EFFECTS OF BURNING AND THINNING ON SPECIES COMPOSITION AND FORAGE PRODUCTION IN BRITISH COLUMBIA GRASSLANDS

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ABSTRACT

The structural integrity of fire-dependent ecosystems, such as ponderosa pine (*Pinus ponderosa* Dougl.) and Interior Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) biogeoclimatic zones in Interior British Columbia (BC) is changing. The problems within these ecosystems include decreased rangeland area, reduced carrying capacity and loss of biodiversity due to tree encroachment and forest ingrowth caused mainly by fire suppression. The goal of this study was to determine the effect of burning and thinning on understory vegetation of grassland and forested sites. The burning experiment took place at Dew Drop (Tranquille Ecological Reserve) located 20 km northwest of Kamloops, BC. Thinning was done at two upper grassland sites near Cache Creek, BC; Coal Mine Pasture and Gladys Lake Pasture. Species evenness and values of the Shannon-Weiner Diversity Index (H’) were reduced (13 and 27%, respectively) within three years following burning ($P = 0.014$ and $P = 0.038$, respectively). Burning reduced canopy cover of shrubs on grassland sites ($P = 0.005$) and it reduced graminoid cover on forest sites ($P = 0.014$) immediately after the treatment (1999) but both functional groups had recovered by 2002. Litter depth and total canopy cover of plants were reduced in grasslands and forests immediately following burning (1999) but litter depth and canopy cover had recovered by 2002. Litter cover and litter depth beneath the tree canopy were reduced by burning ($P = 0.037$ and $P = 0.009$, respectively). Trends in the data indicate forb standing crop increased and total understory standing crop increased following burning in the grassland compared to the control. Graminoid standing crop was reduced 47% by burning in the forests ($P = 0.049$). Thinning reduced species richness in the first ($P = 0.033$) and fourth ($P = 0.030$) years, and H’ in the first year ($P = 0.037$) following the thinning at Coal Mine Pasture. Trends in the data suggest understory standing crop increased at Coal Mine and Gladys Lake Pastures following thinning. At both locations, thinning reduced litter depth. Therefore, burning and thinning kills trees, reduces fuel loads, and increases standing crop of the understory.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1.0 INTRODUCTION

Forest ingrowth is an increase in tree density in the lower canopy layers of previously open, or semi-open forests, while encroachment is the establishment of a significant number of tree seedlings on open grasslands (Ross 2000). Tree encroachment and forest ingrowth can have major influences on rangelands, affecting biodiversity, site productivity, carrying capacity for livestock and wildlife, soil development and stability, water quality and distribution, and recreational opportunities (Archer 1994). Possible reasons for the expansion of woody species include fire suppression, human disturbances, climatic variation, livestock grazing, and combinations of all of these factors (Bai et al. 2004). Shifts in grassland and forest vegetation can also affect carbon sequestration and greenhouse gas emissions (Archer 1994).

The cattle industry in British Columbia (BC) is reluctantly being forced to adapt to the reduction in grazing lands due to tree encroachment. Problems such as decreased rangeland area, reduction in carrying capacity, altered plant communities, deteriorated range condition caused by land misuses such as overgrazing, and losses of biodiversity are associated with forest ingrowth and tree encroachment (Ross 2000). Impeded cattle movement (Strang and Parminter 1980) and reduced access to forage are major concerns for ranchers. The abundance of understory shrubs and herbs are inversely correlated with tree ingrowth in Interior Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and ponderosa pine (*Pinus ponderosa* Dougl.) sites of BC (Page 2002). If the beef industry is to remain competitive and viable in BC, methods to control tree encroachment and restore understory vegetation must be found. The forests of Interior BC dominated by Interior Douglas-fir and ponderosa pine, and the grasslands were historically dependent on fires to restrict ingrowth and tree encroachment, respectively (Gayton 1996).

Prescribed burning and thinning have been used to control tree encroachment and forest ingrowth.

Fire-maintained stands benefit from surface fires that keep tree densities in balance with the site capability (Gayton 1996). However, fire suppression in forests allows fuel buildup, thus increasing the risk of intense wildfires (Covington and Sackett 1984, Daigle 1996, Gayton 1996). Uncontrolled wildfires after fuel buildup are usually destructive to understory vegetation, and increase soil and water erosion (Humphrey 1962). Controlled burns can be used to reduce wildfire damage, increase forage production, and thin tree stands with little damage to understory vegetation (Humphrey 1962).

Thinning, the removal of selected trees from a forest stand, can increase understory plant cover, species richness (Thomas et al. 1999) and biomass production (McConnell and Smith 1970). Removal of trees can increase the amount of light reaching understory (Thomas et al. 1999). In addition, eliminating some of the competition from trees can increase the availability of soil water and nutrients for plants (Thomas et al. 1999). Thinning decreases stand density, thus reducing the likelihood and destructiveness of a wildfire.

The effects of burning on ponderosa pine forests have been studied extensively, covering such issues as light interception and soil nitrogen (Moir 1966), organic matter and nutrients in woody debris (Covington and Sackett 1984), plant phenology (White et al. 1991), species composition (Johnson and Strang 1983), biomass production (Johnson and Strang 1983, Vose and White 1991, White et al. 1991), site productivity (Lindeburgh 1990), and shrub cover (Fraas et al. 1992). Studies on the effects of thinning include aboveground and belowground competition (Riegel et al. 1995), species composition (Uresk and Severson 1998, Thomas et al. 1999, Brockway et al. 2002), and understory biomass (McConnell and Smith 1970, Brockway et al. 2002).

Wikeem and Strang (1983) noted many researchers studied the effects of fire in the ponderosa pine zone, but little is known regarding the specific effects of fire on understory vegetation. Prior to 1983, Wikeem and Strang noted that the effects of fire on individual plant species, botanical composition, vigor and productivity as well as fuel and weather conditions were poorly understood. Vose and White (1991) also recognized the need for studies to focus directly on the effects of burning on individual plant species. More recent studies have identified the lack of research on understory vegetation responses to restoration treatments such as burning and thinning (Abella and Covington 2004).

Studying the effects of thinning and burning allows land managers to understand the consequences of these treatments and to develop site-specific management plans. Coal Mine and Gladys Lake Pastures used in the study have been grazed and ranchers expect that treatments such as thinning would increase the amount of palatable and nutritious forage. Management goals in an ecological reserve such as Dew Drop, are less concerned with forage production and more focused on maintaining the integrity and/or restoring the ecosystem to its natural state. Objectives of BC Parks at Dew Drop were to prevent a catastrophic wildfire. Prescribed burning at Dew Drop was intended to use a medium intensity burn to 'fire proof' the stand by removing pine needles that had accumulated, reducing woody debris and ladder fuels (small trees) which could carry the fire up into the tree canopy (Don Thompson, personal communication).

The objectives of this project were to determine the effects of burning and thinning on; 1) the control of trees that have encroached onto grasslands, and 2) species diversity and understory standing crop of understory vegetation in BC grasslands. Knowing more about the specific effects of burning and thinning on understory vegetation may help to maximize forage production in grasslands and maintain long-term sustainability of the grassland ecosystems.

2.0 LITERATURE REVIEW

2.1 General characteristics of British Columbia grasslands

2.1.1 Physical environment

Two factors control the climate of BC grasslands; 1) intermountain location results in a rainshadow effect, and 2) elevation (McLean 1970). The climate is mild continental (van Ryswyk et al. 1966). Precipitation falling during the growing season has little effect on plant growth because most effective soil moisture is supplied by melting snow (van Ryswyk et al. 1966). About one-quarter of the annual precipitation falls during August and September, a time of high evaporation (van Ryswyk et al. 1966). As elevation increases, air temperature decreases, evaporation decreases, and precipitation effectiveness increases (McLean and Marchand 1968). Solar radiation (particularly ultraviolet radiation), wind, and snow depth and duration increase with increasing elevation (Peet 2000). Another important factor in determining the climate in BC grasslands is the topographic–moisture gradient (Peet 2000), which controls vegetation distribution. South-facing slopes, for example, receive high amounts of incident solar radiation and tend to be warmer and drier than north-facing slopes and sheltered valley bottoms (Peet 2000).

Glaciation is responsible for the soils and valley formations of BC (van Ryswyk et al. 1966). Soils above 580 m in elevation tend to be fine and silty in texture, whereas soils below this elevation are generally cobbly and gravelly. Soil texture and depth affect plant community composition (Peet 2000). Trees tend to inhabit thin or rocky soils, whereas grasses and forbs usually inhabit fine-textured soils (Peet 2000). Soils change from Brown to Dark Brown, to Black Chernozems with elevation(van Ryswyk et al. 1966). As soils change with increasing elevation, the depth to the calcareous layer increases as do cation exchange capacity and bulk density. In general, soils in Interior

BC tend to have low amounts of sodium and potassium and high amounts of calcium and magnesium.

2.1.2 Biogeoclimatic classification

In 1975, BC Ministry of Forests began using a resource management system that incorporates climate, soil and vegetation data. Dr. V.J. Krajina and his students at the University of British Columbia developed this system to provide a framework for resource management (Pojar et al. 1991). The idea behind the classification system is to provide a "permanent", land-based, ecological classification system, which organizes knowledge of the structure, function and relationships of terrestrial ecosystems. There are 14 major Biogeoclimatic (BGC) zones in BC, 11 of which are used for grazing (Wikeem et al. 1993). The present project deals with the Bunchgrass, Ponderosa pine and Interior Douglas-fir Biogeoclimatic zones.

The Bunchgrass zone is one of the driest BGC zones in BC (McLean 1970). It is generally located on lower valley slopes between 300 and 1,000 m in elevation (Wikeem et al. 1993). The climate in this zone is semi-arid, characterized by hot, dry summers and moderately cold winters with annual precipitation less than 335 mm (Wikeem et al. 1993). In the sagebrush (*Artemisia tridentata* ssp. *wyomingensis* Beetle and Young) - bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith) association, soils are Rego Brown, calcareous or saline in nature, well-drained, and loam to silty loam in texture (McLean 1970).

Three grasslands, lower (bluebunch-big sagebrush), middle (bluebunch-bluegrass [*Poa sandbergii* Vasey]) and upper (bluebunch-fescue [*Festuca scabrella* ssp. *scabrella* Torr.] grasslands, can be identified according to elevation within the Bunchgrass zone (van Ryswyk et al. 1966). Brown soils and elevation between 345-610 m characterizes the lower grassland (McLean and Marchand 1968). Precipitation in the lower grassland ranges between 230-250 mm annually. The middle grassland lies between 610-825 m in the Dark Brown soil zone (van Ryswyk et al. 1966) with precipitation averaging 300 mm (McLean and Marchand 1968). Black soils are dominant in the upper grassland, at elevations between 825-975 m (van Ryswyk et al. 1966) and 280-330 mm precipitation are received each year (McLean and Marchand 1968).

The Ponderosa pine BGC zone occupies lower slopes and outwash terraces at elevations from 600 and 800 m (McLean 1970). The average annual precipitation in this BGC zone is 270-380 mm. Soils are developed on shallow bedrock and river sediments (McLean 1970) and are Brown Wooded or Dark Gray (McLean and Marchand 1968). The ponderosa pine-bluebunch wheatgrass association is located between 500-600 m in elevation, and has Dark Brown soils that are sandy to gravelly loam in texture with very low water-holding capacity (McLean 1970). In areas where bedrock is close to the soil surface, needle-and-thread (*Stipa comata* Trin. & Rupr.) dominates instead of bluebunch wheatgrass (McLean 1970). The ponderosa pine-Idaho fescue (*Festuca idahoensis* Elmer) association occurs on sites between 600 and 800 m in elevation (McLean 1970). A topoedaphic climax in these areas is Interior Douglas-fir and rough fescue. Heavily textured soils, eroded, or heavily grazed south- or west-facing slopes have large populations of balsam root (*Balsamorhiza sagittata* (Pursh) Nutt.) (McLean 1970).

The Interior Douglas-fir BGC zone is generally characterized by a cool climate and occurs between 300 and 1400 m in elevation (Wikeem et al. 1993). Precipitation in this BGC zone ranges from 300 to 750 mm annually (Wikeem et al. 1993). Bluebunch wheatgrass dominates at lower elevations, but in general, the zone is dominated by pinegrass (*Calamagrostis rubescens* Buckl.) (Wikeem et al. 1993). Soils in this BGC zone are typically made of colluvium and bedrock and they are generally Regosols, Eutric Brunisols, degraded Eutric Brunisols and Grey Luvisols (McLean 1970).

2.1.3 Natural disturbances and utilization of BC grasslands

Sousa (1984) defines a disturbance as "a discrete, punctuated killing, displacement, or damaging of one or more individuals (or colonies) that directly or indirectly creates an opportunity for new individuals (or colonies) to become established". He explains further that disturbances may be physical or biological processes. Examples of biological processes include predation, grazing (Sousa 1984) or insect outbreaks (Attiwil 1994). Physical disturbances, specific to forests, can include tree fall, wind, fire, and changes in land base of agriculture (Attiwill 1994). Characteristics of disturbance regimes include areal extent, magnitude (intensity and severity), frequency, predictability and turnover rate or rotation period (Sousa 1984).

Fire is the major disturbance of the forest/grassland zones of Interior BC. BC Ministry of Forests designed a classification system to recognize the frequency and intensity of fires within a specific region to allow sound ecological decisions and management decisions. Grassland, shrubland, and forested communities that normally experience frequent low-intensity fires are deemed 'NDT4' or Natural Disturbance Type IV (BC Ministry of Forests 1995). Management of NDT4 encourages characteristics such as moderately open, uneven stands, an understory free of trees and shrubs, with ground cover of grasses and forbs (Humphrey 1962). The sites in the present study are within NDT4. Periodic surface fires in Ponderosa pine and Interior Douglas-fir BGC zones consume woody fuels, rejuvenate most herb and shrub species while selecting against others, thin young stands, and raise the height of tree crowns. In addition, fires maintain species composition and forest stand structure, and regulate coarse woody debris loading (BC Ministry of Forests 1995). When there is little or no fuel accumulation, fires are relatively 'cool' and tend not to develop into crown fires (Humphrey 1962).

Nearly 85% of the land used for grazing in BC is crown land, and is administered by the BC Ministry of Forests (Wikeem et al. 1993). Use of these rangelands is primarily for cow-calf and yearling operations (Wikeem et al. 1993). About 60% of the total annual forage requirements are provided on pastures, or approximately one million animal unit months (AUMs) (Wikeem et al. 1993). BC grasslands are generally used for late fall and early spring grazing (van Ryswyk et al. 1966).

In the following, focus will be placed on each biogeoclimatic zones with regards to optimum grazing times and changes in indicator species after grazing. The Lower grasslands are grazed in late-fall, winter and early-spring (Wikeem et al. 1993). Decreasers on this range are bluebunch wheatgrass and in some soils, needle-and-thread. Increasers include big sagebrush, pusstytoes (*Antennaria spp.* Gaert.), needle-and-thread, rabbitbrush (*Chrysothamnus nauseosus* (Pall.) Britt.), sand dropseed (*Sporobolus* *cryptandrus* (Torr.) A. Gray), and Sandberg bluegrass. Invaders include tansy mustard (*Descurania sophia* L.), downy brome (*Bromus tectorum* L.), and Russian thistle (*Salsola kali* L.) (McLean and Marchand 1968). Forage production ranges from 400 to 900 kg ha^{-1} on range in excellent condition (McLean and Marchand 1968).

The Middle grasslands are used for spring and fall grazing. The growing season is late-March to early-July (McLean and Marchand 1968). Decreasers in the Middle grasslands are the same as the Lower grasslands. Increasers are pussytoes, Junegrass (*Koeleria cristata* L.), needle-and-thread, pasture sage (*Artemisia frigida* Willd.), rabbitbrush, sand dropseed, and Sandberg's bluegrass (McLean and Marchand 1968). Invaders include downy brome, dandelion (*Taraxacum officinale* Weber), six-weeks fescue (*Festuca octoflora* Walt.), and woolly plantain (*Plantago patagonica* Jacq.) (McLean and Marchand 1968). The Upper grasslands are best suited for spring (late May and June) and fall grazing (October). Decreasers include bluebunch wheatgrass, Idaho fescue, and rough fescue. Needle-and-thread, Junegrass, Kentucky bluegrass (*Poa pratensis* L.), pasture sage, pusstyoes, Sandberg's bluegrass, lupine (*Lupinus sericeus* Pursh), timber milk-vetch (*Astragalus miser* Dougl. ex. Hook.), and yarrow (*Achillea millefolium* L.) are increasers in the Upper grasslands (McLean and Marchand 1968). Invaders are downy brome, dandelion, mullein (*Verbascum thapsus* L.) and cut-leaved daisy (*Erigeron compositus* Pursh) (McLean and Marchand 1968).

Forests account for nearly 80% of the provincial-crown range resources of BC (Wikeem et al. 1993). The Ponderosa pine zone is usually grazed in early spring and late fall (Wikeem et al. 1993). Balsamroot, big sagebrush, pusstytoes, Junegrass, needle-and-thread, rabbitbrush, Sandberg's bluegrass, lupine and yarrow are increasers in the Ponderosa pine zone. Phlox (*Phlox gracilis* L.), downy brome, dandelion and wooly plantain are invaders (McLean and Marchand 1968).

The Interior Douglas-fir zone is usually grazed late spring, summer and early fall (Wikeem et al. 1993). The Interior Douglas-fir zone is the principal area where grazing occurs in BC (McLean 1970), and in open stands at lower elevations, bluebunch wheatgrass is the primary forage. However, pinegrass is generally the most dominant herbaceous species over most of this BGC zone (Wikeem et al. 1993).

An important consideration for differentiating grasslands and forest regions is the environmental factors that determine forage production. The interaction of soil water and canopy cover dictate forage yield in forested range (Dodd et al. 1972). As elevation increases, forage yield is no longer limited by water as much as by density of the tree canopy because light becomes the limiting factor in growth (Dodd et al. 1972). As tree density increases carrying capacity is reduced (McLean et al. 1971).

2.2 Vegetation dynamics and tree encroachment

2.2.1 Ecology of ecotones between forests and grasslands

The transitional area between grasslands and forests is an ecotone (Morris 1974). Ecotones tend to be richer and more diverse in species of plants and animals than either of the communities they separate because they contain species from both communities and may even have species unique to the ecotone itself (Morris 1974). The environment of the ecotone may be different from adjacent communities and it is not necessarily intermediate between the two. Old-growth Douglas-fir forests consistently had cooler soil and air temperatures, lower wind velocities, and short-wave radiation than in clearcut areas (Chen et al. 1993). Although edges had intermediate wind velocities and solar radiation, the most extreme daytime temperatures and relative humidities occurred in ecotones (Chen et al. 1993). Changes in the environment may be further complicated because boundaries between the plant communities may overlap (Cadenasso et al. 1997). Therefore, the functional edge may not be easily defined, but it may extend into the neighbouring plant communities (Cadenasso et al. 1997).

2.2.2 Tree encroachment and ecological impact

Forest encroachment generally precedes forest ingrowth, however in some forest situations, ingrowth may precede encroachment (Ross 2000). Tree encroachment begins with the forest edge extending into grasslands. One generation of trees encroaching is generally followed by ingrowth from a later generation (Ross 2000). The initial encroachment is accomplished by moisture conditions that favour establishment of tree seedling (Arno and Gruell 1983). The woody species that encroach grasslands are usually native and increase as a result of changes in abiotic or biotic environmental conditions that encourage woody species expansion (Van Auken 2000). Within Interior BC, Douglas-fir and ponderosa pine are the primary species involved with ingrowth and encroachment (Ross 2000).

