-939m. Avg. based on 22 yrs. (1968-1990).

Table 4.2. Average canopy cover (%) of pinegrass across 3 blocks before and after pinegrass removal.

	no pinegrass removal		pinegrass removal		
Block	**Mean**	**Stdev**	**Mean**	**Stdev**	**p-value**[1]
North	20.96	19.54	7.18	9.53	<0.001
South	6.35	7.09	3.34	5.28	0.31
Zehnder	16.54	16.33	10.16	12.82	0.01

[1] Means are compared across the rows.

Table 4.3. Survival of *S. richardsonii* and *P. spicatum* plugs in 2001 across 3 blocks, 4 treatments and 2 seasons. No plugs were planted at the south block in the spring.

			Survival %[1]		
Season	Block	Species	Pinegrass Removed	With Pinegrass	Combined
Fall[1]	North	*S. richardsonii*	35	25	30
		P. spicatum	95	95	95
	South	*S. richardsonii*	15	10	12.5
		P. spicatum	55	53	54
	Zehnder	*S. richardsonii*	25	10	17.5
		P. spicatum	100	90	95
	Total	*S. richardsonii*	**25**	**15**	**20**
		P. spicatum	**83**	**79**	**81**
Spring[2]	North	*S. richardsonii*	78	67	73
		P. spicatum	50	38	44
	Zehnder	*S. richardsonii*	63	63	63
		P. spicatum	50	37	44
	Total	*S. richardsonii*	**71**	**65**	**68**
		P. spicatum	**50**	**38**	**44**

[1] n=20 per treatment combination at each block.
[2] n=8 per treatment combination at each block.

Table 4.4. ANOVA summary table of effect of planting season, pinegrass removal and species on survival of bunchgrass plugs.

Source	DF	Type III SS	Mean Square	F Value	Pr>F
Season	1	39.06	39.06	1.23	0.31
Block(season) Error1	2	110.12	55.06	1.74	0.25
Pinegrass	1	280.56	280.56	8.84	**0.02**
Species	1	2139.06	2139.06	67.42	**0.0002**
Pinegrass*species	1	0.56	0.56	0.02	0.90
Season*pinegrass	1	0.56	0.56	0.02	0.90
Season*species	1	9264.06	9264.06	291.97	**<0.0001**
Season*pinegrass*species	1	68.06	68.06	2.15	0.19
Error2	6	190.38	31.73		

Table 4.5. Comparison of initial morphologic characteristics between bunchgrass plugs that subsequently survived and died during the 2001 growing season.

Characteristic	Species	Live Plugs Mean	Live Plugs StDev	Dead Plugs Mean	Dead Plugs StDev	p-value[1]
Basal area (cm^2)	*S. richardsonii*	2.80	0.84	1.45	0.3	**0.02**
	P. spicatum	1.65	0.48	1.02	0.66	**0.002**
Height (cm)	*S. richardsonii*	14.2	7.39	9.52	4.63	**<0.001**
	P. spicatum	27.0	5.80	25.16	7.83	**0.07**
Tiller number (#)	*S. richardsonii*	12.3	3.96	13.1	5.16	0.38
	P. spicatum	8.14	3.49	6.35	3.86	**0.009**

[1] p-values compare means between live and dead plugs within a row.

Table 4.6. ANOVA summary table of the effects of plug species and pinegrass removal on changes in tiller numbers during the 2001 growing season. Data includes both planting seasons.

Source	DF	Type III SS	Mean Square	F Value	Pr>F
Season	1	3.69	3.69	0.18	0.67
Block(season) Error1	2	97.51	48.75	2.40	0.10
Pinegrass	1	30.73	30.73	1.51	0.22
Species	1	15.87	15.87	0.78	0.38
Pinegrass*species	1	21.47	21.47	1.06	0.31
Pinegrass*season	1	44.01	44.01	2.16	0.14
Season*species	1	276.86	276.86	13.6	**0.0003**
Pinegrass*species*season	1	16.56	16.56	0.81	0.37
Error2	118	2401.97	20.36		

Table 4.7. ANOVA table testing effects of plug species and pinegrass removal on changes in tiller numbers from October 2000 to August 2001 (fall planting) at 2 blocks (North and Zehnder) ($p<0.1$).

Source	DF	Type III SS	Mean Square	F Value	Pr>F
Block	1	17.75	17.75	1.02	0.32
Species	1	71.64	71.64	4.11	**0.05**
Pinegrass	1	104.79	104.79	6.01	**0.02**
Species*pinegrass	1	58.97	58.97	3.38	**0.07**
Block*pinegrass	1	0.65	0.65	0.04	0.85
Block*species	1	40.07	40.07	2.30	0.13
Block*pinegrass*species	1	2.26	2.26	0.13	0.72
Error	84	1465.28	17.44		

Table 4.8. ANOVA table testing effects of plug species and pinegrass removal on changes in tiller numbers from October 2000 to August 2001 (fall planting) at the South block ($p<0.1$).