Recent research indicates that landscape plays a critical role in forest ingrowth and tree encroachment in Interior BC (Bai et al. 2004). South-facing slopes at mid-elevations are more susceptible to tree encroachment than north-facing slopes (Bai et al 2004). Elevation and aspect affect evapo-transpiration, which in turn affects the degree of ingrowth and encroachment (Ross 2000). As the degree of slope increases, the probability of a vegetation shift from open to treed (5 to 15% tree canopy cover) grasslands decreases, which is most likely due to less favorable moisture regimes (Bai et al. 2004). As slopes erode and become less steep, particularly on south-facing slopes, treed grasslands shift to open forests. Northerly facing slopes most often contain closed forests (Bai et al. 2004). Gullies provide sites for tree establishment, however, slope is not required for encroachment (Ross 2000). Bragg and Hulbert (1976) found that woody species invasion was rapid on lowland, lower-slopes, and steep, rocky soils, and slowest on upland soils having the greatest clay content in Kansas. Variation in soil texture and distance to seed sources can contribute to the rate of woody species expansion (Bragg and Hulbert 1976). Management plans must incorporate topography, vegetation and tree regeneration and survival to target areas that are most susceptible to tree encroachment and to control trees.

Consumption by domestic animals, coupled with a reduction of grassland fires, may be the most critical factors in causing woody plant encroachment (Madany and West 1983, Savage and Swetnam 1990, Archer 1994, Van Auken 2000). Chronic herbivory can reduce aboveground biomass of herbaceous plants, thus reducing fuel loads and fire frequency (Bachelet et al. 2000, Van Auken 2000). Without these periodic fires, woody species have a growth advantage over grasses (Bachelet et al. 2000, Van Auken 2000). Other factors contributing to expansion of woody species include reduced competition from grasses, seed dispersal of woody plants by herbivores, and changes in rodent, lagomorph, and insect populations (Van Auken 2000).

Ponderosa pine seedlings were taller and had longer leaders following season-long grazing than without grazing; however, no differences in seedling survival were noted between grazing treatments (Ratcliff and Denton 1995). Contrasting studies suggest that high stocking rates may cause trees to be trampled and subsequent death in areas with new two- to three-year old lodgepole pine (*Pinus contorta* Loudon) seedlings (Newman and Powell 1997).

Arno and Gruell (1983) studied the fire history over the past century in southwestern Montana at the forest-grassland ecotone. Sagebrush cover increased, conifer forests thickened, and trees spread down slope into former grassland or sagebrush communities. "Pioneer, early, and mid-successional species are promoted by various combinations of higher light intensities and moisture and nutrients" (Riegel et al. 1995). The majority of increases in cover or density of plants was related to increased light in ponderosa pine forests (Riegel et al. 1995). Fifty-seven percent of the species that increased were rhizomatous (Riegel et al. 1995).

Woody plants can alter plant species composition, spatial distribution, and productivity in grassland/forest ecotones (Scholes and Archer 1997) through changes in microclimate, soil fertility and root competition (Gibbs et al. 1999). The influence of trees on grass growth depends on the growth characteristics of each; including canopy architecture or rooting pattern, photosynthetic pathway, life form (evergreen or deciduous), and resource requirements (Scholes and Archer 1997).

As ponderosa pines age, their bark becomes thicker, lower branches are shed, and their needles drop to the ground (DeBano et al. 1998). These growth characteristics suppress the growth of herbaceous vegetation, which alters the composition of the fuels and therefore changes fire behaviour (DeBano et al. 1998). Litter from ponderosa pine tends to decompose slowly because of high content of resins, lignins, and other organic compounds that are generally resistant to chemical breakdown (Moir 1966). With further growth, ponderosa pine cast increasing shade on the site and can completely cover the mineral soil with needles, thus suppressing tree regeneration (Scholes and Archer 1997, DeBano et al. 1998). Regeneration of vegetation therefore only occurs following fire, disease, or insect pest infestations (DeBano et al. 1998). Ponderosa pine trees that are at least six years of age resist surface fires (Arno 1980). The timing of burning is critical in this ecosystem. When fires occur before enough fuel has accumulated, the fuel may not sustain a fire. If fires occur when too much fuel is present, the fire may be high-intensity, killing many trees (DeBano et al. 1998). Douglas-fir trees require about 40 years to develop a thick and corky outer bark allowing it to survive light to moderate fires (Arno and Gruell 1983).

The presence and structure of the overstory canopy directly influence the amount of light reaching the understory; productivity of understory vegetation and vegetation structure are the result of competition for light (Dodd et al. 1972, Peltzer et al. 1998, Gibbs et al. 1999, Harrington and Edwards 1999, Aubin et al. 2000). Increasing forest ingrowth and decreasing light are inversely associated with the abundance of understory species in ponderosa pine and Interior Douglas-fir forests (Page 2002). As trees encroach, the initial, first wave of seedlings likely makes the site more suitable for the second and third generations by conserving soil water through increased shading (Ross 2000). These 'waves' are dependent on favourable soil water and a copious seed crop; the waves may last 10 to 15 years (Ross 2000). Periods of drought usually end these pulses of tree establishment.

Woody plants affect understory vegetation by competing for resources or altering resources. Competition for resources such as soil water and light may inhibit herbaceous vegetation. For example, tree roots that lie at the same depth as those of the grasses compete directly for soil water (Humphrey 1962). Vertical distribution of roots between grasses and trees may also influence species composition. Gibbs et al. (1999) studied the relationship between tree cover of stringybark (*Eucalyptus laevopinea* R. Baker) and grass dominance in the New England region of Australia. Certain grasses had an advantage over one another by inhabiting different zones in the soil, avoiding competition with trees for resources. Grasses that utilize areas not inhabited by tree roots grow better than grasses that occupy the same space as trees (Gibbs et al. 1999). Deep-rooted trees are able to obtain nutrients beyond the canopy, thus they have an advantage over grasses beneath the canopy (Gibbs et al. 1999). When trees access nutrients beyond the canopy, grasses may benefit because they can utilize nutrients that accumulate in the soil beneath the canopy (Gibbs et al. 1999). Tree canopies provide shade and produce additional litter, which may not affect grass production (Humphrey 1962). The effects of the physical presence of trees on grasses may change with tree age, size or density (Scholes and Archer 1997). This competition for resources along with changes in the local environment can allow woody plants to increase to a point where the site can no longer support the density of grasses (Humphrey 1962).

Well-established grasses can limit woody species expansion onto grasslands. The mass of mesquite (*Prosopsis glandulosa* Torr.), a deep-rooted, woody heliophyte, was reduced by belowground interference from grasses (Van Auken and Bush 1997). Cool season grass species, such as *Oryzopsis, Poa* and *Stipa,* exert the most direct competition to ponderosa pine seedlings because they are at their peak physiological activity at the time when seedlings of the tree emerge (Mandany and West 1983). This critical time tends to correspond with limited precipitation, and perennial grasses may use the available soil water that is vital for seedling survival (Mandany and West 1983). While grazing may reduce grass biomass, the situation can be reversed. Grazing reduces root growth, which in turn reduces grass competition with woody plants (Schuster 1964). Following grazing, the roots of grasses in the ponderosa pine-bunchgrass range in Colorado occupied less space, and thus growth of roots was limited because nutrients and water were less available (Schuster 1964).

Grasses may affect the establishment of woody plants. Recruitment of woody plants can be directly affected by grasses through competition for light, water or nutrients, or indirectly by providing fine fuel that influence fire frequency and intensity (Scholes and Archer 1997). Perhaps a combination of these factors may determine establishment of tree seedlings. Tree seedlings established at sites with high and low amounts of litter in Interior BC (Ross 2000). Periods of available soil water may encourage tree recruitment because competition from grasses is minimal (Scholes and Archer 1997). Grass litter may encourage seedling establishment during drought by maintaining greater soil water (Ross 2000).

Grasslands in good health promote fire due to the accumulation of litter, which in turn reduces seedling growth and survival of trees because fire can kill seedlings and consume live foliage (Bachelet et al. 2000). Tree and shrub seedlings must compete with understory vegetation in the initial 5 to 10 years following fire if they are to establish (DeBano et al. 1998). Subsequently, dense stands of young trees develop and without recurrent fires, fuel loads increase (Peet 2000). Catastrophic fires may then kill many of the trees in the forest (Peet 2000). The combination of grazing and fire suppression favours the establishment of trees such as ponderosa pine (Mast et al 1998). Grassland seedbeds promote germination of Douglas-fir and ponderosa pine seeds over that of the needle-covered forest (Bai et al. 2000). The physical and chemical characteristics of grassland seedbeds favor the germination of these trees, if soil water is sufficient (Bai et al. 2000).

2.2.3 Forest ingrowth and tree encroachment in BC

Disturbances constantly affect and change forests. Disturbances such as fire, wind, insects, disease, ungulate browsing, extreme weather and human activities challenge the ability of the landscape to resist and recover (Peet 2000). Disturbances determine that vegetation is not uniform and stable, instead, vegetation constantly changes and borders separating different vegetation types are re-defined by these disturbances (Peet 2000). Forest ingrowth and encroachment studies in Interior BC demonstrated that open grassland area decreased while grassland containing trees and closed forests increased (Ross 200, Bai et al. 2004); open grassland was reduced 18% to 51%, with more than 1800 ha of open grassland being lost.

Fire suppression is the major factor allowing woody plant expansion in Interior BC. Tree establishment in areas such as ponderosa pine forests is a combination of a good seed year (Peet 2000), ample spring and summer precipitation (Mast et al. 1997, Peet 2000), and the absence of fire (Peet 2000). These conditions can lead to forest ingrowth (Peet 2000) and/or tree encroachment onto grasslands (Mast et al. 1997). Changes in grassland vegetation with tree encroachment are not necessarily uniform or gradual, but they may be relatively sharp between vegetation types (Van Ryswyk et al. 1966).

2.3 Control, restoration and management

Prescribed burning, mechanical treatments such as thinning, or a combination of treatments are used to control tree encroachment, forest ingrowth, and to restore grassland ecosystems in Interior BC (Daigle 1996). Management on crown lands has focused on integrating forage and livestock management, but other issues such as timber, wildlife, watershed quality and capacity, and recreation are also of interest (McLean et al. 1993). Management focus has been redirected from timber and wild ungulates towards environmental issues and conservation in recent years (McLean et al. 1993).

2.3.1 The ecological effects of prescribed burning

Increased mining, logging, settlement, and railway activities together with the drought during the 1930s created many large and damaging fires in BC (Gayton 1996). These activities lead to organized fire suppression, until prescribed burning was introduced in BC in the 1940s (Gayton 1996). Forest managers use prescribed burning to reduce the incidence and damage of wildfires (Humphrey 1962, Convington and Sackett 1984). Prescribed burning, whole-tree thinning and thinning following prescribed burning can reduce fire severity and crown fires in ponderosa pine forests (Pollet and Omi 2002). Before the twentieth century the historic fire return interval was 6 to 12 years in ponderosa pine, Douglas-fir and grassland ecosystems of BC (Gayton 1996). The effects of fire are less when the fire interval is shorter than return intervals are longer (Gayton 1996). Fuel load, tree size, tree arrangement, and chemical composition of plant materials are major factors determining the effects of fires on forest communities (Fonda et al. 1998). Vegetation characteristics affect the fire and the inter-fire intervals that regulate composition, structure, and quantity of the living plants and dead fuels (Sousa 1984).

Deciding whether prescribed burning is the best course of action requires serious consideration. The Canadian Forest Service uses a specific procedure to determine whether to use prescribed burning including; 1) evaluation of pretreatment vegetation and ecosystem; 2) determination of the management objective and target species; 3) evaluation of the fire environment of the proposed treatment area; 4) prediction of the occurrence, abundance, and growth rates of plants following fire, and alternative management treatments, and; 5) choice of appropriate treatment (Feller 1996).

Nitrogen is the main nutrient limiting growth of plants in the Pacific Northwest (Edmonds et al. 1989). Because conifers are efficient in nitrogen resorption, the annual input of nitrogen required for growth of a mature forest is small, and much of the nitrogen is stored in aboveground materials. Therefore, natural disturbances such as fire cause large nitrogen losses (Thompson et al. 2000, Choromanska and DeLuca 2001). Exposure of mineral soil and blackening of the soil surface can increase soil temperature (Lindeburgh 1990, Gayton 1996). Thinning and prescribed burning along with litter addition and complete removal of trees and forest floor materials as treatments were used to determine the response of soil respiration in ponderosa pine-bunchgrass ecosystems (Kaye and Hart 1998). Soil respiration rates do not automatically decrease after fire because of increased soil temperature (Kaye and Hart 1998). Fires destroy the surface layer of duff (Humphrey 1962) which can lead to increased loss of water. This water loss affects soil surfaces by clogging the A horizon with fine particles from muddy water, creating a hard, compacted layer that reduces water infiltration (Humphrey 1962, DeBano 2000). The degree to which water percolation is affected depends on the fire intensity. For example, hot wildfires will completely burn the duff, which in turn exposures mineral soil to the environment. Light, controlled burns, on the other hand, can have negligible effects on water percolation (Humphrey 1962). The total nutrient pool can be reduced via atmospheric losses, leaching, and erosion (Lindeburgh 1990). However, short-term availability of nutrients may be increased after burning by rapid mineralization (Lindeburgh 1990) and accelerated nutrient mobilization (Covington and Sackett 1984).

Controlled burning usually kills few perennial grasses and forbs, but woody species are damaged (Humphrey 1962). Regrowth of plants following controlled burning is usually rapid and yield can exceed that of pre-burn conditions (Humphrey 1962). Fire severity can play major roles in determining successional pathways (Feller 1996). For example, severe fires can reduce the number of plant propagules on the forest floor and encourage colonizer species rather than the plants present before burning

(Feller 1996). Regrowth of plants following severe wildfires is slow. Burning may also encourage tree regeneration by increasing available microsites for germination (Gayton 1996). Plant morphology also determines the response of vegetation to burning. Tree death is common for ponderosa pine and Douglas-fir with < 5cm diameter at breast height (DBH) (Johnson and Strang 1980).

Seed production, stand productivity, population growth, and interactions among species can all be affected by fire and stand-structure (White et al. 1991). Individual plant response to fire is variable. For example, wildfire reduced grass cover by 40% and increased forb cover in foothills grassland dominated by rough fescue, Idaho fescue and bluebunch wheatgrass; bluebunch wheatgrass cover was unaffected by burning, possibly due to the small plant size and lower location of the perenating buds (Antos et al. 1983). In comparison, the basal area of bluebunch wheatgrass was reduced most after burning when compared to other species in grasslands of southern Interior BC (Johnson and Strang 1980). Cover of early maturing grasses such as Sandberg's bluegrass more than doubled following burning (Antos et al. 1983) and the cover of Junegrass had increased 14 months after burning (Johnson and Strang 1983). Perennial forbs such as yarrow and death camas (*Zigadenous venenosus* S. Wats), initially increased following the burn (2.5 to 5.4% and 0.3 to 0.9%, respectively), and then decreased to unburned levels (Antos et al. 1983). Cover of pusstyoes (*Antennaria microphylla* Rydb.) also increased 14 months after burning in bunchgrass/sagebrush grasslands (Johnson and Strang 1983). Fourteen months after burning, the cover of big sagebrush and gray rabbitbrush was reduced by 90% compared to an unburned control (Johnson and Strang 1983).

Environmental conditions partially determine the ability of a community to recover following fire and may outweigh burning impacts (Grilz and Romo 1994). For example, fire severity tends to be less in areas with more precipitation and recovery of vegetation is generally faster than in drier areas (Feller 1996). Water use was higher and volumetric water content of soil was consistently less than control areas following burning in *Paspalum quadrifarium* grassland in Florida (Saskalauskas et al. 2001). Burning Fescue prairie also reduced soil water and increased water stress in plants (Grilz and Romo 1994). Burning of *Festuca* and *Stipa-Agropyron* grasslands in spring or autumn delayed early season growth and peak green biomass in the first year following burning (Redmann et al. 1993). In the second year following burning, plant growth in burned plots and peak biomass was reached earlier than in unburned control areas (Redmann et al. 1993). Burning may also affect plant phenology. Flowering of Arizona fescue (*Festuca arizonica* Vasey) and Montana muhly (*Muhlenburgia montana* (Nutt.) Hitche) was substantially delayed the year after burning (White et al. 1991). Vegetative growth of plants was the primary priority in the first year following fire, while reproduction was the primary priority in the second (White et al. 1991).

Other consequences of severe fires can be seen with increased soil and water losses (Humphrey 1962), water repellency (Lindeburgh 1990, DeBano 2000), reduced bulk density of soil, and nutrient deficiencies (Lindeburgh 1990). Comparisons between controlled burns and wildfires in ponderosa pine and Douglas-fir forests in Montana revealed that potential mineralizable N, NH_4-N and NO_3-N were significantly greater after wildfire compared to controlled burning (Choromanska and DeLuca 2001). Twenty one months following wildfire, microbial biomass was reduced to 52 $\mu g\ g^{-1}$. In contrast, microbial biomass three months after prescribed burning remained constant at 100 $\mu g\ g^{-1}$ (Choromanska and DeLuca 2001).

2.3.2 The ecological impacts of thinning

Different approaches to thinning and the combination of burning and thinning are used to control forest ingrowth and tree encroachment. The selection of thinning techniques is based on the seral stage of vegetation and management goals. In young, even aged forests, partial cutting prescriptions are used to create a variety of patch types within the landscape (BC Ministry of Forests 1995). A combination of partial clear-cutting and sporadic smaller clearcuts is often used in areas that were typically more open in the past (BC Ministry of Forests 1995).

Thinning forests can increase the productivity of native forage species (Dodd et al. 1972) and improve ecosystem health. Ideally, removing the tree canopy will increase the amount of light reaching understory vegetation and consequently increase cover of understory plants and species richness (Thomas et al. 1999). Removal of the tree canopy can also cause soil temperatures to increase (Strickler and Edgerton 1976) and soil respiration to decrease, possibly because of root death (Streigl and Wickland 1998). In addition, reducing or eliminating competition from overstory trees can make soil water and mineral nutrients more available for forages (Peltzer et al. 1998), and allow colonization of bare soil by herbaceous plants (Smit and Rethman 2000). With increasing productivity of understory vegetation, however, competition for light and soil resources also increases (Peltzer et al. 1998). Removing small diameter trees from ponderosa pine forests reduced the severity of subsequent wildfires (Pollet and Omi 2002).

Disturbance brought on by thinning can increase the availability of new microsites for plant establishment and growth, possibly leading to increased species richness (Brockway et al. 2002). The disturbances associated with thinning can reduce understory vegetation cover, in particular, trampling and smothering of vegetation by cut trees (Thomas et al. 1999). Other research has demonstrated that timber harvesting and extensive site preparation can reduce the amount of surface, organic matter (Jurgensen et al. 1997). The detrimental effects of tree ingrowth must be weighed against the disturbances associated with thinning, because both can reduce the cover of many understory species (Page 2002). In particular, cover of herbs, pinegrass and bryophytes respond negatively to thinning, however, there is no indication that long-term benefits of thinning are outweighed by an initial decreases in cover and biomass of understory vegetation (Page 2002).