Source	DF	Type III SS	Mean Square	F Value	Pr>F
Species	1	0.67	0.67	0.04	0.84
Pinegrass	1	33.95	33.95	2.09	0.16
Species*pinegrass	1	75.09	75.09	4.63	**0.04**
Error	22	356.81	16.82		

Table 4.9. ANOVA table testing effects of plug species and pinegrass removal on changes in tiller numbers from May 2001 to August 2001 (spring planting) at 2 blocks (North and Zehnder) ($p<0.1$).

Source	DF	Type III SS	Mean Square	F Value	Pr>F
Block	1	97.95	97.95	3.31	**0.08**
Species	1	159.03	159.03	5.37	**0.03**
Pinegrass	1	1.56	1.56	0.05	0.82
Species*pinegrass	1	0.23	0.23	0.01	0.93
Block*species	1	27.94	27.94	0.94	0.34
Block*pinegrass	1	34.04	34.04	1.15	0.29
Block*species*pinegrass	1	7.62	7.62	0.26	0.62
Error	28	828.98	29.61		

Figure 4.1. Invermere Forest District. The location of each of the three planting blocks is denoted with an (*)

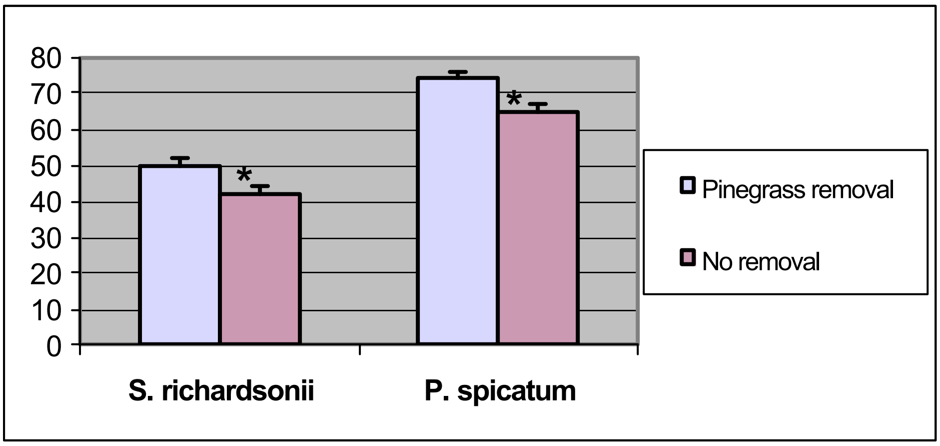

Fig. 4.2. Effect of pinegrass removal on survival (%) of two species of bunchgrass plugs in 2001. Within a species, an (*) indicates a significant difference (p<0.1). Data includes both planting seasons.

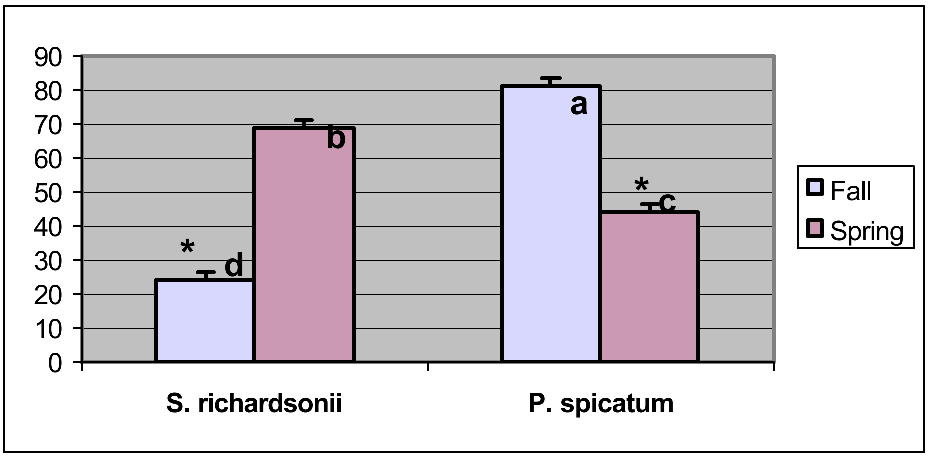

Fig. 4.3. Effect of planting season on survival (%) of two species of bunchgrass plugs in 2001. Within a species, an (*) indicates a significant difference (p<0.05). Among all treatments, means with different letters differ significantly (p<0.05).

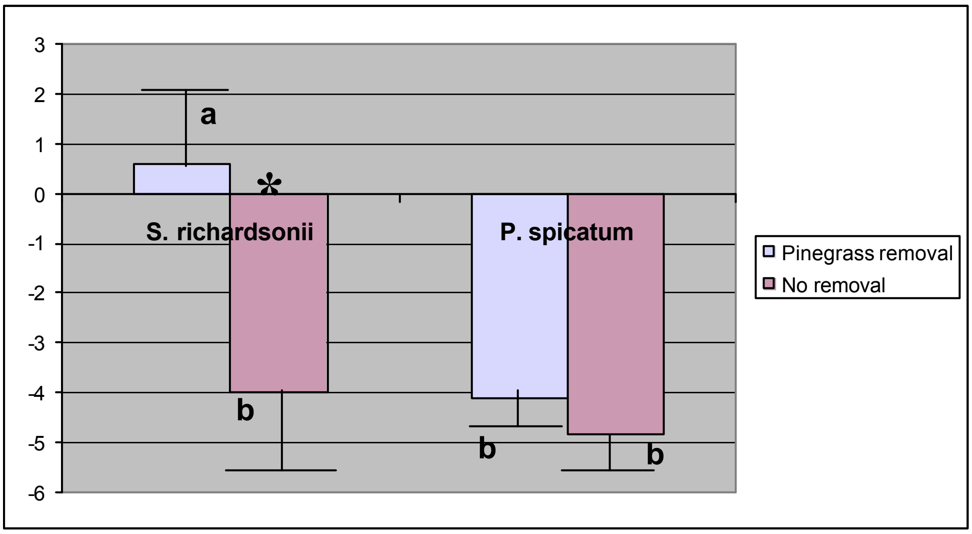

Fig. 4.4. Change in tiller number during the 2001 growing season for each of 2 species and 2 levels of pinegrass at 2 blocks following fall planting. Within a species, an (*) indicates a significant difference ($p<0.1$). Among all treatments, means with different letters differ significantly ($p<0.05$).

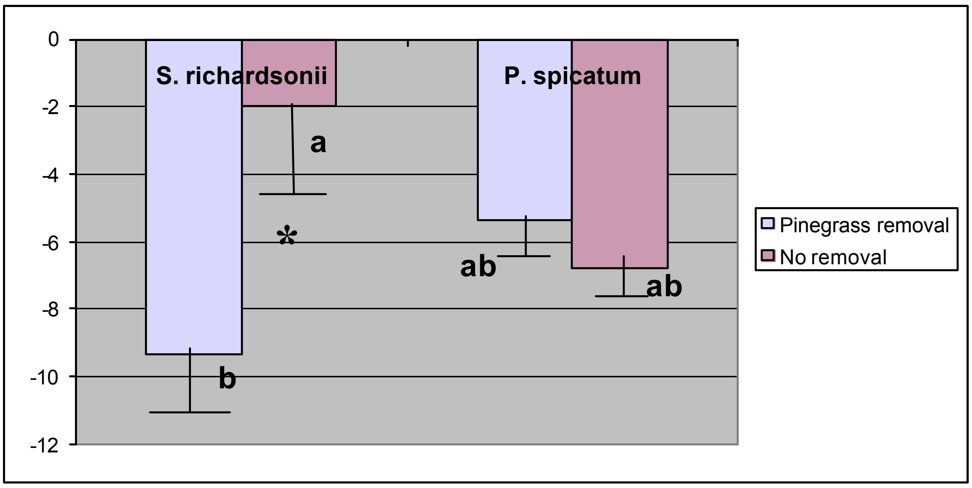

Fig. 4.5. Change in tiller number during the 2001 growing season for each of 2 species and 2 levels of pinegrass at 1 block following fall planting. Within a species, an (*) indicates a significant difference ($p<0.1$). Among all treatments, means with different letters differ significantly ($p<0.05$).

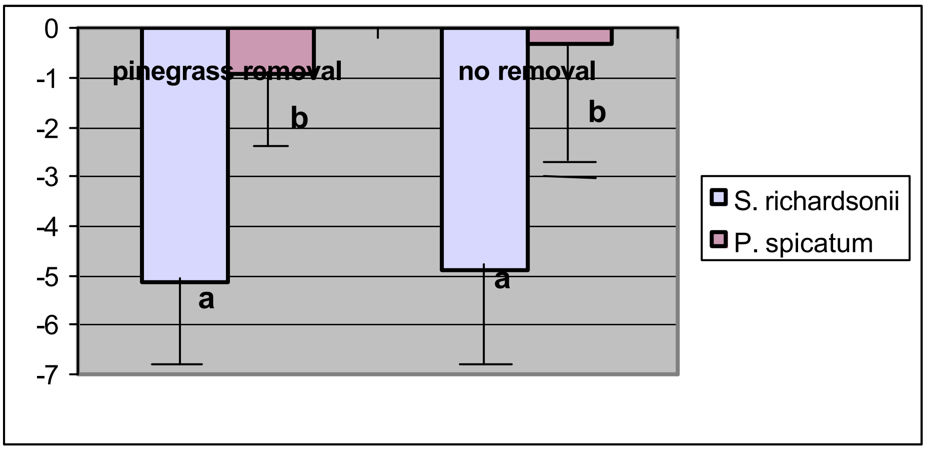

Fig. 4.6. Change in tiller number during the 2001 growing season for each of 2 species and 2 levels of pinegrass at 2 blocks following spring planting. Among all treatments, means with different letters differ significantly ($p<0.05$).