The effects of thinning on understory vegetation are site specific. Thinning in pinyon (*Pinus edulis* Englem.) - juniper (*Juniperus monosperma* (Engelm.) Sarg.) woodlands increased the cover of native grasses and to a lesser extent, forbs and shrubs (Brockway et al 2002). Plant species richness, standing crop, and litter cover increased the most where thinning was followed by complete removal or scattering of the tree material (Brockway et al 2002). After gap formation in a hemlock (*Tsuga canadensis* L.) forest, understory species exhibited various responses ranging from positive to negative, and some species responded immediately to gap creation while others responded later (Rankin and Tramer 2002). Growth of understory vegetation after thinning is attributed to increased light, therefore, the greatest response should occur during the first several years following thinning (Thomas et al. 1999). Factors affecting herbaceous vegetation responses to thinning and herbicide application in longleaf pine (*Pinus palustris* Mill.) plantations were ranked as follows: light > soil > water > herbicides > litterfall (Harrington and Edwards 1999).

2.3.3 Policy consideration in ecosystem management

Shinneman and Baker (1997) discussed "equilibrium" and "nonequilibirum" hypotheses for ecosystem management in ponderosa pine forests. The equilibrium hypothesis views ecosystems in which a dynamic balance exists between low-intensity fires and stable, long-lasting, old-growth communities. The alternate view of nonequilibrium involves unpredictable, random, and sometimes catastrophic disturbances, that create different dynamics, structures and components in ecosystems.

Management decisions implemented to control forest ingrowth and tree encroachment in ponderosa pine/bunchgrass and Douglas-fir ecosystems can take either the equilibrium or nonequilibrium approach. For instance, the equilibrium method would mimic historical fire return intervals. Historical fire return intervals of open grasslands were 7 to 10 years, while the ponderosa pine forests had fire return intervals of 10 to 12 years (Morrow 1993). A nonequilibrium approach would be to let wildfires, insects, and diseases go unchecked. Although nonequilibrium management approaches may be used to maintain landscape heterogeneity and native species biodiversity through natural disturbances, which vary in time and space (Shinneman and Baker 1997), this management is not safe and practical in all areas of Interior BC. Fire-maintained ecosystems need frequent fires to sustain an optimum state of health, diversity and stand structure that are resistant to crown fires (Daigle 1996). At this point, there has been insufficient practice of "equilibrium" management in fire-dependent ecosystems in Interior BC (Daigle 1996). Continually suppressing fires will cause stand structure to be characterized by higher density, stagnated growth or slow growing trees, increased mortality due to drought, and accumulation of dead, dry and ladder fuels (Daigle 1996). Ladder fuels are fuels that provide vertical continuity between surface fuels and crown fuels in a forest stand, which contributes starting crown fires (BC Ministry of Forests 2005a). Increased density of shade tolerant trees such as Douglas-fir, insects and diseases, deteriorated timber, water quality, and finally, an increased need to deploy fire suppression to protect surrounding people and structures are all consequences of fire suppression (Daigle 1996). Weber and Taylor (1992) reviewed the role of prescribed fires in reducing the hazards of wildfire for plant regeneration, enhancement of wildlife habitat, insect and disease control, and for forest ecosystem diversity conservation in Canada. They concluded that prescribed burning requires a 'vigorous public awareness campaign' to make people more aware of the dynamic nature of ecosystems and the ecological goals of management, but it is ecologically compatible and cost effective.

Treatments that control the expansion of woody species and forest ingrowth can be aimed at increasing forage production for livestock. Fullbright (1996) pointed out that with proper design, a treatment plan can be applied in a manner that also maintains or increases species diversity. Maximum forage production would be created with a mosaic of intermediately disturbed patches by varying the interval and timing of treatments as well as varying the actual wooded area treated (Fullbright 1996). A mosaic of treatments would create a variety of communities in various stages of succession (Fullbright 1996).

The BC Ministry of Forests deems "prescribed fire will be used where it is the most suitable ecological approach to site treatment to achieve desired objectives of land and resource management prescriptions" (BC Ministry of Forest 2005b). The conditions include having an approved burn prescription and plan, a burning permit, the objectives of the district fire management plan, BC Ministry policy and procedures on managing smoke, and regional and district plans on managing smoke. The prescribed burning must be monitored to ensure site preparation objectives are met, including assessment of fuel consumption (BC Ministry of Forests 2005b).

One of the components of the present study was to understand changes in understory plant communities, by recognizing a gradient of environmental conditions exists from the tree base to the canopy edge. Scholes and Archer (1997) compiled numerous studies which demonstrate that species composition of the herbaceous layer may change along gradients extending from the base of the tree to the canopy drip-line and into the adjoining inter-tree area. The presence and structure of the overstory canopy modifies light reaching the understory (Peltzer et al. 1998, Aubin et al. 2000). Production of biomass by understory vegetation and structure of vegetation results from competition for light. This study incorporates burning and thinning because these two treatments affect ecosystem processes and function differently, and they pose their own unique problems and advantages for controlling tree encroachment and for enhancing forage production. Burning can be dangerous and costly, especially if the fire becomes uncontrollable. It can, however, be advantageous in locations that are remote and pose low risk to humans. Thinning may not be possible in some areas because of access, but it can be successful in areas that are densely stocked with trees that cannot be burned safely.

Management of crown lands in BC is challenging because of the diversity of vegetation and because of the extensive overlap of land use with other resource interests (McLean et al. 1993). Issues such as public demand for energy, fiber, food, water, recreation, and minerals must be balanced with environmental issues such as biodiversity and conservation (McLean et al. 1993). The primary concerns of BC Ministry of Forests are to encourage maximum productivity of the forest and range resources and to manage, protect, and conserve those resources (BC Ministry of Forests 2005b). BC livestock producers must work with government and industry leaders to achieve maximum production while maintaining ecosystem health and sustainability.

3.0 PRESCRIBED BURNING FOR RESTORING UNDERSTORY VEGETATION IN FORESTED RANGE IN INTERIOR BRITISH COLUMBIA

3.1 Introduction

Forests account for nearly 80% of the range resources of BC crown lands (Wikeem et al. 1993). The dry forests in BC were historically dependent on fires to restrict forest ingrowth and tree encroachment. Fire-maintained stands benefit from surface fires by keeping tree densities at levels that the site can support (Gayton 1996). However, the fire suppression in forest management has allowed fuel accumulations, increasing the risk of intense wildfires (Covington and Sackett 1984, Daigle 1996, Gayton 1996). Decreased rangeland area, reduced carrying capacity, altered plant communities, deteriorated range condition, and loss of biodiversity are associated with forest ingrowth and tree encroachment (Ross 2000). Impeded cattle movement (Strang and Parminter 1980) and reduced access to forage supplies are of major concerns to ranchers. If the beef industry is to remain competitive and viable in BC, methods to control tree encroachment and restore understory vegetation must be found.

Fire can act as a natural "thinning" agent within forest stands which in turn reduces competition for resources (Humphrey 1962) such as light and nutrients (Gayton 1996). In addition, fires can remove accumulated conifer needles that can inhibit grass production (Weaver 1951). Several factors interact to determine vegetation response to fire, including climate, grazing, fire severity, ecosystem type, plant morphology, plant vigor and plant phenology (Feller 1996). The effects of fire on ground cover of *Festuca*- and *Agropyron*-dominated grasslands include reduced moss and lichen cover, increased soil temperatures, and reduced near-surface soil water (Antos et al. 1983).

Large trees tend to be resistant to fire damage because of their thicker bark, higher position of the foliage, and greater heat sink capacity (Costa et al. 1991). Tree mortality is common for trees with < 5 cm Diameter at Breast Height (DBH) for ponderosa pine (*Pinus ponderosa* Dougl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) (Johnson and Strang 1980). Tree age and fire severity affect the response to fire (Fonda et al. 1998). Fonda et al. (1998) concluded that ponderosa pine trees are considered fire-resistant and can exist in fire-stable forests, where frequent under-burns are supported largely by non-woody fuels. Physical characteristics of ponderosa pine allow them to survive surface fires that rapidly and almost completely consume the available fuels (Fonda et al. 1998). Douglas-fir trees are less flammable and are considered to be post-fire invaders that depend on abundant seed crops to restore forest communities (Fonda et al. 1998). Tree growth can be reduced by crown scorch, even in low-intensity fires (Busse et al. 2000).

Prescribed burning is the use of fire with pre-set characteristics to achieve a specified goal in a known plant community (Wikeem and Strang 1983). Prescribed burning is effective in situations where slopes preclude mechanical treatments (Pollet and Omi 2002). Goals of prescribed burning include increased forage production and quality, control of undesirable plant species, alteration of the botanical composition of the plant community (Wright 1974, Wikeem and Strang 1983), enhancement of decomposition rates, nutrient cycling, and net primary productivity (Moore et al. 1999), reduction in fire hazard, facilitation of planting and provision of soil seedbed for establishment and growth of plants (Lindeburgh 1990). In addition, prescribed burning can be used to reduce woody competition from shrubs and trees, thin stands, reduce diseases and insects, improve esthetics, enhance browse or grazing potential, and improve wildlife habitat (Lindeburgh 1990). Controlled burns usually cause more damage to shrubs and small tress than perennial grasses and forbs (Humphrey 1962). Timing of prescribed burning can be a major determinant of the effects of the fire. For example, burning tends to be more damaging in autumn than spring in grasslands, especially in *Festuca* and *Stipa-Agropyron* grasslands (Redmann et al. 1993). Burning when plants are dormant will impact plants least (Wright 1974).

The objectives of this research were to determine the short-term effects of burning on understory species diversity and standing crop of plants in the ponderosa pine forests of Interior BC. Burning is important for ecosystem restoration and it may increase forage production in grasslands and aid in maintaining long-term sustainability. It was hypothesized that the reducing tree canopy with burning increases species diversity and standing crop of the understory vegetation in grasslands of BC.

3.2 Materials and methods

3.2.1 Site description

Dew Drop (Tranquille Ecological Reserve) is located 20 km northwest of Kamloops, BC (115° 36' W 50° 45' N), within the ponderosa pine and Interior Douglas-fir biogeoclimatic zones. It covers 235 ha of forests and grasslands, ranging from 610 to 1,160 m in elevation. Vegetation changes with increasing elevation from open grasslands with big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) and bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith) as dominant species, to ponderosa pine forests and eventually Douglas-fir forests. Soils are Dark Brown Chernozems with sandy loam texture (McLean and Tisdale 1972). Average precipitation in Kamloops was 264 mm in 1998, 268 mm in 1999, 277 mm in 2000, 254 mm in 2001 and 221 mm in 2002 (Environment Canada 2005). The area was fenced by the BC Ministry of Forests in 1993 to exclude livestock to preserve a representative ponderosa pine / Douglas-fir ecosystem in a prominent valley (Morrow 1993). Historical fire return intervals in the open grasslands averaged 7 to 10 years, while the fire return interval for ponderosa pine stands averages 10 to 12 years (Morrow 1993). Until the late 1970s the BC Ministry of Environment had a burning program in the Dew Drop area that consisted of controlled burns that were ignited to increase or maintain plant species diversity for wildlife habitats.

3.2.2 Experimental design and treatment design

Five pairs of plots, 40 x 50 m in size, were established in open grassland and in forest in July 1998 (total 10 pairs). Burning was conducted at Dew Drop between 13:20

and 17:20 on April 1 1999 in conjunction with BC Park Service and BC Ministry of Forests. The air temperature was between 14 and 16^{o} C, relative humidity ranged from 36-38% and wind speed of varied between 8-12 km h^{-1}. The forest was burned in one large patch and fire breaks were made before ignition to protect the control areas. This study was a randomized complete block design (RCBD) with 5 replicates. Pre-burning tree density averaged 43 and 790 stems ha^{-1}, respectively, for the grassland and forest plots. Tree densities were similar in the burned and control plots ($P = 0.256$). Post-burning tree density in 2002 averaged 10 and 370 stems ha^{-1}, respectively, for the grassland and forest plots.

Experiment 1: Effect of burning on understory species composition and standing crop

Two transects, 20 m in length, were established at the center of each plot. The first transect was positioned to be representative of the plot and the second transect was placed at the left side and 10 m apart from the first one. DBH of trees within 5 m of both sides of transects were measured. The 20 x 20 m area was subdivided into 16, 5 x 5 m sub-plots. DBH (for trees taller than 2 m) or height (for trees shorter than 2 m) was measured for each tree. The scorch height on ponderosa pine and Douglas-fir trees within each plot were measured immediately after burning and the survival of each tree was estimated 3 years after the burning.

Experiment 2: Burning effects on understory vegetation under and outside of the tree canopy

Four ponderosa pine trees within each plot were selected to study the effect of tree canopy and burning on understory vegetation. Trees selected had intermediate sizes relative to other trees within each plot, relatively straight stems, and regular (round) and even (equal dimensions along all directions) canopies. Trees with partial canopy overlaps were avoided because they may compromise the influence of individual trees. Sizes of the trees selected were similar between the control and burning treatments as measured by basal diameter (Table 3.1). Four transects along N, E, S, and W were established from the stem to the edge of the crown projection area of each tree (Fig.

3.1). Crown projection area was estimated by visually projecting the edges of the canopy down to the soil surface (Barbour et al. 1999).

Table 3.1 Average basal diameter of ponderosa pine (mean ± SE) selected for the effect of canopy on understory vegetation at Dew Drop, BC.

Treatment	Basal diameter (cm)	Range (cm)
Control	26.7 ± 9.1	22.3 – 44.3
Burning	37.8 ± 1.5	30.9 – 43.3
P-value	0.256	

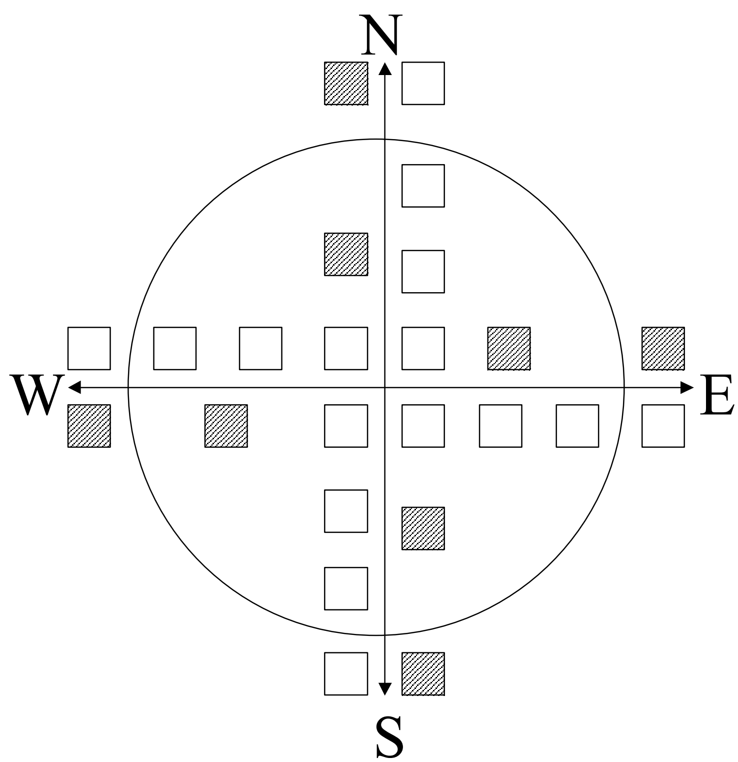

Figure 3.1 Layout of transects and quadrats surrounding the tree stem under and outside the canopy projection area. Unfilled squares were quadrat locations for ground cover and species composition in the order of Q1, Q2, Q3, and Q4 from the center to the edge of canopy. Standing crop was determined in filled squares.

3.2.3 Data collection

Experiment 1: Effect of burning on understory species composition and standing crop

Ten, 0.4 x 0.5 m quadrats, were placed 2 m apart on the right side of each 20 m transect. Percent canopy cover (Barbour et al. 1999) of each vascular plant species, litter (barks, needles, and other dead plant materials), and bare soil were visually estimated in June to July 1998 (before treatment), 1999, and 2002 within these quadrats. The height of live vegetation and the depth of litter within each quadrat were also measured in 3 sub-samples. Clipping of all vegetation to ground level within the Standing crop of understory plants was determined in late July 2002 on the left side of each 0.4 x 0.5 m quadrats, at 5 and 15 m, by hand clipping plants to ground level. Standing crop were oven-dried at 60^{o}C for 12 to 24 hours, and weighed to determine total standing crop. Samples were then sorted into shrubs, forbs, live graminoids, and dead graminoids and weighed separately.

Experiment 2: Burning effects on understory vegetation under and outside of the tree canopy

From the base of each ponderosa pine tree a transect was laid and four locations were determined: 1) tree bole (Q1); 2) halfway between the tree bole and edge of the crown projection area (Q2); 3) 30 cm inside the edge of the crown projection area (Q3); and, 4) 60 cm outside the edge of the crown projection area (Q4) (Fig. 3.1). Quadrat 3 (Q3) was not measured when the crown projection area was too small to separate Q2 from Q3. This procedure was repeated in four directions: north, east, south and west. A 0.3 x 0.3 cm quadrat was used for Q1-Q4 to determine percent cover (Barbour et al. 1999) of ground cover and canopy cover of understory species in June 2001 and 2002. Understory standing crop was determined in 0.3 x 0.3 m quadrats by hand clipping plants to ground level. Quadrats in which standing crop was determined, were placed at the left side of each transect, within and outside the tree canopy (at the same location as Q2 and Q4). Standing crop was pooled for Q2 in all four directions, and pooled for Q4 in all four directions. Standing crop were oven-dried at 60^{o}C for 12 to 24 hours, and weighed to determine total standing crop. Samples were then sorted into shrubs, forbs, live graminoids, and dead graminoids and weighed separately.

3.2.4 Data analysis

Experiment 1: Effect of burning on understory species composition and standing crop

Species canopy cover from quadrats along the two, 20 m transects were pooled for each plot. Species richness (R), species evenness (E) and the Shannon-Weiner Diversity Index (*H*') (Barbour et al. 1999) were calculated for each plot and within years using PC-ORD (McGarigal et al. 2000). Data were then analyzed separately within years using Analysis of Variance (ANOVA) (SAS Institute 1995) to determine the effect of burning treatment on species composition. Data from 1998, 1999 and 2002 were combined for ordination with rare species (species that occurred in ≤ 2 plots or 10% of the plots) being removed (McGarigal et al. 2000). Cover data were relativized by the species maximum and tested with Detrended Correspondence Analysis (DCA). Scores of the first three axes of DCA were calculated and analyzed with ANOVA to determine the effect of year and treatment on plot separation along the 3 axes. The first axis represents the maximum amount of variation possible in a single dimension. The second axis is constrained by orthogonality and maximization of the remaining variance. Therefore, the second axis is statistically independent of the first axis and it explains the maximum, remaining variation not explained by the first axis (McGarigal et al. 2000).

Ground cover including rocks, logs, bare soil, litter and lichens, the cover of functional groups including forbs, graminoids, shrubs and total cover, and litter depth and vegetation height were analyzed within years using ANOVA to determine the effects of burning (Snedecor and Cochran 1980). Graminoids includes grasses, sedges and rushes. The shrub category also included trees (<0.5 m tall), but excluded pasture sage (*Artemisia frigida* Willd.) which was considered a forb. Standing crop of forbs, live graminoids, dead graminoids, shrubs, and total understory standing crop were analyzed using ANOVA. Means were separated using the Least Significant Difference test (LSD) (Snedecor and Cochran 1980). A significance value of $P \leq 0.05$ was used.