5. SYNTHESIS

Fire suppression in combination with other factors has increased the fire return interval in the Rocky Mountain Trench of British Columbia, resulting in significant changes on the landscape (Gayton 1996, Bai et al. 2001). Studies documenting the effect of conifer ingrowth and encroachment in other North American dry forests have documented similar trends. Changes within these forests include decreased N mineralization (Kaye and Hart 1998), decreased understory production (Pase 1958, Cooper 1960, Moir 1966, Borjoquez et al. 1989, Knowles et al. 1999) and decreased species diversity (Covington et al. 1997, Uresk and Severson 1998).

At the ponderosa pine and interior Douglas fir sites investigated in this study, understory light was positively related to species diversity and richness, bunchgrass density and cover, as well as the abundance of key browse species (e.g. Saskatoon) (Chapter #3). Collectively, these findings demonstrate the canopy cover and production of understory vegetation of NDT4 stands is adversely affected by conifer ingrowth and encroachment or lack of understory light. Mesophytic species such as pinegrass and birch-leaved spiraea had either no relationship with tree canopy cover (e.g. pinegrass) or a positive relationship with increasing canopy cover (e.g. birch-leaved spiraea), indicating these species are not adversely affected by ingrowth. Thus, the loss of shade intolerant species will not only reduce diversity in these stands but result in the loss of important forage sources for wild ungulates and livestock.

Although not specifically quantified in the East Kootenay until now, these trends had been suspected by land management agencies. Each year, approximately 3,000 ha of native grassland and open forest in the Trench become closed forest and an estimated

114,000 ha have been impacted since 1952 (Braumandl et al. 1994, Rocky Mountain Trench Ecosystem Restoration Steering Committee 2000). The Rocky Mountain Trench restoration program was designed to address the loss of open, bunchgrass-dominated communities to dense, ingrown forests. As stated by the Restoration Steering committee, restoration activities will achieve target distributions of four "ecosystem components" (i.e., shrublands, grassland, open forest and managed forest across the landscape (Table 5.1). Restoration "treatments will contribute to the creation of a complex, ecologically-appropriate mosaic of habitats over the long-term", and "treatments in open range and open forest will remove excess immature and understory trees and emphasize retention of the oldest and/or largest trees". Once stands have received initial ecosystem restoration treatment, they will become part of a long-term cycle of harvesting, spacing and prescribed burning that will optimize retention of veteran and large trees and snags; conservation of wildlife habitat, biodiversity and endangered and threatened species and communities; optimization of forage production; minimization of weed occurrence; and protection of ecosystem health.

Post-thinning monitoring (Chapter #3) was designed to monitor progress towards meeting the goals of maintaining ecosystem health and biodiversity, forage production, and the avoidance of weeds. Thinning treatments resulted in significant increases in understory light, although the increase in light did not seem to affect the understory. There were notable declines in plant cover at both Sheep Creek (e.g. pinegrass, birch-leaved spiraea, shrub and bryophyte) and Wolf Creek (e.g. pinegrass, sedge and total herb). There were also declines in bunchgrass and forb production at Wolf Creek. These declines are likely due to mechanical disturbance and severe drought (drought occurred over both monitoring seasons) at both sites. Despite disturbance and drought at Sheep Creek, the plant community tended to respond more positively at greater intensities of thinning. Positive responses in understory characteristics such as species diversity and plant cover were observed with greater reductions in tree stem density (stems/ha) and stand volume (m^3/ha), suggesting that the plant community may be responding to increased availability of belowground resources rather than light. This trend is consistent with other studies that have documented the importance of belowground resources relative to light (Della-Bianca and Dils 1960, Riegel et al. 1992, Feeney et al. 1998, Kaye and Hart 1998). The positive changes associated with greater thinning suggest this plant community is poised for recovery, although continued monitoring is needed to verify this. It is also important to note the dry conditions experienced during the time this research was undertaken. Lack of moisture may have both increased the impact of mechanical disturbance, as well as inhibited recovery of the understory. As a result, it is recommended that further research be done examining the role of growing conditions on thinning activities.

It is worth noting that despite widespread drought and the severity of mechanical disturbance, that bunchgrass density increased over the two year post-thinning period at Sheep Creek. Although these species are susceptible to disturbance, it is unlikely that there was a significant increase in recruitment over the 2 years, but rather that suppressed plants were released enabling the plants to be detected. Notably, this change in density coincided with a negative association with the change in light. Thus, the drought-tolerance of bunchgrasses may have enabled this group of species to respond positively unlike other vegetation (e.g. pinegrass).

To prevent the loss of production and plant cover in the short-term, restoration thinning should aim to minimize damage to the understory plant community. This might be done by minimizing the number of roads, choosing low-impact machinery and ensuring post-thinning grazing does not impede recovery. In areas where the plant community has been degraded to the point of non-recovery, more intensive revegetation methods are needed. The BC Ministry of Forests and Ministry of Water, Land and Air Protection is currently the only agency known to be formally investigating the use of plugs to restore degraded range. Provincial land management agencies are currently examining this option because seeding semi-arid rangeland has been largely unsuccessful. In this context, the use of plugs were examined for restoring severely ingrown grasslands and open forests (Chapter #4). Two species were used (*S. richardsonii* and *P. spicatum*) at 2 different planting times (spring and fall) and 2 levels of pinegrass (pinegrass removal and no removal), with plugs monitored for both survival and growth. Choice of species at all blocks had a significant impact on survival. Overall, *S. richardsonii* had poorer survival than *P. spicatum*. This contradicts findings by the Invermere Forest District that found *P. spicatum* survival to be lower than *S. richardsonii*. This contradictory result is likely the result of this project assessing survival over two different seasons of planting instead of only the spring.

There was a strong species by season interaction indicating *P. spicatum* survival was higher in the fall and *S. richardsonii* survival greater in spring. As a result, the biology of these two species necessitates two different planting times in order to optimize survival. *P. spicatum* needs a long growing season and therefore initiates growth very early in the spring. In contrast, *S. richardsonii* is a borderline warm season species that needs warmer temperatures to initiate growth.

Suprisingly, season had the exact opposite effect on subsequent tiller development among surviving plugs, with *S. richardsonii* outperforming *P. spicatum* in the fall and *P. spicatum* outperforming *S. richardsonii* in the spring. These results appear to be a manifestation of 'survival of the fittest', with superior plugs surviving adverse planting conditions. For planning optimal planting times, survival versus optimal growth trade-offs need to be considered.

Results of the plug study have several implications for management. First, *S. richardsonii* and *P. spicatum* plugs appear to be suitable candidates for use in restoring degraded range. However, choice of species, season of planting and competition from surrounding species will all play a significant role in plug survival and growth. Second, the effect of excessive pinegrass competition may have implications for more extensive restoration projects (e.g. Chapter #3). The cover and production of pinegrass in response to restoration activities should be monitored to ensure pinegrass is not affecting the growth and survival of more desirable species, including native bunchgrasses. Additional information is needed, however, on the specific conditions (e.g. threshold abundance) where pinegrass may be a limitation to plug establishment, thereby necessitating its control.

The positive results of the plug experiment also justify larger-scale trials with a greater number of species and greater replication among sites, seasons, and years. Increased replication is clearly needed to assess the variability in plug responses to site (e.g. soil) and growing conditions (e.g. precipitation). A cost-benefit analysis should be included with a larger-scale project, particularly given that the use of plugs for restoration is labour-intensive and expensive. It is also important to note that despite the heavy ingrowth and intense mechanical disturbance within the 2 thinned sites, the apparent slow recovery of natural bunchgrasses suggests that the reduction in understory species may not have proceeded beyond the reversible point. Thus, the use of plugs should only be considered for stands where there is a certainty that the natural bunchgrass community has been extirpated or reduced beyond an unrecoverable threshold.

The restoration programs initiated by the BC Ministry of Forests are intended to create a mosaic of habitats intended to mimic the historical landscape under natural conditions when fire was an integral part of the ecosystem. Restoration monitoring is an essential component of this process. Key restoration response variables need to be identified and monitored on a long-term basis to ensure goals of restoration are being met. Part of this process is to ensure restoration objectives/goals are clearly stated so that clear ecosystem monitoring objectives can be derived. Long-term financial and institutional support is also necessary for this restoration approach to succeed. Through a series of adaptive management projects it is hoped that the conditions of forest ingrowth and encroachment can be alleviated, while simultaneously learning about ecosystem structure and function. A 'healthy' or desirable ecosystem state must be derived from an understanding of ecosystem dynamics. However, only continual monitoring and research can increase this understanding and lead to the eventual restoration of ecosystems.

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Table 5.1 Restoration targets[1] for various habitat components on crown land in the East Kootenay Trench of BC at the end of 30 years (2030).

Habitat component	**Current distribution (% of Trench)**	**Final distribution target (% and ha of Trench)**	**Tree density target (stems/ha)**
Shrubland	5 %	5 % (12,500 ha)	0
Grassland	10 %	23 % (57,500 ha)	? 75
Open forest	Open & managed forest is 85 % combined	31 % (77,500 ha)	76 – 400
Managed forest		41 % (102,500 ha)	400 – 5,000

[1] Targets are achieved within the Crown NDT4 land base at the forest district level (Machmer et al. 2002).

Appendix 1. List of understory characteristics at Sheep and Wolf Creek for the first and third year of sampling (1999&2001).