Experiment 2: Burning effects on understory vegetation under and outside of the tree canopy

Species composition data along the four cardinal directions from each tree bole were pooled according to quadrat positions (Q1, Q2, Q3 or Q4). Data were combined from trees within each plot (sub-samples), then pooled according to quadrat locations. All three quadrats under the canopy (Q1, Q2 and Q3) were averaged for ground cover and referred to as "under canopy". R, E and *H*' were calculated for under the canopy (using Q2 only) and outside of the canopy (Q4) for each plot and year using PC-ORD and analyzed within years (2001 and 2002) using the General Linear Model (GLM) (SAS Institute 1995). Each combination of canopy cover type (under the canopy or outside the canopy) was treated as a "plot" in ordination after removing species that occurred in only two plots (rare species) (McGarigal et al. 2000). Data were relativized by the species maximum and then subject to DCA separately for each year. Scores of the first three axes of DCA were calculated and analyzed with GLM to determine whether the effect of treatment and canopy cover type on species composition can be separated along the three axes.

Ground cover, the canopy cover of functional groups, litter depth and vegetation height were pooled according to quadrat location within each plot and analyzed within years using GLM to determine the effect of canopy removal and quadrat location. Data of the three quadrats under the tree canopy were then combined and analyzed again using GLM. Standing crop of forbs, live graminoids, dead graminoids, shrubs and total understory standing crop were also analyzed with GLM as described above. Means were separated using LSD (Snedecor and Cochran 1980). A significance value of $P \leq 0.05$ was used.

3.3 Results

3.3.1 Burning effects on tree survival and understory plant species composition

Experiment 1: Effect of burning on understory species composition and standing crop

Scorch height was positively correlated with tree size ($P < 0.01$) (Fig. 3.2). Tree survival after burning increased with increasing tree size for both species. The survival of ponderosa pine trees increased by 60% when DBH increased from less than 10 cm to a range of 10 to 19 cm. Survival increased to 100% in Douglas-fir when DBH was at least 20 to 29 cm. Species richness (R), species evenness (E) and the Shannon-Weiner Diversity Index (H') were similar in the control and the plots before burning in the grassland and forest plots (Table 3.2). Burning had no affect on R, E and H' in the year immediately after burning or 3 years after burning in grasslands. Within the forest, E was reduced 13% and H' was 27% lower ($P = 0.014$ and $P = 0.038$, respectively) in the burned plots than the control in three years after burning (2002).

Forb canopy cover was not affected by burning (Table 3.2). Shrub canopy cover was reduced from 8 to 1% ($P = 0.005$) by burning the grasslands in the same year after burning (1999), but the difference in shrub cover between the control and burning was not significant in 2002 ($P = 0.191$). Shrub canopy cover in the forest was limited and not different between the control and burning plots ($P = 0.374$ in 1999 and 2002). Graminoid canopy cover was greater in burned grassland plots (20%) than control (15%) before the burning in 1998 ($P = 0.038$), but it was not significantly different between the control and burning in subsequent years ($P = 0.164$ in 1999, $P = 0.102$ in 2002), indicating a relative reduction in graminoid cover by burning. Within the forested plots, graminoid cover was less in burned plots (2%) than the control plots (7%) in 1999 ($P = 0.014$), but it recovered to 3% for both burned and control plots by 2002 ($P = 0.568$). The most abundant grass in the grassland was needle-and-thread (*Stipa comata* Trin. & Rupr.) (Table 3.3). No obvious changes in the relative ranking of species were noted within grasslands, except that burning reduced big sagebrush. Bluebunch wheatgrass was the most abundant species in the forest, and cover was reduced by burning. Spatial variability in species composition was greater among blocks than that caused by burning or year as revealed by DCA (data not shown); therefore, burning plots cannot be separated from the control.

Experiment 2: Burning effects on understory vegetation under and outside of the tree canopy

No significant effects of burning or quadrat location were found for R, E, and H' ($P > 0.05$). In 2001, R, E and H' were 7.0, 0.66 and 1.29, respectively and in 2002, R, E and H' were 10.0, 0.70, and 1.62, respectively. Neither treatment nor quadrat location affected forb cover ($P > 0.05$; data not shown), overall average forb cover was 1%. Burning reduced shrub cover from 2 to <1% beneath the canopy in 2001 ($P = 0.016$), but forb cover and graminoid cover remained unaffected by burning ($P = 0.864$ and 0.354, respectively). Outside of the tree canopy, graminoid cover was greater than under the canopy in 2001 and 2002 ($P = 0.007$ and $P = 0.032$, respectively). Overall average shrub cover was <1%, grass cover was 3%, and total vegetation cover was 5% (data not shown).

Bluebunch wheatgrass was most abundant under and outside of the canopy (Table 3.4), this was similar to transect data in the forest plots of Experiment 1. Outside of the canopy the cover of rabbitbrush (*Chrysothamnus nauseosus* (Pall.) Britt.) decreased, while that of needle-and-thread increased in 2001 and 2002 after burning.

3.3.2 Tree canopy and burning effects on ground cover

Experiment 1: Effect of burning on understory species composition and standing crop

Before burning, the amount of bare soil was lower ($P = 0.043$) and litter cover was greater ($P = 0.044$) in the control than in burned plots in grasslands, while plots in the forest was similar between the burnt and the control plots (Table 3.5). Litter depth ($P = 0.035$ in grassland and $P = 0.037$ in forest), and total vegetation cover ($P = 0.009$ in grassland and $P = 0.024$ in forest) were reduced by burning for grassland and forest sites in 1999, but they recovered by 2002. Vegetation height was reduced in grasslands in 1999, but plant height had recovered by 2002 ($P = 0.002$). In the forested sites, vegetation height was reduced by burning in 2002 ($P = 0.044$).

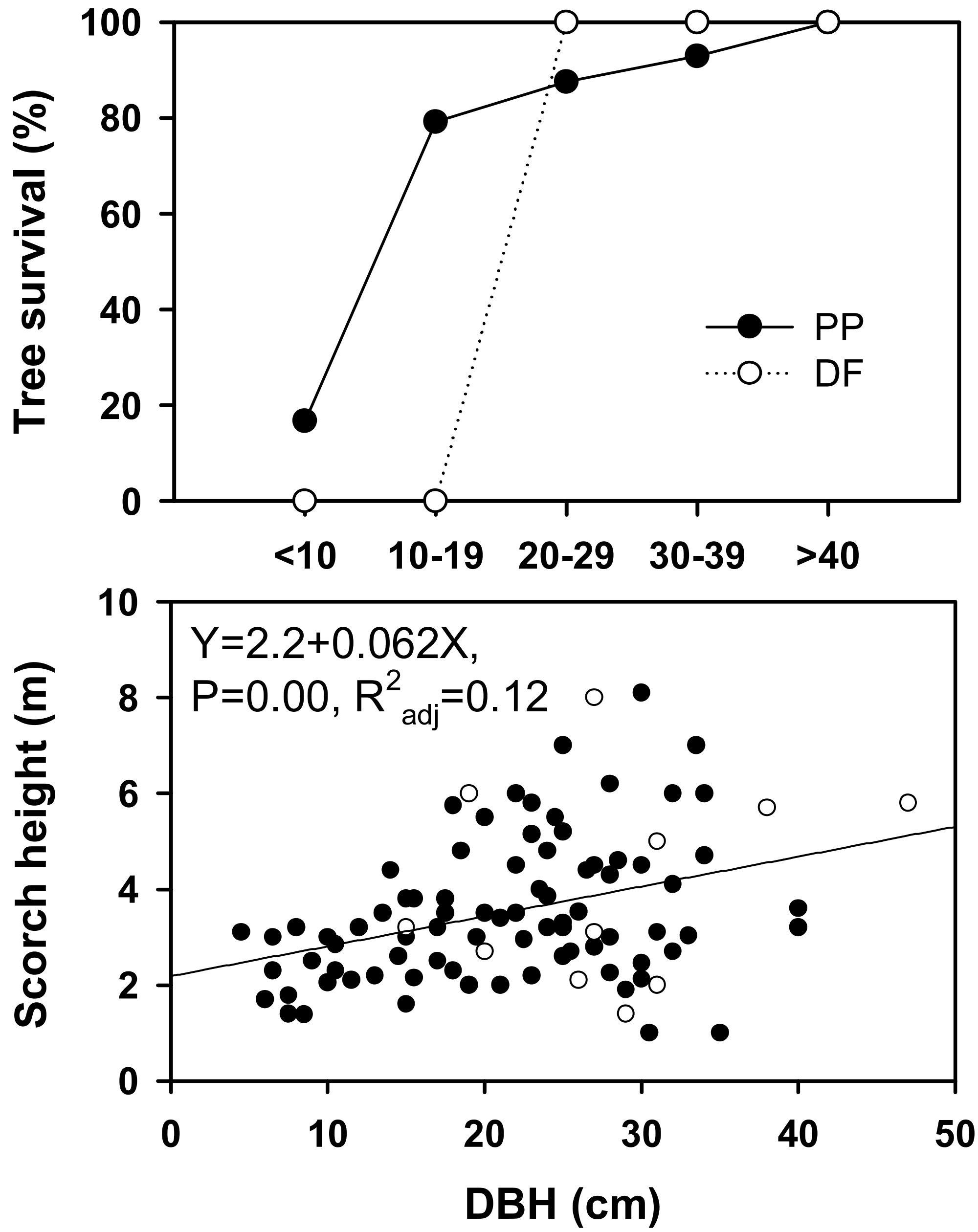

Figure 3.2 a. Survival of ponderosa pine in relation to Diameter at Breast Height (DBH) classes at Dew Drop, BC following prescribed burning in April 1999. b. Relationship between scorch height and DBH of ponderosa pine trees.

Table 3.2 Burning effects on species richness (R), species evenness (E) and the Shannon-Weiner Diversity Index (H') and the canopy cover of functional groups (means ± SE) at Dew Drop, BC. Prescribed burning was conducted in April 1999.

Site/Year	Treatment	R	E	H'	Forb (%)	Shrub (%)	Graminoid (%)
Grassland							
1998	Control	14.0 ± 0.8	0.70 ± 0.02	1.84 ± 0.08	4 ± 0.9	9 ± 1.6	15 ± 1.1 b[1]
	Burning	16.0 ± 2.1	0.62 ± 0.02	1.69 ± 0.05	3 ± 0.5	8 ± 2.4	20 ± 2.2 a
	P	0.389	0.060	0.225	0.785	0.795	0.038
1999	Control	18.4 ± 1.6	0.69 ± 0.02	2.00 ± 0.08	5 ± 0.9	8 ± 1.3 a	19 ± 3.1
	Burning	17.8 ± 0.6	0.71 ± 0.03	2.05 ± 0.09	3 ± 0.9	1 ± 0.4 b	12 ± 0.9
	P	0.722	0.689	0.705	0.364	0.005	0.164
2002	Control	14.4 ± 0.7	0.71 ± 0.03	1.87 ± 0.06	4 ± 0.5	2 ± 0.8	17 ± 0.7
	Burning	17.2 ± 1.0	0.70 ± 0.01	1.98 ± 0.03	5 ± 0.8	<1 ± 0.2	20 ± 1.0
	P	0.094	0.789	0.228	0.490	0.191	0.102
Forest							
1998	Control	8.8 ± 1.8	0.56 ± 0.11	1.25 ± 0.29	1 ± 0.7	2 ± 1.5	8 ± 1.6
	Burning	7.0 ± 0.9	0.53 ± 0.06	0.97 ± 0.06	<1 ± 0.1	2 ± 0.7	5 ± 0.6
	P	0.405	0.743	0.334	0.265	0.896	0.299
1999	Control	9.0 ± 2.2	0.51 ± 0.07	1.09 ± 0.24	1 ± 0.4	<1 ± 0.4	7 ± 1.1 a
	Burning	5.8 ± 0.6	0.45 ± 0.09	0.80 ± 0.21	<1 ± 0.1	0 ± 0.0	2 ± 0.3 b
	P	0.199	0.464	0.293	0.066	0.374	0.014
2002	Control	11.0 ± 1.9	0.80 ± 0.03 a	1.86 ± 0.17 a	2 ± 0.7	<1 ± 0.3	3 ± 0.7
	Burning	6.8 ± 0.5	0.70 ± 0.05 b	1.35 ± 0.13 b	1 ± 0.1	0.0 ± 0.0	3 ± 1.0
	P	0.863	0.014	0.038	0.240	0.374	0.568

[1] Means with the different lower-case letter within a column, site and year are significantly different at $P \leq 0.05$.

Table 3.3 Burning effects on canopy cover (%) and relative rankings (in parentheses) of the dominant species at Dew Drop, BC in 1998, 1999 and 2002. Prescribed burning was conducted in April 1999.

	Control			Burning		
Site/Species	1998	1999	2002	1998	1999	2002
Grassland						
Agropyron spicatum	2.0 (4)	2.1 (4)	1.4 (3)	2.9 (5)	1.9 (4)	1.7 (3)
Antennaria microphylla	0.6 (6)	1.2 (6)	1.4 (3)	0.6 (7)	0.6 (7)	0.6 (7)
Artemisia frigida	1.0 (5)	0.7 (7)	0.5 (7)	1.1 (6)	0.3 (8)	0.9 (6)
Artemisia tridentata	5.4 (2)	6.7 (1)	1.2 (6)	8.1 (2)	1.1 (5)	0.0 (8)
Koeleria macrantha	3.4 (3)	4.2 (3)	7.2 (1)	4.1 (3)	2.8 (2)	6.3 (2)
Poa pratensis	0.0 (7)	0.0 (8)	0.0 (8)	3.7(4)	2.0 (3)	1.0 (5)
Poa sandbergii	0.0 (7)	1.3 (5)	1.4 (3)	0.0 (8)	0.7 (6)	1.3 (4)
Stipa comata	7.7 (1)	6.5 (2)	5.9 (2)	8.8 (1)	7.2 (1)	8.6 (1)
Forest						
Achillea millefolium	0.1 (7)	0.2 (4)	0.2 (7)	0.1 (5)	0.1 (2)	0.2 (4)
Agropyron spicatum	5.0 (1)	5.1 (1)	1.6 (1)	4.2 (1)	1.7 (1)	1.8 (1)
Chrysothamnus nauseosus	0.9 (3)	0.0 (7)	0.3 (3)	2.0 (2)	0.0 (3)	0.0 (6)
Festuca campestris	0.7 (4)	0.0 (7)	0.3 (3)	0.6 (3)	0.0 (3)	0.3 (2)
Juniperus scopulorum	1.0 (2)	1.0 (2)	0.0 (8)	0.0 (7)	0.0 (3)	0.0 (6)
Koeleria macrantha	0.2 (6)	0.2 (4)	0.3 (3)	0.2 (4)	0.0 (3)	0.2 (4)
Oxytropis campestris	0.5 (5)	0.2 (4)	0.4 (2)	0.1 (5)	0.0 (3)	0.3 (2)
Poa sandbergii	0.0 (8)	0.3 (3)	0.3 (3)	0.0 (7)	0.0 (3)	0.0 (6)

Experiment 2: Burning effects on understory vegetation under and outside of the tree canopy

One year after burning (2001), log cover was reduced 4 to 2% under the canopy and 7 to 2% outside the tree canopy (P = 0.005) while the cover of bare soil increased from 3 to 10% under the canopy and 3 to 5% outside the tree canopy in the burned (P = 0.010). Litter cover was similar between quadrat locations within the control (P = 0.078) and within burning (P = 0.484) (Table 3.6). Burning did not affect litter cover outside the canopy (P = 0.149), but it was reduced by burning under the canopy (P = 0.037). Litter depth was not different between treatments outside the canopy (P = 0.123), but it was reduced by burning under the canopy (P = 0.009). Burning reduced lichen cover outside (P = 0.045) and under the canopy (P = 0.044); lichen cover was greater outside the canopy than under the canopy (P = 0.032).

Three years following burning (2002), litter cover, litter depth and lichen cover were less than the control (P = 0.005, 0.010 and 0.016, respectively). Outside the canopy only lichen cover was reduced by burning (P = 0.013). Vegetation height was not affected by burning and quadrat location (P = 0.649 and 0.679, respectively).

Table 3.4 Canopy cover (%) with relative ranking (in parentheses) of the dominant species under and outside of the canopy of ponderosa pine at Dew Drop, BC in 2001 and 2002 after burning.

	Control		Burning	
Year/Species	Under	Outside	Under	Outside
2001				
Achillea millefolium	0.1 (6)	0.3 (5)	0.3 (2)	0.2 (5)
Agropyron spicatum	2.3 (1)	3.2 (1)	1.7 (1)	3.9 (1)
Chrysothamnus nauseosus	2.0 (2)	1.6 (2)	0.0 (7)	0.0 (8)
Erigeron spp.	0.1 (6)	0.4 (4)	0.1 (6)	0.2 (5)
Koeleria macrantha	0.2 (4)	1.0 (3)	0.2 (4)	0.6 (2)
Oxytropis campestris	0.3 (3)	0.0 (7)	0.2 (4)	0.2 (5)
Stipa comata	0.0 (8)	0.0 (7)	0.0 (7)	0.3 (3)
Stipa curtiseta	0.2 (4)	0.1 (6)	0.3 (2)	0.3 (3)
2002				
Achillea millefolium	0.4 (3)	0.4 (3)	0.3 (2)	0.3 (5)
Agropyron spicatum	1.6 (1)	2.0 (1)	2.2 (1)	3.3 (1)
Antennaria microphylla	0.2 (6)	0.4 (3)	0.1 (6)	0.1 (7)
Chrysothamnus nauseosus	1.0 (2)	0.1 (7)	0.0 (8)	0.0 (8)
Erigeron spp.	0.1 (7)	0.2 (6)	0.1 (6)	0.2 (6)
Koeleria macrantha	0.4 (3)	0.9 (2)	0.3 (2)	0.7 (2)
Oxytropis campestris	0.3 (5)	0.0 (8)	0.2 (4)	0.6 (3)
Stipa comata	0.1 (7)	0.3 (5)	0.2 (4)	0.4 (4)

Table 3.5 Effects of burning on cover of bare soil, litter cover, litter depth, vegeation height and total vegetation cover (means ± SE) at Dew Drop, BC. Prescribed burning was conducted in April 1999.

Site/Year	Treatment	Bare Soil (%)	Litter cover (%)	Litter depth (cm)	Vegetation height (cm)	Vegetation cover (%)
Grassland						
1998	Control	67 ± 4.6 a[1]	32 ± 4.8 b	0.5 ± 0.0	41.6 ± 2.3	27 ± 1.9
	Burning	47 ± 4.6 b	52 ± 4.5 a	0.5 ± 0.0	47.5 ± 4.6	31 ± 2.7
	P	0.043	0.044	0.376	0.178	0.426
1999	Control	42 ± 5.2	57 ± 5.9	0.8 ± 0.2 a	43.0 ± 2.2 a	32 ± 2.7 a
	Burning	47 ± 6.7	52 ± 6.4	0.3 ± 0.1 b	32.5 ± 0.8 b	16 ± 1.3 b
	P	0.337	0.370	0.035	0.002	0.009
2002	Control	13 ± 3.2	61 ± 4.4	2.9 ± 0.2	32.3 ± 2.5	23 ± 0.7
	Burning	14 ± 4.3	64 ± 4.3	3.1 ± 0.2	28.9 ± 1.7	25 ± 1.3
	P	0.823	0.588	0.526	0.153	0.291
Forest						
1998	Control	7 ± 2.0	87 ± 4.6	2.9 ± 0.5	29.4 ± 6.2	11 ± 3.2
	Burning	3 ± 0.9	96 ± 0.9	3.2 ± 0.2	27.9 ± 1.6	8 ± 0.9
	P	0.123	0.142	0.673	0.773	0.364
1999	Control	5 ± 1.6	93 ± 2.8	2.7 ± 0.7 a	25.6 ± 4.0	9 ± 1.8 a
	Burning	11 ± 4.3	84 ± 5.2	0.5 ± 0.1 b	11.1 ± 2.0	2 ± 0.3 b
	P	0.236	0.207	0.037	0.053	0.024
2002	Control	2 ± 1.1	87 ± 2.8	4.6 ± 0.6	30.2 ± 2.9 a	5 ± 1.1
	Burning	2 ± 0.5	91 ± 2.6	3.3 ± 0.1	18.8 ± 2.4 b	4 ± 1.0
	P	0.855	0.425	0.076	0.044	0.270

[1] Means with different lower-case letters within a column, site and year are significantly different at $P \leq 0.05$.

Table 3.6 Burning effects on cover of litter cover, litter depth and lichen cover (means ± SE) under and outside the canopy of ponderosa pine trees at Dew Drop, BC. Prescribed burning was conducted in April 1999.

Year/Location	Treatment	Litter (%)	Litter depth (cm)	Lichen (%)
2001				
Under canopy	Control	90 ± 2.1 a[1]	3.8 ± 0.3 a	5 ± 1.4 a
	Burning	74 ± 3.6 b	1.7 ± 0.1 b	0 ± 0.0 b
	Mean	82.2	2.7	2.4 B[2]
Outside canopy	Control	82 ± 2.3	2.5 ± 0.4	11 ± 3.3 a
	Burning	76 ± 4.7	1.6 ± 0.2	1 ± 0.5 b
	Mean	79.0	2.0	5.7 A
2002				
Under canopy	Control	90 ± 2.7 a	3.9 ± 0.4 a	34 ± 1.2 a
	Burn	78 ± 3.5 b	1.9 ± 0.1 b	<1 ± 0.2 b
	Mean	84.0	2.9	1.9 A
Outside canopy	Control	87 ± 1.9	2.7 ± 0.4	7 ± 1.7 a
	Burn	81 ± 4.2	1.8 ± 0.1	1 ± 0.3 b
	Mean	83.4	2.3	3.8 A

[1] Means with different lower-case letters within a column, year and quadrat location are significantly different at $P \leq 0.05$.
[2] Means with different capital letters within a column and year are significantly different at $P \leq 0.05$.

3.3.3 Understory standing crop

Experiment 1: Effect of burning on understory species composition and standing crop

Although results were insignificant ($P > 0.05$), data trends indicated burning increased standing crop of forbs by 173% ($P = 0.064$) and the total for understory species by 50% ($P = 0.057$) in the grassland in 2002 (Table 3.7). Graminoid standing crop was reduced 47% by burning in the forest ($P = 0.049$).

Experiment 2: Burning effects on understory vegetation under and outside of the tree canopy

Standing crop of forbs, shrubs, graminoids and all species combined was not affected by burning in 2001, but graminoid standing crop was greater outside the canopy than under the canopy. Standing crop in the control under the tree canopy in order of forbs, shrubs live graminoids, dead graminoids and total standing crop: 2.5, 0.0, 10.4, 19.5 and 32.4 g m^{-2}. Standing crop in the burn under the tree canopy in order of forbs, shrubs, live graminoids, dead graminoids and total standing crop: 1.4, 0.0, 22.3, 20.2 and 43.9 g m^{-2}. Standing crop in the control outside the tree canopy in order of forbs, shrubs, live graminoids, dead graminoids and total standing crop: 5.5, 0.0, 8.3, 16.3 and 30.1 g m^{-2}. Standing crop in the burn outside the tree canopy in order of forbs, shrubs, live graminoids, dead graminoids and total standing crop: 1.8, 0.0, 18.4, 20.4 and 40.6 g m^{-2}.

Standing crop of forbs and graminoids was not affected by burning or quadrat location in 2002 ($P = 0.177$ and $P = 0.230$, respectively) (Table 3.8). Data trends indicated that burning reduced shrub standing crop under the canopy, but total understory standing crop was not affected by burning or quadrat location ($P = 0.087$ and $P = 0.102$, respectively).

3.4 Discussion

Fire is used within forests to remove smaller trees and to prune lower branches, thus increasing the 'openness' of the stand. Prescribed burning acted as a 'thinning' agent by eliminating smaller trees in this study. Crown scorch severity can be a predictor of tree mortality (van Mantgem and Schwartz 2004). In this study, the positive correlation between scorch height and tree size was weak ($R^2_{adj} = 0.12$). When ponderosa pine trees are at a DBH of 10 to 19 cm and Douglas-fir trees have a DBH of 20 to 29 cm, their survival increased dramatically (60 and 100%, respectively). Increased survival for larger trees is attributed to the increasing bark thickness, the principal factor affecting cambium temperature during burning (Costa et al. 1991). Ponderosa pine trees tend to have fire-resistant bark and dominate the canopy in fire-maintained forests (Gayton 1996). In contrast Douglas-fir trees are relatively thin-barked and are the primary tree that in-grows in stands where fire is excluded in the forests of Interior BC (Gayton 1996). Tree species, size, fuel consumption, and season of burn all impact the amount scorching that trees can withstand (Stephens and Finney 2002).

Table 3.7 Aboveground standing crop in the understory (means ± SE, g m^{-2}) at Dew Drop, BC in 2002, three years after burning. Prescribed burning was conducted in April 1999.

Site	Treatment	Forbs	Shrubs	Standing live graminoids	Standing dead graminoids	Total standing crop
Grassland						
	Control	14.7 ± 4.9 b[1]	19.0 ± 8.2	76.8 ± 7.9	110.6 ± 9.5	211.0 ± 11.3 b
	Burning	40.1 ± 13.3 a	6.2 ± 5.6	98.8 ± 14.5	170.0 ± 30.6	315.1 ± 41.8 a
	P	0.064	0.231	0.146	0.131	0.057
Forest						
	Control	0.8 ± 0.6	0.5 ± 0.5	31.8 ± 5.7 a	31.8 ± 6.5 a	64.9 ± 12.3
	Burning	7.1 ± 2.8	0.2 ± 0.2	16.8 ± 2.9 b	15.4 ± 4.6 b	39.2 ± 8.7
	P	0.104	0.704	0.049	0.097	0.148

[1] Means with different lower-case letters within a column and site are significantly different at P ≤ 0.05.

Table 3.8 Effects of burning on aboveground standing crop in the understory (means ± SE, g m^{-2}) under and outside of the canopy of ponderosa pine trees at Dew Drop, BC in 2002. Prescribed burning was conducted in April 1999.

Location	Treatment	Forbs	Shrubs	Standing live graminoids	Standing dead graminoids	Total standing crop
Under canopy	Control	1.0 ± 0.4	1.3 ± 0.8 a[1]	6.0 ± 1.0	5.5 ± 0.9 b	13.7 ± 1.2
	Burning	0.7 ± 0.1	0.0 ± 0.0 b	10.8 ± 4.3	11.2 ± 5.2 ab	22.7 ± 9.7
Outside canopy	Control	2.3 ± 1.1	0.2 ± 0.2 ab	9.4 ± 3.3	12.9 ± 4.5 ab	24.8 ± 9.0
	Burning	2.0 ± 0.5	0.0 ± 0.0 b	14.2 ± 2.1	20.5 ± 4.1 a	36.7 ± 6.3
	P	0.177	0.087	0.230	0.055	0.102

[1] Means with different lower-case letters within a column, year and quadrat location are significantly different at P ≤ 0.05.

Burning can alter aboveground attributes in grasslands and forests including light, air temperature and soil temperature (Riegel et al. 1995). Changes in species richness, species evenness and species diversity in grasslands after burning were not significant. However, species evenness and diversity were reduced after burning the forest. The different responses of vegetation between grasslands and forest could possibly be due to the higher fire severity associated with more fuel in the latter. Increases in species diversity are desirable, especially in grassland communities, because it may increase the ability of the ecosystem to withstand and recover from drought conditions (Tilman and Downing 1994). The reduction in species evenness and diversity in this study contradicts Busse et al. (2000) in which species richness and diversity increased for 2 years following underburning in thinned ponderosa pine stands. Low severity fires may change species composition because these fires characteristically have low flame length, reduced tree mortality, less consumption of organic material and minimal changes in chemical properties of soils compared to more severe burns (Busse et al. 2000). Fraas et al. (1991) focused on the effects of prescribed burning in a bitterbrush (*Purshia tridentata* Pursh)-mountain big sagebrush (*Artemisia tridentata* Nutt. *ssp. vaseyana* (Rydb.))-bluebunch wheatgrass community for eight years in Montana. Burning increased the plant species present, but cover of graminoids and forbs was unchanged, indicating that burning had a more direct effect on species composition. Species in burned plots included rabbitbrush, threadleaf sedge (*Carex filifolia* Nutt.), cheatgrass (*Bromus tectorum* L.), creeping aster (*Aster conspicuus* Lindl.), and pasture sage.

The presence and structure of the overstory canopy directly influence light reaching the understory; understory vegetation productivity and structure are the result of competition for light (Dodd et al. 1972, Peltzer et al. 1998, Harrington and Edwards 1999, Aubin et al. 2000). Graminoid cover and standing crop were greater outside the canopy than under the canopy, indicating a possible response to light by these species. Burning in forests removes lower branches and smaller trees, thus opening the area. Burning contrasts with thinning whereby the tree canopy is completely removed. Ideally, the removal of tree canopy will increase the amount of light reaching understory vegetation, and consequently, increase understory plant cover and species richness (Thomas et al. 1999, Fornwalt et al. 2003). Altering canopy structure within ponderosa pine forests by thinning changed species composition, while the relative composition of graminoids, forbs, and shrubs remained unchanged (Riegel et al. 1995). Thinning and burning may produce similar results, but they differ in their feasibility. Burning may be more effective on sites with steep slopes that preclude mechanical treatment and in locations where burning is possible (Pollet and Omi 2002). Thinning tends to be most effective in forests that are too dense to burn and areas that have nearby markets for small diameter trees (Pollet and Omi 2002). Both treatments are used to counteract the dominant influence that tree canopies can have on the environment. The additional litter and shading from ingrowth and tree encroachment can possibly inhibit aboveground production, and without intervention, woody plants can increase to the point where the site can no longer produce the same amount (Humphrey 1962).

Burning at Dew Drop took place in early April, after grasses had already started growing. Timing of the burning focused on meeting the objectives of BC Parks; reducing fuel and woody debris, and killing smaller trees. This study revealed that burning initially reduced shrub and graminoid cover within the grassland and the forest, with both functional groups recovering 3 years after burning. Significant changes within the first year after burning (increased cover and density), followed by less dramatic results in the second year occurred in ponderosa pine forests (Riegel et al. 1995). A low-severity prescribed burn in the spring in thinned, ponderosa pine forests of central Oregon reduced shrub cover by more than 50%, but burning had little effect on forb or graminoid cover (Busse et al. 2000). Similar responses were observed in Montana eight years following burning; bitterbrush and mountain big sagebrush cover was reduced in burned plots (Fraas et al. 1991). Fire did not carry through the Dew Drop forest sites evenly because the main fuel, pine needles, varied in density throughout the stand. This unevenness of litter cover resulted in variations of fire intensity and some patches of grasses were untouched by fire (Don Thompson, personal observation). Minimal response of shrubs to canopy reduction by burning was observed in a ponderosa pine forest possibly due the inability of shrubs to respond to the additional light as quickly as graminoids or forbs (Riegel et al. 1995). Graminoid and forb cover increased 16 months following a spring fire in a sand prairie (Schulten 1985). Fires in grasslands are used to stimulate forage production and control species such as broomweed (*Gutierrezia sarothrae* (Pursh) Britt. & Rusby) and juniper (*Juniperus* L.) (Wright 1974). In ponderosa pine forests, controlled burns can reduce the wildfire hazard, thin stands, create a mineral seedbed for regeneration and maintain a healthy understory (Weaver 1951).

Beneath the canopy, burning reduced litter cover and litter depth. Changes in ground cover have major impacts on soil water, temperatures and nutrient cycling (Lindeburgh 1990). Reduced litter cover and depth will reduce damage caused by future fires (Kauffman and Martin 1989). For example, large amounts of dead woody fuels and deep litter and duff layers release only a fraction of their energy in the flaming front (Kauffman and Martin 1989). Once the flaming front has passed, most of the energy is released by smoldering and intermittent combustion (Kauffman and Martin 1989). Burning reduced lichen cover, and areas outside the tree canopy had less lichen cover than beneath the canopy the second year following burning in the present study. Lichen colonies can be killed or severely damaged by fire in sand prairie (Schulten 1985) and *Festuca* and *Agropyron* grasslands (Antos et al. 1983).

Patch burning, along with subsequent increased grazing pressure after burning promoted forb production and changed the plant community from being grass-dominated to forb-dominated (Vermeire et al. 2004). Increases in standing crop do not necessarily equate with an increase in palatable forage species. For example, herbage production increased three years following burning in southwestern ponderosa pine forests, but the amount of palatable forage produced was reduced (Oswald and Covington 1984). Data trends from the present study indicated that burning of grasslands increased standing crop of forbs and total understory standing crop, which agrees with previous reports that burning favours forbs over grasses (Daubenmire 1968, Antos et al. 1983). In particular, burning increased forbs such as three-flowered avens (*Geum triflorum* (Pursh)), milk vetch (*Astragalus striatus* (Nutt.)), yarrow (*Achillea millefolium* L.), and pussytoes (*Antennaria spp.* Gaertn.) (Bailey and Anderson 1978). Within forested sites, burning reduced standing crop of graminoids but had no effect on forbs or total understory standing crop. The growth form of bluebunch wheatgrass varies between the forest and grassland plots. Large tussocks of bluebunch wheatgrass in forested areas grew further apart than the smaller but more dense plants in the grassland (Ducherer, personal observation). Resprouting of rough fescue (*Festuca scabrella* Torr.) following wildfire decreased with increasing plant size (Antos et al. 1983). The greater amount of detritus present in basal tufts of larger plants can enhance fire severity and kill the perennating buds (Wright 1970, Johnson 1998). Changes in graminoid abundance may be attributed to changes in light and structural conditions in the forest (Naumburg and DeWald 1999). Therefore, it is possible that plant size, along with changes in growing conditions had an impact on recovery of grasses following burning.

Spring burns have higher potential for killing fine-roots and reducing tree growth (Busse et al. 2000) because root activity is generally greater near the surface in the spring compared to the fall (Grulke et al. 1998). Plants such as rough fescue that typically begin growth early in the season are more easily damaged by spring burning than by fall burning (Bailey and Anderson 1978). Spring burning allows greater recovery time for plants that have sufficient carbohydrate reserves to sprout and resume photosynthesis (Agee 1993). Bitterbrush and Idaho fescue (*Festuca idahoensis* Elmer) were reduced for 5 to 6 years after burning (Busse et al. 2000). However, Busse et al. (2000) found no evidence that prescribed burning would have long-term impacts on stand productivity.

Cattle prefer to graze burned sites, and therefore, prescribed burning can possibly be used control animal distribution (Vermeire et al. 2004). Burning enhanced early season growth of rough fescue, possible due to a warming of the microclimate because of a blackened surface and the absence of insulating litter and standing dead plant material (Redmann et al. 1993). Increased nutrient concentration and a decreased proportion of cell walls in the foliage following burning (Willms et al. 1981) may increase palatability and attract grazers (Wright 1974). It is for this reason that changes in species composition after burning must be interpreted with caution because these changes may be induced by greater grazing and browsing (Oswald and Covington 1984).

Using fire to control trees that have encroached and forest ingrowth is complicated because each fire is a unique event, differing in fuel, topography and weather conditions. Thus, it is difficult to predict the effects of prescribed burning on forage, habitat and range management (Wikeem and Strang 1983). Recent research has shown that landscape dynamics play critical roles in forest ingrowth and tree encroachment in Interior BC (Bai et al. 2004). For instance, south-facing slopes in grasslands at mid-elevations are more susceptible to tree encroachment than north-facing slopes (Bai et al 2004). North-facing slopes most often have closed forests (Bai et al. 2004). Management plans must incorporate topography, species diversity and tree survival initiatives to target areas that are most susceptible to tree encroachment and to achieve desired results.

3.5 Conclusions and Practical Implications

Burning reduced species evenness and diversity in ponderosa pine and Douglas-fir forests. In addition, burning reduced litter cover, litter depth and lichen cover in the forest. Data from the present study indicate burning increased standing crop of forbs and the total standing crop in the understory in the grassland; however, standing crop of graminoids was reduced in the forest. It is concluded that burning effectively controls trees that have encroached onto grasslands, for fuel reduction, and for increasing standing crop of understory plants in the forest range of Interior BC. Extended monitoring is necessary to determine the long-term effects of burning on species diversity and productivity of these ecosystems.

4.0 SHORT TERM EFFECTS OF THINNING ON UNDERSTORY VEGETATION IN FORESTED RANGE IN INTERIOR BRITISH COLUMBIA

4.1 Introduction

Forest ingrowth and tree encroachment are reducing grazing land area in Interior BC. Possible reasons for the expansion of woody species include fire suppression, human disturbance, climatic variation, livestock grazing, and the combination of all of these factors (Bai et al. 2004). The abundance of understory shrubs and herbs were inversely correlated with increasing tree ingrowth in Interior Douglas-fir and ponderosa pine sites of BC (Page 2002). Rangeland health, productivity and carrying capacity are of major concern to forest administrators as well as the ranchers that utilize the lands.

Understanding forest/grassland ecosystems is key to proper management. Yield of rangelands that have both grassland and forested areas is highly dependent on soil water and canopy cover (Dodd et al. 1972). As elevation increases, tree canopy may limit forage production more than water. The presence and structure of the overstory canopy directly influence light reaching of the understory and the productivity understory vegetation and it's structure are the result of competition for light (Dodd et al. 1972, Peltzer et al. 1998, Harrington and Edwards 1999, Aubin et al. 2000). Therefore, increasing density of tree stands reduces carrying capacity (McLean et al. 1971).

Thinning forested areas is a management technique designed to increase plant productivity. Ideally, removing the tree canopy will increase the amount of light reaching understory vegetation, and consequently, increase understory plant cover and species richness (Thomas et al. 1999). Removing the tree canopy can also increase soil temperatures (Strickler and Edgerton 1976) and decrease soil respiration (Streigl and Wickland 1998). In addition, reducing or eliminating competition from overstory trees can increase soil water and mineral nutrients for understory plants (Peltzer et al. 1998) and allow colonization by herbaceous plants (Smit and Rethman 2000). With increasing productivity, however, competition for light and soil resources also increases (Peltzer et al. 1998). Removing small diameter trees from ponderosa pine forest stands reduces subsequent wildfire severity (Pollet and Omi 2002).

Thinning pinyon-juniper (*Pinus edulis* Englem.) - (*Juniperus monosperma* (Engelm.) Sarg.) woodlands increased the cover of native grasses and to a lesser extent, forbs and shrubs (Brockway et al 2002). Plant species richness, biomass, and litter cover increased the most where the thinning was followed by complete removal or scattering of the tree material two years after treatment (Brockway et al 2002). Response of understory species to increased light is not necessarily uniform. For example, after gap formation in a hemlock (*Tsuga canadensis*(L.)) forest, understory species exhibited positive to negative responses; some species responded immediately to gap creation while others responded later in gap succession (Rankin and Tramer 2002).