	Sheep Creek (IDF)		Wolf Creek (PP)	
Variable	**1999**	**2001**	**1999**	**2001**
Bunchgrass Cover[1] (%)	1.62	2.38	7.22	5.35
Pinegrass Cover (%)	9.93	7.65	16.67	9.81
Shrub Cover (%)	14.69	4.86	7.07	2.59
Carex Cover[1] (%)	0.61	1.45	4.9	2.51
Forb Cover (%)	8.38	7.12	8.72	6.28
Exotics species cover[2]	0	0.36	0	0.008
Bunchgrass production (kg/ha)	n/a	6.2	37.8	14.5
Pinegrass production (kg/ha)	n/a	13.6	23.8	9.5
Shrub production (kg/ha)	n/a	8.5	10.7	13.3
Carex production (kg/ha)	n/a	1.4	8.1	8.6
Forb production	n/a	15.6	36.7	10.4

[1]Native bunchgrasses considered historically common as listed on pp.51.
[2]Exotics include noxious and nusiance weeds as listed by the B.C. MoF/MoAFF Noxious/Nuisance Weed List.

Appendix 2. % cover and frequency of all species at Sheep and Wolf Creek for all 3 years of sampling.

Sheep Creek	1999		2000		2001	
	Cov	Freq	Cov	Freq	Cov	Freq
Grass / Grasslike						
Calamagrostis rubescens	10.0%	64.5%	10.8%	69.3%	7.8%	58.4%
Festuca campestris	0.9%	10.0%	1.3%	11.3%	1.1%	10.9%
Carex concinnoides	0.0%	0.0%	1.4%	16.7%	1.6%	15.3%
Stipa richardsonii	0.0%	0.0%	0.3%	2.7%	0.7%	5.3%
Elymus spicatus	0.1%	2.1%	0.5%	5.0%	0.3%	3.8%
Poa compressa	0.0%	0.0%	0.0%	0.0%	0.9%	6.9%
Carex sp.	0.6%	8.3%	0.0%	0.0%	0.0%	0.0%
Koeleria macrantha	0.5%	4.5%	0.1%	1.0%	0.1%	0.9%
Festuca sp.	0.0%	0.0%	0.1%	1.0%	0.2%	1.3%
Festuca idahoensis	0.1%	1.0%	0.0%	0.0%	0.1%	1.6%
Elymus glaucus	0.0%	0.0%	0.0%	0.0%	0.1%	1.3%
Stipa occidentalis	0.1%	1.7%	0.0%	0.0%	0.0%	0.3%
Stipa Sp.	0.0%	0.0%	0.0%	0.3%	0.0%	0.0%
Forb						
Balsamorhiza sagittata	1.1%	8.3%	1.5%	9.7%	0.7%	8.1%
Antennaria neglecta	1.0%	7.9%	0.6%	6.0%	0.6%	4.7%
Apocynum androsaemifolium	0.4%	6.9%	0.7%	9.7%	1.0%	10.6%
Hieracium scouleri	0.6%	7.2%	0.7%	10.0%	0.5%	5.6%
Aster foliaceus	0.1%	0.3%	0.6%	10.3%	0.9%	10.9%
Campanula rotundifolia	0.0%	0.0%	0.2%	4.7%	1.2%	15.0%
Arnica cordifolia	0.9%	5.9%	0.3%	5.0%	0.0%	0.0%
Lithospermum ruderale	0.5%	1.4%	0.2%	1.7%	0.4%	3.4%
Fragaria virginiana	0.3%	7.6%	0.3%	8.3%	0.3%	9.4%
Crepis atrabarba	0.0%	1.4%	0.5%	12.3%	0.3%	4.4%
Hedysarum sulphurescens	0.3%	3.4%	0.2%	3.0%	0.3%	4.1%
Achillea millefolium	0.3%	8.6%	0.2%	5.3%	0.3%	4.4%
Viola adunca	0.4%	9.7%	0.3%	6.7%	0.0%	0.0%
Collinsia parviflora	0.1%	3.8%	0.5%	5.3%	0.1%	1.3%
Cerastium arvense	0.1%	1.0%	0.4%	4.3%	0.2%	4.1%
Solidago spathulata	0.3%	2.8%	0.2%	0.7%	0.1%	1.3%
Taraxacum officinale	0.1%	4.5%	0.2%	3.0%	0.3%	4.4%
Calochortus apiculatus	0.0%	0.0%	0.5%	18.3%	0.1%	2.5%
Astragalus miser	0.3%	2.4%	0.1%	1.3%	0.1%	1.6%
Agoseris glauca	0.1%	2.1%	0.3%	5.3%	0.1%	2.5%
Penstemon sp.	0.4%	7.2%	0.0%	0.0%	0.0%	0.0%
Penstemon confertus	0.0%	0.0%	0.3%	3.3%	0.1%	4.1%
Aster conspicuus	0.4%	7.6%	0.0%	0.0%	0.0%	0.0%
Allium cernuum	0.2%	7.6%	0.1%	5.3%	0.0%	1.3%
Antennaria microphylla	0.1%	2.8%	0.1%	2.0%	0.1%	1.6%
Elymus repens	0.0%	0.0%	0.0%	0.0%	0.2%	0.9%
Hieracium umbellatum	0.1%	2.4%	0.1%	2.3%	0.0%	0.6%
Senecio streptanthifolius	0.2%	4.5%	0.0%	0.0%	0.0%	0.0%
Lomatium triternatum	0.0%	1.0%	0.1%	3.3%	0.0%	0.0%