Disturbance brought on by thinning increase the availability of new microsites for plant establishment and growth, which could possibly lead to increased species richness (Brockway et al. 2002). Reduction in cover following thinning may be a result of the disturbance. For example, cut trees may crush and smother understory vegetation (Thomas et al. 1999). Other research has demonstrated that timber harvesting and extensive site preparation can reduce the amount of surface organic matter (woody residues and forest floor layers) (Jurgensen et al. 1997). Although the effects of tree ingrowth reduce understory species, the effects of disturbances such as thinning can reduce the abundance of many important understory characteristics (Page 2002). Therefore, treatments such as thinning should be applied with caution. In particular, negative responses to thinning included the loss of herbs, pinegrass (*Calamagrostis rubescens* Buckl.) and bryophytes, however, there is no indication that long-term benefits of thinning are outweighed by initial decreases in cover and biomass (Page 2002).

Understory vegetation respond to increased light and therefore the greatest response to the increases in light following thinning are expected within the first several years (Thomas et al. 1999). Factors affecting herbaceous vegetation responses to thinning in longleaf pine (*Pinus palustris* Mill.) plantations were ranked as follows: light > soil > water > herbicides > litterfall (Harrington and Edwards 1999).

The majority of studies have focused on the response of tree species to thinning or gap formation, while fewer studies have examined the long-term effects of thinning on understory plant communities (Thomas et al. 1999). The objectives of this study were to determine the short-term effects of thinning on understory species diversity and standing crop in the ponderosa pine (*Pinus ponderosa* Dougl.) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests of Interior BC. Knowing the effects of thinning may help increase species diversity and increase forage production in grasslands. We hypothesized that the removing trees by thinning increases species diversity and biomass production of the understory vegetation in ponderosa pine and Douglas-fir forests.

4.2 Materials and methods

4.2.1 Site description

Grasslands in BC are classified into three zones according to elevation: lower grassland (*Agropyron-Artemisia*), middle grassland (*Agropyron-Poa*) and upper grassland (*Agropyron-Festuca*) (van Ryswyk et al. 1966). Two pastures, Coal Mine Pasture (121^{o} 35' W 50^{o} 46' N) and Gladys Lake Pasture (121^{o} 33' W 50^{o} 44' N), near Cache Creek in south-central BC, were selected for this study. These study sites are located in the upper grassland zone.

Average precipitation data from the closest weather station in Kamloops was 264 mm in 1998, 268 mm in 1999, 277 mm in 2000, 254 mm in 2001 and 221 mm in 2002 (Environment Canada 2005). The upper grassland zone is characterized by 280 to 330 mm of precipitation with the growing season lasting from April to mid-July; some plants also grow in September (McLean and Marchand 1968). Black soils are dominant in the upper grassland zone (van Ryswyk et al. 1966). The plant association at these sites are Douglas-fir-Idaho fescue (*Fesctuca idahoensis* Elmer), especially between 600 and 900 m (McLean 1970). Plants known to decrease with grazing in this area are bluebunch wheatgrass (*Agropyron spicatum* (Pursh) Scribn. & Smith), Idaho fescue, and rough fescue (*Festuca scabrella* Torr.). Plants that increase with grazing in the upper grassland zone are needle-and-thread (*Stipa comata* Trin. & Rupr.), Junegrass (*Koeleria cristata* (Ledeb.) J.A. Schultes f.), Kentucky bluegrass (*Poa pratensis* L.), pasture sage (*Artemisia frigida* Willd.), dwarf pussytoes (*Antennaria dimorpha* (Nutt.) T. & G.), Sandberg bluegrass (*Poa sandbergii* Vasey), lupine (*Lupinus sericeus* Pursh), timber milk vetch (*Astragalus miser* Dougl. *ex* Hook), and yarrow (*Achillea millefolium* L.). Plant species that are not native to the ecosystem are considered 'invaders'. In the upper grassland zone invaders are downy brome (*Bromus tectorum* L.), dandelion (*Taraxacum officinale* Weber), mullein (*Verbascum thapsus* L.) and fleabane (*Erigeron spp.* L.).

Gladys Lake Pasture is grazed by cattle and it's managed by the BC Ministry of Forests. Grazing was scheduled for July 1 to July 15 in 2000 and 2001 for Gladys Lake Pasture with a stocking rate of 100 animal unit months (AUM) on 231 ha. This schedule was recommended and it was to be overridden by range readiness or by the level of forage utilization. Coal Mine Pasture was used only by wildlife.

4.2.2 Experimental design and treatment design

Five pairs of plots, 40 x 50 m in size, were established at each site in April 1998. One-half of the plots were thinned in October 1998. This study was a randomized complete block design (RCBD) with five replicates and it was repeated at two sites. All trees within the plots with a diameter at breast height (DBH) of $\leq$ 20 cm were cut at ground level by chainsaws, dragged from the plots, piled and burned. Larger trees were left standing to maintain the forest stand. Pre-thinning tree densities for Coal Mine Pasture and Gladys Lake Pasture were similar ($P > 0.05$), 998 and 830 stems ha^{-1}, respectively. Post-thinning tree densities averaged 115 and 73 stems ha^{-1}, respectively, for Coal Mine Pasture and Gladys Lake Pasture.

Experiment 1: Effect of thinning on understory species composition and standing crop

Two transects, 20 m in length, were established at the center of each plot. The first transect was positioned to be representative of the plot and the second transect was placed at the left side and 10 m away from the first one.

Experiment 2: Thinning effects on understory vegetation under and outside of the tree canopy

Four ponderosa pine and four Douglas-fir trees within each plot were selected to study the effect of tree canopy and thinning on understory vegetation. Trees selected had intermediate sizes relative to other trees within each plot, relatively straight stems, and regular (round) and even (equal dimensions along all directions) canopies. Trees with partial canopy overlaps were avoided because they may compromise the influence of individual trees. Regression equations were developed for each tree species to describe the relationship between basal diameter and crown projection area (Table 4.1). Basal diameters of tree stems in the thinned plots that had been removed were also measured. Four Douglas-fir and four ponderosa pine trunks were selected in each thinned plot based on basal diameters that were similar to the control plots (Table 4.2). The crown projection areas of trees removed by thinning were estimated using the regression equations mentioned previously. Four transects along N, E, S, and W were established from the stem to the edge of the crown projection area of each tree (Fig. 3.1). Crown projection area was estimated by visually projecting the edges of the canopy down to the soil surface (Barbour et al. 1999).

4.2.3 Data collection

Experiment 1: Effect of thinning on understory species composition and standing crop

Ten, 0.4 x 0.5 m quadrats, were placed 2 m apart on the right side of each 20 m transect. Percent canopy cover (Barbour et al. 1999) of each vascular plant species, litter (barks, needles, and other dead plant materials), and bare soil were visually estimated in June to July 1998 (before treatment), 1999, and 2002 within these quadrats. The height of live vegetation and the depth of litter within each quadrat were also measured in 3 sub-samples. Standing crop of understory plants was determined in late July 2002 on the left side of each, at 5 and 15 m, in 0.4 x 0.5 m quadrats by hand clipping plants to ground level. Standing crop were oven-dried at 60°C for 12 to 24 hours, and weighed to determine total standing crop. Samples were then sorted into shrubs, forbs, live graminoids, and dead graminoids and weighed separately.

Table 4.1 Regression equations used to describe the relationship between tree basal diameter and crown projection area of ponderosa pine (PP) and Douglas-fir (DF) trees selected for the effect of canopy on understory vegetation at Coal Mine Pasture and Gladys Lake Pasture, BC.

Site	Tree Species	Regression Equation	R^2_{adj}
Coal Mine Pasture	DF	Y = 1.14x + 802	-0.02
	PP	Y = 1.39x + 487	0.77
Gladys Lake Pasture	DF	Y = 3.15x + 28	0.51
	PP	Y = 1.64x + 429	0.43

Table 4.2 Average basal diameter of ponderosa pine (PP) and Douglas-fir (DF) trees selected for the effect of canopy on understory vegetation at Coal Mine Pasture and Gladys Lake Pasture, BC.

Site	Treatment	Tree Species	Basal Diameter (cm)	Range (cm)
Coal Mine Pasture	Control	DF	16.0 a ± 0.93	11.8 – 19.7
		PP	16.2 a ± 0.93	11.1 – 20.4
	Thinned	DF	14.5 a ± 0.83	11.1 – 19.4
		PP	15.0 a ± 0.76	10.8 – 20.1
Gladys Lake Pasture	Control	DF	21.8 a ± 1.59	16.2 – 24.5
		PP	15.6 a ± 0.64	15.0 – 16.2
	Thinned	DF	21.7 a ± 1.42	18.5 – 25.2
		PP	12.9 a ± 2.71	10.2 – 15.6

[1] Means with different letters within a column and site are significantly different at $P \leq 0.05$.

Experiment 2: Thinning effects on understory vegetation under and outside of the tree canopy

Coal Mine Pasture was sampled in August 2001 and July 2002. Gladys Lake Pasture was sampled in July and early August 2001 and in June 2002. From the base of each ponderosa pine tree a transect was laid and four locations were determined: 1) tree bole (Q1); 2) halfway between the tree bole and edge of the crown projection area (Q2); 3) 30 cm inside the edge of the crown projection area (Q3); and, 4) 60 cm outside the edge of the crown projection area (Q4) (see Fig. 3.1). Quadrat 3 (Q3) was not measured when the crown projection area was too small to separate Q2 from Q3. This procedure was repeated in four directions: north, east, south and west. A 0.3 x 0.3 cm quadrat was used for Q1-Q4 to determine percent cover (Barbour et al. 1999) of ground cover and canopy cover of understory species in June 2001 and 2002. Understory standing crop was determined in 0.3 x 0.3 m quadrats by hand clipping plants to ground level. Quadrats in which standing crop was determined, were placed at the left side of each transect, within and outside the tree canopy (at the same location as Q2 and Q4). Standing crop was pooled for Q2 in all four directions, and pooled for Q4 in all four directions. Standing crop were oven-dried at 60^{o}C for 12 to 24 hours, and weighed to determine total standing crop. Samples were then sorted into shrubs, forbs, live graminoids, and dead graminoids and weighed separately.

4.2.4 Data analysis

Experiment 1: Effect of thinning on understory species composition and standing crop

Species cover from quadrats along the two, 20 m transects were pooled for each plot. Species richness (R), species evenness (E) and the Shannon-Weiner Diversity Index (*H*') (Barbour et al. 1999) were calculated for each plot and within years using PC-ORD (McGarigal et al. 2000). Data were then analyzed separately within years using Analysis of Variance (ANOVA) (SAS Institute 1995) to determine the effect of burning treatment on species composition. Data from 1998, 1999 and 2002 were combined for ordination with rare species (species that occurred in ≤ 2 plots or 10% of the plots) being removed (McGarigal et al. 2000). Cover data were relativized by the species maximum and tested with Detrended Correspondence Analysis (DCA). Scores of the first three axes of DCA were calculated and analyzed with ANOVA to determine the effect of year and treatment on plot separation along the 3 axes. The first axis represents the maximum amount of variation possible in a single dimension. The second axis is constrained by orthogonality and maximization of the remaining variance. Therefore, the second axis is statistically independent of the first axis and it explains the maximum remaining variation not explained by the first axis (McGarigal et al. 2000).

Ground cover including rocks, logs, bare soil, litter and lichens, the cover of functional groups including forbs, graminoids, shrubs and total cover, and litter depth and vegetation height were analyzed within years using ANOVA to determine the effects of thinning (Snedecor and Cochran 1980). Graminoids included grasses, sedges and rushes. The shrub category also included trees (<0.5 m tall), but excluded pasture sage (*Artemisia frigida* Willd.) which was considered a forb. Standing crop of forbs, live graminoids, dead graminoids, shrubs, and total understory standing crop were analyzed using ANOVA. Means were separated using the Least Significant Difference test (LSD) (Snedecor and Cochran 1980). A significance value of $P \leq 0.05$ was used.

Experiment 2: Thinning effects on understory vegetation under and outside of the tree canopy

Species composition data along the four cardinal directions from each tree bole were pooled according to quadrat positions (Q1, Q2, Q3 or Q4). Data were combined from trees within each plot (sub-samples), then pooled according to quadrat locations. All three quadrats under the canopy (Q1, Q2 and Q3) were averaged for ground cover and referred to as "under canopy". R, E and *H*' were calculated for under the canopy (using Q2 only) and outside of the canopy (Q4) for each plot and year using PC-ORD and analyzed within years (2001 and 2002) using the General Linear Model (GLM) (SAS Institute 1995). Each combination of canopy cover type (under the canopy or outside the canopy) was treated as a "plot" in ordination after removing species that occurred in only two plots (rare species) (McGarigal et al. 2000). Data were relativized by the species maximum and then subject to DCA separately for each year. Scores of the first three axes of DCA were calculated and analyzed with GLM to determine whether the effect of treatment and canopy cover type on species composition can be separated along the three axes.

Ground cover, the canopy cover of functional groups, litter depth and vegetation height were pooled according to quadrat location within each plot and analyzed within years using GLM to determine the effect of canopy removal and quadrat location. Data of the three quadrats under the tree canopy were then combined and analyzed again using GLM. Standing crop of forbs, live graminoids, dead graminoids, shrubs and total understory standing crop were also analyzed with GLM as described above. Means were separated using LSD (Snedecor and Cochran 1980). A significance value of $P \leq 0.05$ was used.

4.3 Results

4.3.1 Understory plant species composition and diversity

Experiment 1: Effect of thinning on understory species composition and standing crop

R, E and *H'* at both sites were not different ($P > 0.05$) between the control and the thinning treatment before thinning in 1998 (Table 4.3). R was greater in the control than the thinning treatment for Coal Mine Pasture in the first (1999, $P = 0.033$) and fourth (2002, $P = 0.030$) year after thinning. E was similar in the control and the thinning treatment at Coal Mine Pasture in 1999 ($P = 0.130$) and 2002 ($P = 0.132$). H' was greater for the control in 1999 ($P = 0.037$), but not in 2002. None of these measures of diversity were different between the control and the thinning treatment at Gladys Lake Pasture in 1999 and 2002.

Plots were separated along the first three axes of DCA based on plant species composition (Fig. 4.1). For Coal Mine Pasture, plots of the three years were separated by Axes 1 and 2 and the control was separated from the thinning treatment by Axis 3 ($P < 0.001$). The order of year along Axis 1 was 1998, 1999 and 2002 and the order of treatment along Axis 3 was control and thinning. The separation of treatments within years along Axis 3 was only significant in 2002 (P = 0.010), indicating that species composition was similar in the control and thin plots the first year, but it was different between treatments in the fourth year. For Gladys Lake Pasture, treatments were separated by Axes 1 (P = 0.011), from thinning to control along the axis and Axes 3 (P = 0.049), from control to thinning along the axis, but years were not separated by the first three axes (Fig. 4.2).

Coal Mine Pasture was dominated by rough fescue and arrow-leaved balsamroot (*Balsamorhiza sagittata* (Pursh) Nutt.) (Table 4.4). Gladys Lake Pasture was dominated by Kentucky bluegrass but rough fescue and arrow-leaved balsamroot were not common. Bluebunch wheatgrass was of minor importance at both sites. The abundance of individual species fluctuated from year-to-year. The relative importance of pinegrass in the control plots at Coal Mine Pasture tended to be lower in 1999 and 2002 compared to 1998. However, importance of pinegrass in the thinned plots was consistent among the three years. Timber milk vetch increased after thinning at Gladys Lake Pasture.

Experiment 2: Thinning effects on understory vegetation under and outside of the tree canopy

R was greater outside the canopy than under the canopy of Douglas-fir and ponderosa pine in 2001 at Coal Mine Pasture (P = 0.017), and in 2001 and 2002 at Gladys Lake Pasture (P = 0.025 and P = 0.007) (Tables 4.5, 4.6). R was not different between the control and the thinning treatments under or outside the canopy. E was greater outside the canopy than under the canopy only for Coal Mine Pasture in 2001 (P = 0.023), but not in other years or at other sites. *H'* was greater outside the canopy than under the canopy in 2001 at Coal Mine Pasture (P < 0.001) and Gladys Lake Pasture (P = 0.046) and 2002 at Gladys Lake Pasture (P = 0.009). Forb cover was greater after thinning outside the canopy of ponderosa pine at Gladys Lake Pasture in 2001 (P = 0.034) and under Douglas-fir at Coal Mine Pasture in 2002 (P = 0.026). Shrub cover was limited and similar under and outside the canopy and between treatments, averaging 4.5 ± 1.4 % (2001) and 2.7 ± 1.0 % (2002) at Coal Mine Pasture, and 0.4 ± 0.2 % (2001) and 0.2 ± 0.1 % (2002) at Gladys Lake Pasture.

Table 4.3 Thinning effects on species richness (R), species evenness (E) and the Shannon-Weiner Diversity Index (H') at Coal Mine Pasture and Gladys Lake Pasture, BC. Thinning was conducted in October 1998.

Site/Year	Treatment	R	E	H'
Coal Mine Pasture				
1998	Control	20.4 a[1]	0.75 a	2.3 a
	Thinning	18.2 a	0.63 a	1.8 a
	P	0.097	0.099	0.077
1999	Control	21.2 a	0.76 a	2.3 a
	Thinning	13.4 b	0.67 a	1.7 b
	P	0.033	0.130	0.037
2002	Control	23.0 a	0.74 a	2.3 a
	Thinning	17.0 b	0.65 a	1.84 a
	P	0.030	0.132	0.061
Gladys Lake Pasture				
1998	Control	22.0 a	0.80 a	2.5 a
	Thinning	23.6 a	0.75 a	2.4 a
	P	0.347	0.167	0.478
1999	Control	26.2 a	0.77 a	2.5 a
	Thinning	24.4 a	0.73 a	2.3 a
	P	0.360	0.330	0.280
2002	Control	26.8 a	0.78 a	2.6 a
	Thinning	24.2 a	0.74 a	2.4 a
	P	0.180	0.285	0.123

[1] Means with different letters within a column, site and year are significantly at $P \leq 0.05$.