Cirsium arvense	0.0%	0.0%	0.0%	0.0%	0.1%	0.6%
Calochortus sp.	0.1%	4.5%	0.0%	0.0%	0.0%	0.0%
Castilleja thompsonii	0.0%	0.0%	0.0%	0.7%	0.1%	0.9%
Aster sp.	0.1%	1.4%	0.0%	0.0%	0.0%	0.0%
Zigadenus elegans	0.0%	0.0%	0.0%	1.0%	0.0%	1.9%
Phlox caespitosa	0.0%	0.7%	0.0%	1.0%	0.0%	0.6%
Medicago lupulina	0.0%	0.0%	0.0%	0.0%	0.1%	0.6%
Senecio canus	0.0%	0.0%	0.1%	0.3%	0.0%	0.0%
Polygonum douglasii	0.0%	0.0%	0.0%	1.7%	0.0%	0.0%
Stellaria nitens	0.0%	0.0%	0.0%	0.0%	0.0%	0.9%
Antennaria umbrinella	0.0%	0.0%	0.0%	0.7%	0.0%	0.0%
Arabis holboellii	0.0%	0.0%	0.0%	0.0%	0.0%	0.6%
Heterotheca villosa	0.0%	0.0%	0.0%	0.0%	0.0%	0.6%
Potentilla sp.	0.0%	0.3%	0.0%	0.0%	0.0%	0.0%
Zygadenus sp.	0.0%	0.3%	0.0%	0.0%	0.0%	0.0%
Anemone patens	0.0%	0.0%	0.0%	0.3%	0.0%	0.0%
Anemone multifida	0.0%	0.0%	0.0%	0.0%	0.0%	0.3%
Dwarf Shrub						
Arctostaphylos uva-ursi	4.3%	33.8%	3.2%	28.3%	2.5%	24.1%
Shrub / Tree						
Spiraea betulifolia	8.3%	60.7%	5.6%	60.0%	3.6%	51.9%
Amelanchier alnifolia	3.6%	44.5%	2.5%	43.0%	2.8%	31.6%
Pseudotsuga menziesii	2.9%	35.9%	2.3%	28.0%	1.5%	18.8%
Shepherdia canadensis	2.4%	12.8%	2.0%	8.7%	1.0%	5.6%
Purshia tridentata	1.4%	12.4%	1.5%	12.0%	0.8%	9.7%
Rosa acicularis	0.1%	1.7%	0.2%	4.0%	0.4%	8.1%
Mahonia aquifolium	0.1%	2.4%	0.1%	2.3%	0.1%	2.2%
Rosa sp.	0.1%	5.2%	0.0%	0.0%	0.0%	0.0%
Juniperus communis	0.1%	1.4%	0.0%	0.7%	0.0%	0.3%
Symphoricarpos occidentalis	0.0%	0.7%	0.0%	0.7%	0.1%	2.2%
Juniperus scopulorum	0.0%	0.0%	0.1%	1.0%	0.0%	0.3%
Pinus ponderosa	0.0%	1.0%	0.0%	1.0%	0.0%	0.6%
Pinus contorta	0.0%	1.0%	0.0%	0.3%	0.0%	0.6%
Rosa gymnocarpa	0.0%	0.0%	0.0%	0.3%	0.0%	0.0%
Other						
Litter	65.6%	96.9%	70.0%	99.0%	70.0%	99.4%
Bryophytes	14.7%	55.5%	9.8%	34.0%	7.6%	28.8%
Dead Wood	6.1%	26.9%	8.7%	43.3%	9.7%	46.9%
Soil	0.9%	6.6%	11.1%	33.3%	8.2%	32.8%
Rock	0.5%	5.5%	1.2%	11.3%	1.1%	11.3%
Live Wood	0.3%	0.7%	0.3%	1.7%	0.1%	0.6%
Unknown sp._1	0.0%	0.0%	0.0%	1.0%	0.0%	0.0%