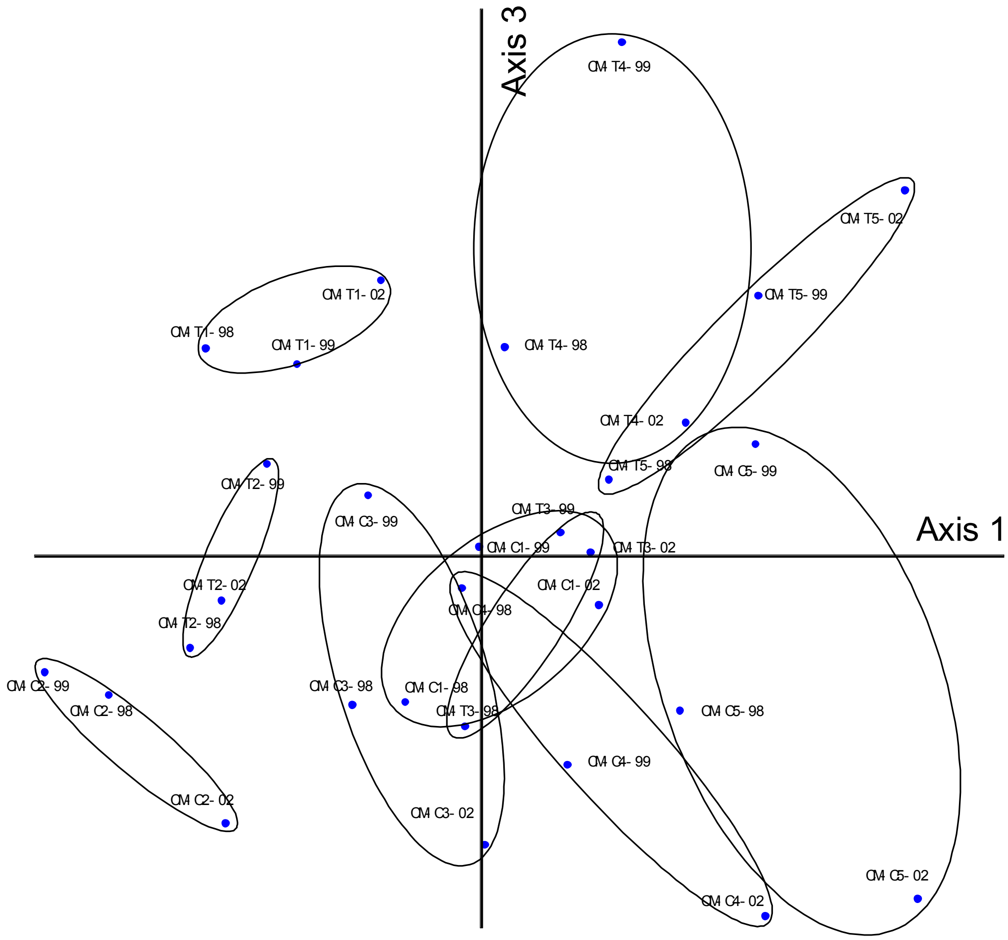

Figure 4.1 Separation of thinned (T) and control (C) plots along Axes 1 and 3 of Detrended Correspondence Analysis (DCA) at Coal Mine Pasture, BC. Repeated measurements of the same plots in 1998, 1999 and 2002 were grouped.

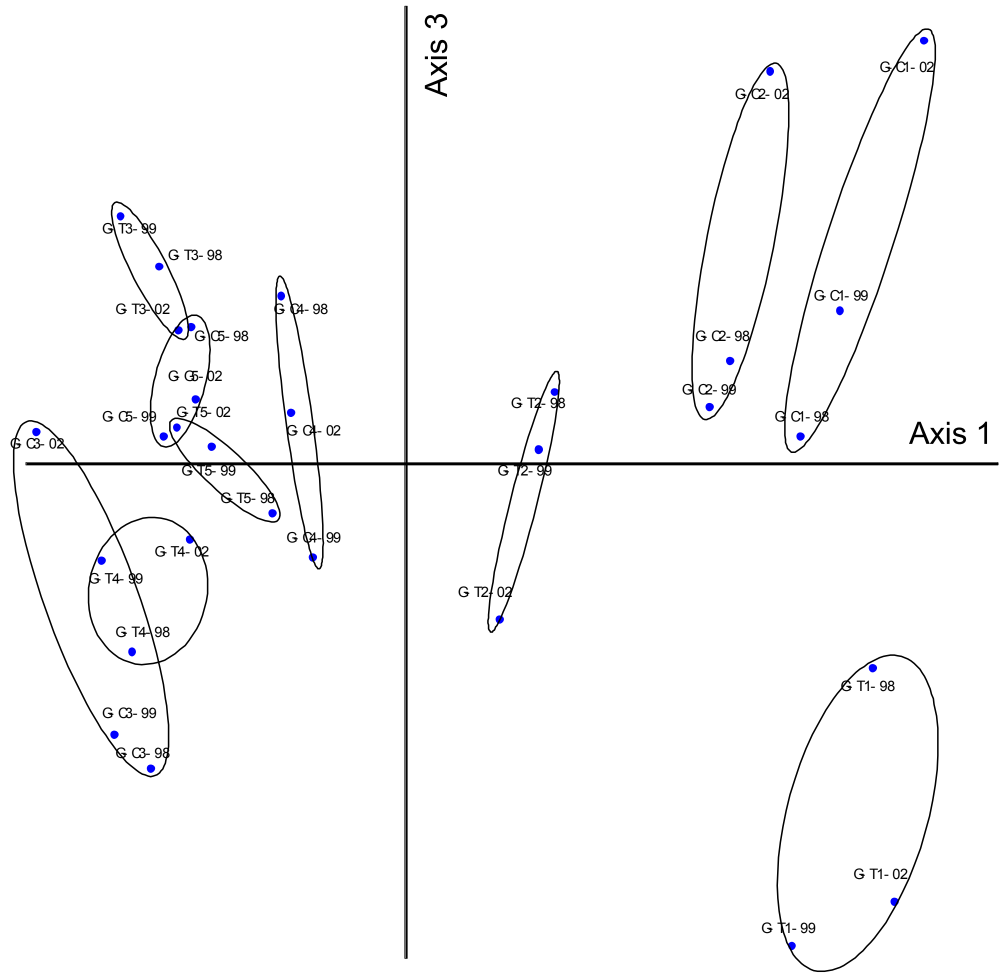

Figure 4.2 Separation of thinned (T) and control (C) plots along Axes 1 and 3 of Detrended Correspondence Analysis (DCA) at Gladys Lake Pasture, BC. Repeated measurements of the same plots in 1998, 1999 and 2002 were grouped.

Table 4.4 Thinning effects on canopy cover (%) of dominant species at Coal Mine Pasture and Gladys Lake Pasture, BC in 1998, 1999 and 2002.

	Control			Thinning		
Site/Species	1998	1999	2002	1998	1999	2002
Coal Mine Pasture						
Agropyron spicatum	2	1	<1	2	1	<1
Arctostaphylos uva-ursi	3	4	3	9	11	7
Astragalus miser	1	2	<1	<1	2	<1
Balsamorhiza sagittata	6	4	6	13	7	9
Calamagrostis rubecens	4	1	2	3	2	2
Festuca campestris	7	5	4	12	8	6
Geum triflorum	3	2	2	1	2	<1
Stipa richardsonii	3	2	2	2	1	1
Gladys Lake Pasture						
Agropyron spicatum	3	4	2	2	2	<1
Antennaria spp.	4	3	5	3	1	3
Astragalus miser	3	3	4	2	5	7
Fragaria virginiana	1	1	1	3	2	3
Geum triflorum	1	1	<1	1	<1	2
Poa pratensis	8	6	5	9	9	8
Potentilla glandulosa	1	<1	4	1	2	2
Stipa richardsonii	3	4	3	10	5	7
Taraxacum officinale	4	2	<1	5	4	2

Table 4.5 Effects of thinning on species richness (R), species evenness (E), the Shannon-Weiner Diversity Index (H') and canopy cover of forbs and grasses under and outside the canopy of ponderosa pine (PP) and Douglas-fir (DF) at Coal Mine Pasture, BC. Thinning was conducted in October 1998.

Year/ location	Tree species/ Treatment	R	E	H'	Forb cover (%)	Graminoid cover (%)
2001						
Under canopy	PP-Control	10.7 a[1]	0.66 a	1.5 a	12 a	7 a
	DF-Control	10.4 a	0.71 a	1.6 a	7 a	7 a
	PP-Thinned	10.0 a	0.64 a	1.5 a	9 a	9 a
	DF-Thinned	11.8 a	0.74 a	1.7 a	11 a	8 a
	Mean	10.7 B[2]	0.69 B	1.6 B	10 A	8 A
Outside canopy	PP-Control	13.3 a	0.77 a	2.0 a	11 a	7 a
	DF-Control	13.6 a	0.75 a	1.9 a	9 a	9 a
	PP-Thinned	12.5 a	0.71 a	1.8 a	13 a	9 a
	DF-Thinned	14.0 a	0.72 a	1.9 a	11 a	7 a
2002	Mean	13.4 A	0.73 A	1.9 A	11 A	8 A
Under canopy	PP-Control	13.1 a	0.69 a	1.8 a	11 ab	5 a
	DF-Control	12.5 a	0.75 a	1.8 a	8 b	5 a
	PP-Thinned	11.4 a	0.67 a	1.6 a	12 a	6 a
	DF-Thinned	14.2 a	0.74 a	1.9 a	13 a	6 a
	Mean	12.8 A	0.71 A	1.8 A	11 A	6 B
Outside canopy	PP-Control	16.3 a	0.79 a	2.2 a	10 a	7 a
	DF-Control	15.2 a	0.72 a	1.9 a	13 a	7 a
	PP-Thinned	14.8 a	0.74 a	2.0 a	15 a	7 a
	DF-Thinned	15.8 a	0.73 a	2.0 a	11 a	6 a
	Mean	15.5 A	0.74 A	2.0 A	12 A	7 A

[1] Means with different lower-case letters within a column, site and year are significantly at $P \leq 0.05$.
[2] means with different capital letters within a column and year are significantly different at $P \leq 0.05$.

Table 4.6 Effects of thinning on species richness (R), species evenness (E), the Shannon-Weiner Diversity Index (H') and canopy cover of forbs and grasses under and outside the canopy of ponderosa pine (PP) and Douglas-fir (DF) at Gladys Lake Pasture, BC. Thinning was conducted in October 1998.

Year/ location	Tree species/ Treatment	R	E	H'	Forb cover (%)	Graminoid cover (%)
2001						
Under canopy	PP-Control	9.7 a[1]	0.76 a	1.7 a	13 a	10 a
	DF-Control	13.4 a	0.75 a	2.0 a	13 a	13 a
	PP-Thinned	18.0 a	0.85 a	2.4 a	23 a	12 a
	DF-Thinned	16.3 a	0.71 a	2.0 a	14 a	18 a
	Mean	14.3 B[2]	0.77 A	2.0 B	16 A	13 A
Outside canopy	PP-Control	14.5 a	0.81 a	2.1 a	14 b	8 a
	DF-Control	15.0 a	0.78 a	2.1 a	19 ab	13 a
	PP-Thinned	20.5 a	0.83 a	2.5 a	23 a	18 a
	DF-Thinned	17.4 a	0.70 a	2.0 a	15 b	21 a
2002	Mean	16.9 A	0.78 A	2.2 A	23 A	15 A
Under canopy	PP-Control	21.0 a	0.83 a	2.5 a	23 a	12 a
	DF-Control	20.5 a	0.78 a	2.4 a	20 a	16 a
	PP-Thinned	12.8 a	0.81 a	2.0 a	26 a	11 a
	DF-Thinned	16.0 a	0.80 a	2.2 a	22 a	18 a
	Mean	17.6 B	0.80 A	2.3 B	23 A	14 A
Outside canopy	PP-Control	24.0 a	0.88 a	2.8 a	24 a	12 a
	DF-Control	21.6 a	0.77 a	2.4 a	30 a	14 a
	PP-Thinned	16.5 a	0.85 a	2.4 a	30 a	10 a
	DF-Thinned	20.3 a	0.83 a	2.5 a	28 a	18 a
	Mean	20.6 A	0.83 A	2.3 A	28 A	14 A

[1] Means with different lower-case letters within a column, site and year are significantly at $P \leq 0.05$.
[2] means with different capital letters within a column and year are significantly different at $P \leq 0.05$.

Dominant species surrounding ponderosa pine and Douglas-fir were similar along the transects at both sites (Tables 4.7 and 4.8 respectively). Richardson's needle grass was less abundant outside the canopy of ponderosa pine, but remained unchanged under Douglas-fir at Coal Mine Pasture (Table 4.7). At Gladys Lake Pasture, timber milkvetch increased after thinning under ponderosa pine, but remained the same under Douglas-fir (Table 4.8).

Table 4.7 Effects of thinning on canopy cover (%) of dominant species under and outside the canopy of ponderosa pine (PP) and Douglas-fir (DF) at Coal Mine Pasture, BC in 2001 and 2002.

	Control				Thinned			
	--------PP--------		--------DF--------		--------PP--------		--------DF--------	
Year/Species	Under	Outside	Under	Outside	Under	Outside	Under	Outside
2001								
Agropyron spicatum	<1	2	<1	<1	2	2	<1	<1
Arctostaphylos uva-ursi	<1	<1	9	9	1	2	5	9
Astragalus miser	<1	2	1	3	<1	<1	<1	<1
Balsamorhiza sagittata	6	4	5	4	2	5	5	4
Festuca campestris	5	2	6	6	4	4	4	3
Geum triflorum	2	2	<1	2	<1	<1	2	1
Stipa spp.	<1	2	<1	1	<1	1	<1	1
2002								
Agropyron spicatum	<1	<1	1	<1	<1	<1	<1	1
Arctostaphylos uva-ursi	<1	<1	1	<1	8	2	3	7
Astragalus miser	1	1	<1	1	2	2	<1	<1
Balsamorhiza sagittata	4	3	6	8	6	8	7	4
Festuca campestris	3	2	2	2	3	4	2	2
Geum triflorum	2	<1	<1	<1	1	2	2	2
Stipa spp.	<1	1	<1	<1	<1	1	2	2

Table 4.8 Effects of thinning on canopy cover (%) of dominant species under and outside the canopy of ponderosa pine (PP) and Douglas-fir (DF) at Gladys Lake Pasture, BC in 2001 and 2002.

	Control				Thinned			
	--------PP--------		--------DF--------		--------PP--------		--------DF--------	
Year/Species	Under	Outside	Under	Outside	Under	Outside	Under	Outside
2001								
Antennaria spp.	<1	1	<1	3	3	2	2	3
Astragalus miser	<1	1	5	7	3	5	3	3
Fragaria virginiana	3	3	3	3	1	2	2	3
Geum triflorum	<1	<1	2	4	1	2	1	<1
Poa pratensis	9	5	6	7	9	9	11	12
Potentilla glandulosa	3	2	2	<1	<1	<1	<1	1
Stipa richardsonii	<1	<1	3	4	2	2	3	4
Taraxacum officinale	2	4	<1	<1	1	<1	1	1
2002								
Antennaria spp.	1	2	3	3	1	3	2	4
Astragalus miser	2	3	2	7	8	5	3	5
Fragaria virginiana	3	4	2	2	3	2	2	2
Geum triflorum	<1	<1	1	1	1	<1	<1	<1
Poa pratensis	5	5	6	5	2	<1	6	7
Potentilla glandulosa	3	2	<1	<1	3	3	2	2
Stipa richardsonii	<1	<1	<1	<1	2	4	2	2
Taraxacum officinale	4	7	3	3	1	1	3	4

4.3.2 Tree canopy and thinning effects on ground cover

Experiment 1: Effect of thinning on understory species composition and standing crop

The cover of rocks and decomposed logs was low and similar between treatments. The cover of bare soil, litter, lichens, total vegetation (overall average = 43%), litter depth, and vegetation height was similar between treatments. Cover for rocks, logs, bare soil, litter cover, litter depth, vegetation height and lichen cover for Coal Mine Pasture 2002 were: 1%, 8%, 4%, 88%, 3.3 cm, 32.0 cm and 5%, respectively. Respectively, the cover for rocks, logs, bare soil, litter cover, litter depth, vegetation height and lichen cover at Gladys Lake Pasture in 2002 averaged: 1%, 2%, 1%, 94%, 2.6 cm, 29.5 cm and 6%.

Experiment 2: Thinning effects on understory vegetation under and outside of the tree canopy

Litter cover was greater under the canopy than outside the canopy at Coal Mine Pasture in 2001 ($P = 0.001$) and 2002 ($P = 0.001$) (Table 4.9), the same response with litter cover was observed at Gladys Lake Pasture in 2001 ($P = 0.017$) (Table 4.10). Under the canopy litter was mainly composed of coniferous needles, while dead forbs and grasses composed litter outside the canopy. Litter cover was greater under ponderosa pine than under Douglas-fir at both sites but the reduction of litter cover after thinning was only significant between tree species at Coal Mine Pasture in 2002 ($P = 0.036$) (Tables 4.9 and 4.10). Litter depth was greater under the canopy than outside the canopy in Coal Mine Pasture in 2001 and 2002 ($P = 0.001$ and < 0.001, respectively) and Gladys Lake Pasture in 2001 and 2002 ($P = 0.004$ and 0.004, respectively). Litter cover and litter depth decreased while lichen cover increased from the center to the edge of the canopy (Fig. 4.3). Generally, thinning reduced litter depth under both tree species at both sites (Tables 4.9 and 4.10). Litter depth under the canopy of Douglas-fir was reduced after thinning in 2001 ($P = 0.026$) and it was reduced beneath the canopy of ponderosa pine in 2002 ($P = 0.005$) at Coal Mine Pasture. Lichen cover was generally greater outside the tree canopy than under the canopy, but the effect of thinning on lichen cover was significant in 2002 at Coal Mine Pasture (P = 0.001). Total vegetation cover was greater outside the canopy at Gladys Lake Pasture in 2002 (P = 0.034). After thinning, vegetation cover was consistently greater outside the tree canopy than under the canopy, and vegetation cover increased after thinning for both tree species between sites and quadrat locations. Except for Coal Mine Pasture in 2001, where plant heights were greater under the canopy than outside the canopy (P = 0.044), vegetation height was not significantly different under or outside the tree canopy.

4.3.3 Standing crop in the understory

Experiment 1: Effect of thinning on understory species composition and standing crop

Trends in the data suggest total understory standing crop after thinning was greater than the control at both sites (Fig. 4.4, P = 0.081 and 0.090 for Coal Mine Pasture and Gladys Lake Pasture, respectively). The standing crop of forbs, shrubs, and grasses were not significantly different between treatments at both sites (P > 0.05).

Experiment 2: Thinning effects on understory vegetation under and outside of the tree canopy

Total understory standing crop was between 60 and 140 g m^{-2} for ponderosa pine and Douglas-fir trees, respectively (Fig. 4.5). Total understory standing crop was generally greater after thinning compared to the control under both trees. Greater total understory standing crop after thinning at Gladys Lake Pasture was caused by increased forb standing crop in 2001 (P = 0.009) and both greater forb (P = 0.002) and grass (P = 0.057) standing crop in 2002. Greater total understory standing crop after thinning at Coal Mine Pasture was caused by greater shrub standing crop in 2001 (P = 0.017). Differences between tree species of understory standing crop were not significant (P > 0.05).

Table 4.9 Effects thinning on cover of litter cover, litter depth, lichen cover, total vegetation canopy cover, and vegetation height under and outside the canopy of ponderosa pine (PP) and Douglas-fir (DF) at Coal Mine Pasture, BC. Thinning was conducted in October 1998.

Year/ Location	Tree species/ Treatment	Litter (%)	Litter depth (cm)	Lichen (%)	Vegetation (%)	Vegetation height (cm)
2001						
Under canopy	PP-Control	97 a[1]	4.3 a	<1 a	20 a	29.7 a
	DF-Control	90 a	3.1 b	3 a	16 a	27.3 a
	PP-Thinning	95 a	3.7 ab	2 a	26 a	32.8 a
	DF-Thinning	94 a	2.7 c	2 a	24 a	29.6 a
	Mean	94 A[2]	3.4 A	2 A	22 A	39.8 A
Outside canopy	PP-Control	87 a	2.3 a	6 a	19 a	27.5 a
	DF-Control	80 a	2.0 a	6 a	20 a	25.3 a
	PP-Thinning	87 a	2.9 a	5 a	31 a	29.5 a
	DF-Thinning	83 a	2.3 a	8 a	27 a	24.9 a
	Mean	84 B	2.4 B	6 A	24 A	26.8 B
2002						
Under canopy	PP-Control	96 a	4.8 a	1 a	16 a	35.7 a
	DF-Control	89 b	3.5 b	4 a	13 a	30.4 a
	PP-Thinning	92 ab	3.2 b	3 a	25 a	32.1 a
	DF-Thinning	90 b	3.0 b	5 a	22 a	31.2 a
	Mean	92 A	3.6 A	3 B	19 A	32.3 A
Outside canopy	PP-Control	79 a	2.6 a	12 a	18 a	33.4 a
	DF-Control	78 a	2.1 a	8 a	20 a	31.5 a
	PP-Thinning	82 a	2.6 a	12 a	23 a	31.1 a
	DF-Thinning	79 a	2.5 a	10 a	24 a	30.9 a
	Mean	80 B	2.4 B	10 A	21 A	31.7 A

[1] Means with different lower-case letters within a column, year and quadrat locations are significantly different at $P \leq 0.05$.
[2] means with different capital letters within a column and year are significantly different at $P \leq 0.05$.