Premier Lake	1999		2001	
	Cov	Freq	Cov	Freq
Grass / Grasslike				
Calamagrostis rubescens	15.4%	59.0%	9.7%	58.3%
Stipa richardsonii	4.5%	20.6%	4.5%	25.2%
Festuca campestris	2.8%	25.8%	1.9%	23.1%
Carex sp.	3.8%	31.0%	0.0%	0.0%
Carex concinnoides	0.0%	0.0%	2.3%	31.9%
Koeleria macrantha	0.7%	11.6%	0.9%	16.2%
Elymus spicatus	0.2%	3.5%	0.1%	2.9%
Stipa comata	0.1%	0.6%	0.2%	2.4%
Stipa occidentalis	0.2%	6.8%	0.0%	0.0%
Festuca idahoensis	0.0%	1.6%	0.0%	0.7%
Elymus trachycaulus	0.0%	0.0%	0.1%	1.7%
Poa compressa	0.0%	0.0%	0.0%	1.7%
Forb				
Agoseris glauca	1.4%	19.4%	0.9%	21.4%
Astragalus miser	1.4%	19.7%	0.6%	11.0%
Solidago spathulata	0.8%	12.3%	0.8%	10.5%
Fragaria virginiana	0.9%	19.7%	0.6%	13.8%
Penstemon sp.	1.0%	23.2%	0.0%	0.0%
Penstemon confertus	0.0%	0.0%	0.9%	25.7%
Antennaria microphylla	0.5%	8.7%	0.3%	7.9%
Phlox caespitosa	0.4%	10.0%	0.4%	8.1%
Viola adunca	0.3%	10.6%	0.2%	8.6%
Potentilla sp.	0.5%	4.5%	0.0%	0.0%
Achillea millefolium	0.3%	10.3%	0.2%	6.0%
Aster foliaceus	0.1%	2.6%	0.3%	4.8%
Geum triflorum	0.3%	3.5%	0.1%	2.4%
Allium cernuum	0.3%	8.4%	0.1%	3.3%
Taraxacum officinale	0.2%	6.1%	0.2%	6.7%
Crepis atrabarba	0.0%	1.6%	0.1%	2.1%
Lithospermum ruderale	0.1%	1.0%	0.1%	0.7%
Anemone patens	0.2%	2.9%	0.0%	0.0%
Campanula rotundifolia	0.1%	2.9%	0.1%	1.7%
Antennaria unknown sp.	0.1%	0.6%	0.0%	0.0%
Zygadenus sp.	0.1%	4.2%	0.0%	0.0%
Anemone multifida	0.0%	1.6%	0.1%	2.4%
Gaillardia aristata	0.1%	1.6%	0.0%	0.5%
Balsamorhiza sagittata	0.0%	0.3%	0.1%	0.5%
Lotus corniculatus	0.1%	1.3%	0.0%	0.0%
Castilleja thompsonii	0.1%	1.0%	0.0%	0.2%
Lomatium triternatum	0.0%	1.9%	0.0%	0.5%
Hedysarum sulphurescens	0.0%	0.0%	0.1%	1.0%
Stellaria nitens	0.0%	0.0%	0.1%	1.0%
Collinsia parviflora	0.0%	1.6%	0.0%	0.5%
Erigeron pumilus	0.0%	0.0%	0.0%	1.9%
Oxytropis sericea	0.0%	0.0%	0.0%	1.7%
Hieracium scouleri	0.0%	0.0%	0.0%	0.2%
Aster conspicuus	0.0%	1.3%	0.0%	0.0%

Heuchera cylindrica	0.0%	0.6%	0.0%	0.5%
Tragopogon dubius	0.0%	0.6%	0.0%	0.5%
Arabis holboellii	0.0%	0.3%	0.0%	0.7%
Epilobium angustifolium	0.0%	0.6%	0.0%	0.2%
Antennaria neglecta	0.0%	0.3%	0.0%	0.5%
Hieracium albiflorum	0.0%	0.0%	0.0%	0.7%
Calochortus sp.	0.0%	0.6%	0.0%	0.0%
Oxytropis monticola	0.0%	0.6%	0.0%	0.0%
Vicia americana	0.0%	0.6%	0.0%	0.0%
Lathyrus nevadensis	0.0%	0.3%	0.0%	0.0%
Microsteris gracilis	0.0%	0.3%	0.0%	0.0%
Senecio streptanthifolius	0.0%	0.3%	0.0%	0.0%
Cirsium arvense	0.0%	0.0%	0.0%	0.2%
Dwarf Shrub				
Arctostaphylos uva-ursi	20.6%	70.6%	10.9%	51.7%
Shrub / Tree				
Amelanchier alnifolia	5.3%	68.1%	3.2%	61.2%
Purshia tridentata	2.5%	12.6%	1.4%	9.8%
Pseudotsuga menziesii	1.5%	25.5%	1.4%	13.8%
Rosa acicularis	0.4%	1.3%	0.2%	5.2%
Shepherdia canadensis	0.1%	1.0%	0.3%	1.2%
Pinus contorta	0.2%	7.4%	0.1%	3.1%
Rosa sp.	0.3%	10.0%	0.0%	0.0%
Symphoricarpos occidentalis	0.0%	0.6%	0.1%	1.7%
Pinus ponderosa	0.0%	1.0%	0.1%	2.9%
Rosa gymnocarpa	0.0%	0.0%	0.0%	1.4%
Juniperus communis	0.0%	0.3%	0.0%	0.0%
Rosa woodsii	0.0%	0.0%	0.0%	0.2%
Spiraea betulifolia	0.0%	0.0%	0.0%	0.2%
Other				
Litter	66.8%	99.0%	66.3%	98.1%
Dead Wood	5.8%	29.7%	10.4%	53.8%
Bryophytes	4.9%	19.4%	5.5%	19.5%
Soil	0.6%	4.8%	5.5%	26.0%
Live Wood	0.7%	3.5%	0.4%	1.7%
Rock	0.1%	1.0%	0.4%	3.8%
Unknown sp._1	0.0%	0.3%	0.0%	1.0%