Table 4.10 Effects thinning on cover litter cover, litter depth, lichen cover, total vegetation canopy cover, and vegetation height under and outside the canopy of ponderosa pine (PP) and Douglas-fir (DF) at Gladys Lake Pasture, BC. Thinning was conducted in October 1998.

Year/ location	Tree species/ Treatment	Litter (%)	Litter depth (cm)	Lichen (%)	Vegetation (%)	Vegetation height (cm)
2001						
Under canopy	PP-Control	100 a[1]	3.2 a	<1 a	24 a	32.4 a
	DF-Control	98 a	32.8 a	1 a	27 a	27.3 a
	PP-Thinning	99 a	2.2 a	<1 a	35 a	26.8 a
	DF-Thinning	97 a	2.0 a	2 a	32 a	27.0 a
	Mean	98 A[2]	2.5 A	1 A	29 A	28.4 A
Outside canopy	PP-Control	99 a	1.9 a	<1 a	25 a	30.7 a
	DF-Control	76 a	2.0 a	3 a	32 a	21.8 a
	PP-Thinning	96 a	2.1 a	4 a	42 a	28.1 a
	DF-Thinning	96 a	2.0 a	4 a	36 a	26.9 a
	Mean	92 B	2.0 B	3 A	33 A	26.9 A
2002						
Under canopy	PP-Control	98 a	4.1 a	1 a	35 a	35.4 a
	DF-Control	97 a	3.1 a	3 a	35 a	29.8 a
	PP-Thinning	97 a	2.4 a	2 a	38 a	28.5 a
	DF-Thinning	91 b	2.2 a	9 a	41 a	27.9 a
	Mean	96 A	2.9 A	4 A	37 B	30.4 A
Outside canopy	PP-Control	95 a	2.2 a	7 a	36 a	32.3 a
	DF-Control	89 a	2.2 a	11 a	44 a	29.8 a
	PP-Thinning	95 a	2.4 a	8 a	39 a	26.5 a
	DF-Thinning	90 a	2.0 a	12 a	46 a	27.8 a
	Mean	92 A	2.3 B	10 A	41 A	29.1 A

[1] Means with different lower-case letters within a column, year and quadrat locations are significantly different at $P \leq 0.05$.
[2] means with different capital letters within a column and year are significantly different at $P \leq 0.05$.

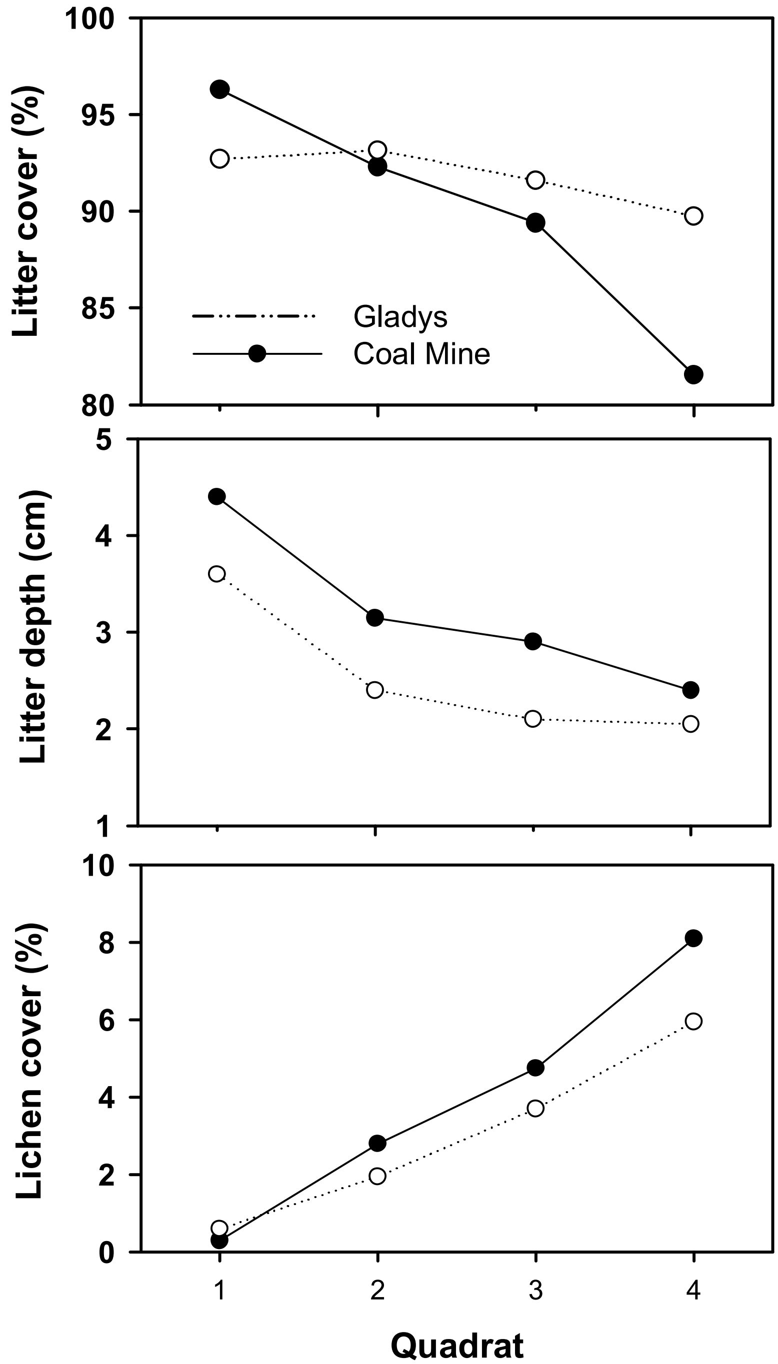

Figure 4.3 Effect of thinning on litter cover, litter depth and lichen cover from the center to the edge of canopy at Coal Mine Pasture and Gladys Lake Pasture, BC in 2001 and 2002.

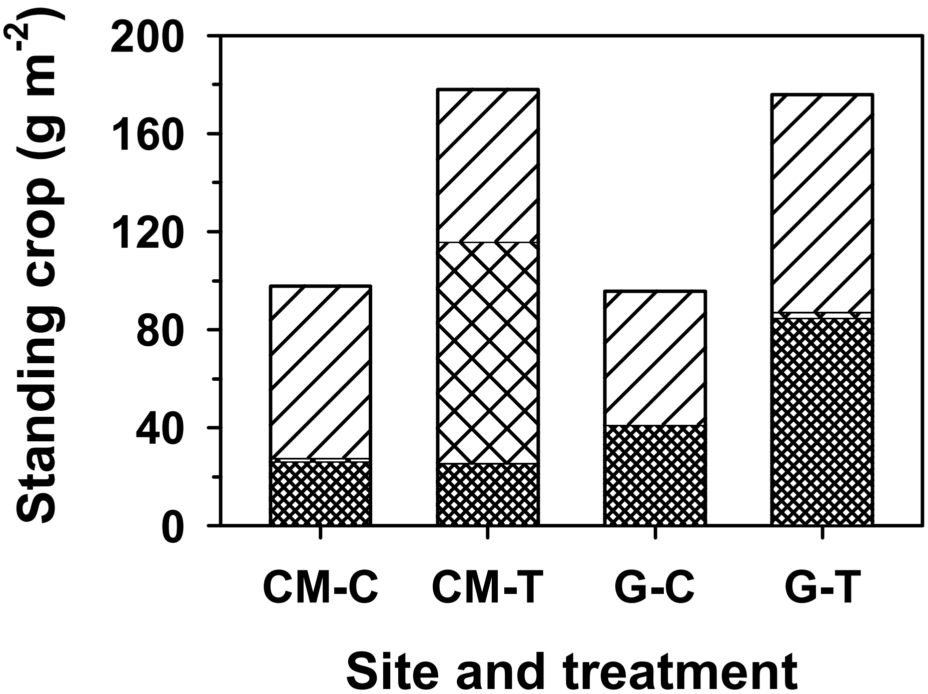

Figure 4.4 Comparison of understory standing crop between thinned (T) and control (C) at Coal Mine Pasture (CM) and Gladys Lake Pasture (G), BC in 2002. The order of stacked bars is forbs, shrubs, and grasses from the bottom to the top.

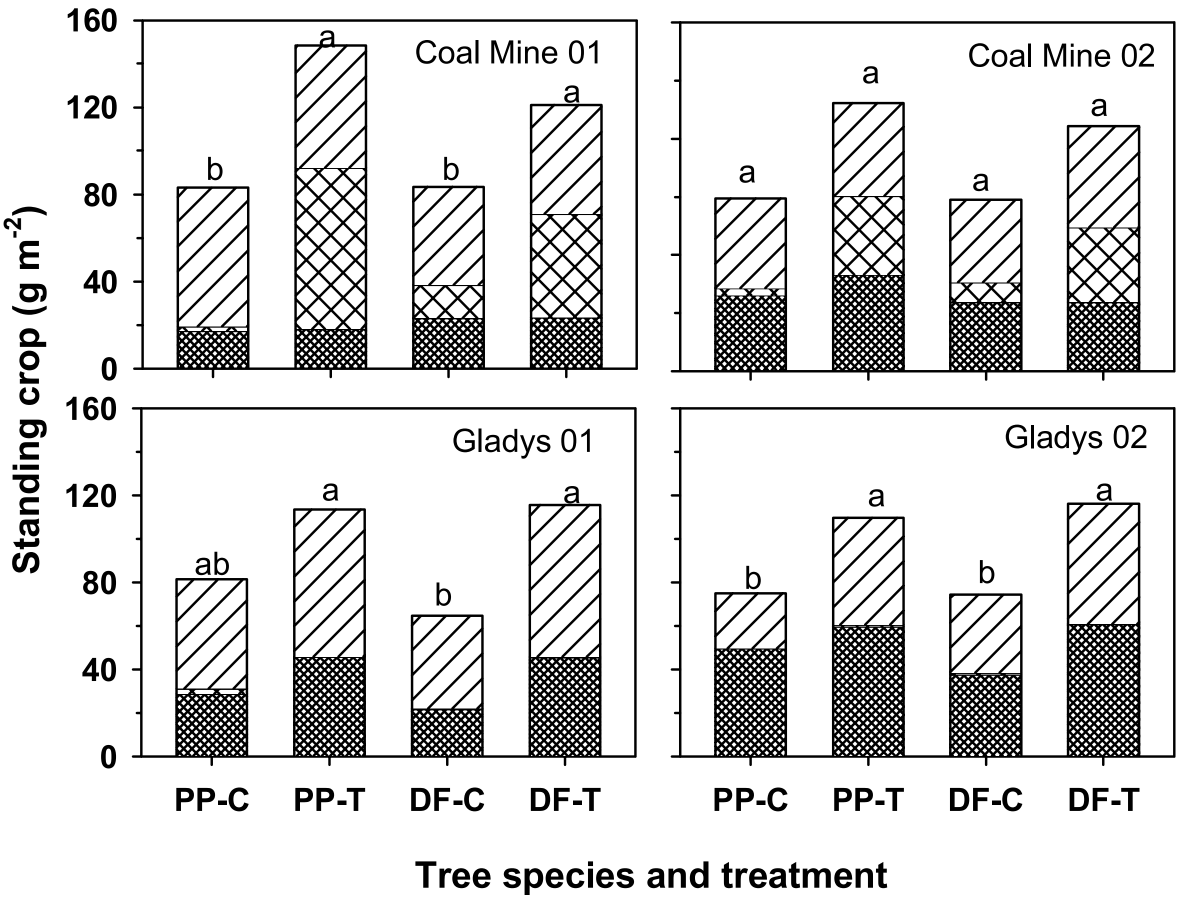

Figure 4.5 Comparison of understory standing crop between thinned (T) and control (C) at Coal Mine Pasture (CM) and Gladys Lake Pasture, BC (G) in 2001 and 2002. Data of under and outside canopy were pooled according to tree species. The order of stacked bars was forbs, shrubs, and grasses from the bottom to the top. PP: ponderosa pine, DF: Douglas-fir. Means with different lower-case letters within tree species are significantly different at P ≤ 0.05.

4.4 Discussion

Previous references suggested that thinning trees enhances species diversity, but the increase is usually more than 10 years after thinning. Examples of delayed response to thinning include: thinning of forests in Washington increased species richness 12 to 16 years later (Thomas et al. 1999); species richness and cover were greater 10 to 24 years after thinning in western Oregon (Bailey et al. 1998). Responses observed in the present study indicate short-term effects of thinning on species diversity are not consistent. Even though species richness and diversity were generally less under the canopy of ponderosa pine and Douglas-fir trees than areas outside of the canopy, thinning had little effect on these plant community characteristics the first 4 years after treatment. Thinning either reduced species richness and diversity of understory vegetation or it had no effect. When Douglas-fir forests were repeatedly thinned, understory cover initially decreased one year after thinning, but it recovered and species richness increased within three years (Thysell and Carey 2001). Species richness increased two years after thinning in pinyon-juniper woodlands in central Mexico (Brockway et al. 2002). Low precipitation, minimum disturbance to the soil during thinning, and relatively low tree density before thinning in the dry, ponderosa pine and Douglas-fir forests in Interior BC may have contributed to the response of understory species to the thinning in the present study.

Relatively stable species richness but increased cover of native grasses, was reported 5 to 12 years following thinning in the ponderosa pine-bunchgrass forests of Arizona (Griffis et al. 2001). Increases in the abundance of graminoids after thinning are related to increased amounts of light in ponderosa pine forests (Naumburg and DeWald 1999). Forb and grass cover increased by 13 and 8%, respectively, two years following thinning in longleaf pine (*Pinus palustris* Mill.) plantations (Harrington and Edwards 1999). At both Coal Mine and Gladys Lake Pastures, cover of forbs, grasses, and shrubs generally increased after thinning. Bailey and Tappeiner (1998) also reported an increase in shrub cover 10 to 25 years following thinning in Douglas-fir forests. Light intensity increases after thinning ponderosa pine forests (Pase 1958). Changes in light intensity were not measured in this study and therefore can not be used to explain whether or not the response of understory species richness and abundance was due to light or some other factor.

Thinning of trees generally increases biomass productivity of understory plants (McConnel and Smith 1970, Uresk and Severson 1998, Brockway et al. 2002). Total understory standing crop increased up to 80% within 3 to 4 years after thinning in the ponderosa pine and Douglas-fir forests in the present study. Depending on sites and years, standing crop of one or more functional groups, including forbs, shrubs, or grasses, may increase. Uresk and Severson (1998) also reported that eliminating or reducing the overstory in ponderosa pine forests increases plant production.

Overstory canopy partially determines litter cover and litter depth. Litter cover, mainly ponderosa pine and Douglas-fir needles, decreased from tree stems toward the canopy edge. Thinning reduced the thickness of the litter because needle sources were removed and needles decomposed (Bai et al. 2000).

Crown structure differs between ponderosa pine and Douglas-fir, the dominant overstory species at the study sites. Ponderosa pine has much more open and elevated canopy than Douglas-fir. Crown structure affects light reaching the understory and influences species richness and abundance (Fornwalt et al. 2003). The impact of thinning on species richness and composition is more dramatic as the tree canopy closes, in limestone grasslands (Dzwonko et al. 1998). In the present study, understory species diversity, abundance and understory standing crop were mostly similar under the two trees. Differences in crown structure between ponderosa pine and Douglas-fir were expected to impact species composition beneath the canopy. Characteristics of the stand including crown structure, density, and canopy closure may have been more similar than they were different, but because light, soil water, and nutrients were not measured, we are unable to quantify the differences. The relatively short time after thinning, along with a large spatial variability between the two sites and among blocks within sites could explain the responses of species diversity and standing crop in this study. The two sites differed in elevation, which in turn affects temperature and precipitation (van Ryswyk et al. 1966, Peet 2000), and understory species composition. Coal Mine Pasture is at a lower elevation and precipitation may be more limiting to plants. Differences in land use between Coal Mine Pasture and Gladys Lake Pasture are reflected in species composition. The dominance of Kentucky bluegrass at Gladys Lake Pasture reflects its grazing history because the grass is an increaser species (McLean and Marchand 1968).

Increaser and invader plants became more abundant and decreasers declined after thinning at Gladys Lake Pasture while the opposite responses were observed at Coal Mine Pasture. At Coal Mine Pasture, plant species responses to thinning included greater cover of increasers and decreasers under the canopy of both tree species.

However, in the control at Coal Mine Pasture, there was greater cover of increasers and decreasers outside the canopy. At Gladys Lake Pasture, following thinning, the greatest cover of increasers was beneath the canopy. The greatest cover of invaders for both tree species was found in the control at Gladys Lake Pasture. Outside the canopy, thinned plots the highest cover of increaser and decreasers at Gladys Lake Pasture.

Plant responses to thinning in this study were variable. The cover of invaders was expected to increase after thinning. For example, Thysell and Carey (2001) demonstrated exotic species were more abundant within 1 and 3 years following thinning in Douglas-fir forests. A similar response in old and young Douglas-fir forests was observed 10 to 24 years following thinning (Bailey et al. 1998). In the present study the greatest cover of invaders was observed in the control plots.

Spatial variability within sites was reflected in the heterogeneity within each plot and the overwhelming, inherent effect of block. While the heterogeneity within plots was addressed by sampling along the 20 m transects and comparing areas under and outside the canopy, more blocks or replicates would be recommended for future studies in forested range when feasible.

4.5 Conclusions and practical implications

The hypothesis that thinning increases species diversity within a relatively short period following was rejected. However, standing crop increased and coniferous litter decreased after thinning. The cover of forbs, grasses and shrubs generally increased after thinning. Thinning can control trees that have encroached in BC grasslands and enhance plant production, both of which contribute to ecosystem health. Thinning is a useful alternative to prescribed burning.

5.0 GENERAL DISCUSSION AND CONCLUSIONS

Knowledge of the causative factors of tree encroachment and forest ingrowth including fire suppression, human disturbances, climatic variations, and livestock grazing, are increasing, but unless increased awareness is coupled with action, tree encroachment and forest ingrowth will still remain a problem. Forest ecologists acknowledge the importance of fire for maintaining species diversity and ecosystem health, and foresters are making efforts to use prescribed burning when feasible and appropriate (BC Ministry of Forests 2005b). Expanding rural developments are reducing rangelands, thus forcing land managers to utilize less land more efficiently and effectively. Changes in land use may mask the influence that weather conditions can have on the regeneration of forest species (Savage and Swetnam 1990). Climate warming over the past 150 years appears to have increased tree growth and forest expansion into adjacent ecosystems in the tundra of Alaska (Suarez et al. 1999).

Prescribed burning is used to kill small-diameter trees, thereby reducing fuel, and raising the height of the forest crown (Fule et al. 2002). Thinning of smaller trees reduces canopy closure and the hazards of crown fires (Fule et al. 2002). These treatments attempt to mimic the natural disturbance regime of fire-dependent ecosystems. Ideally, woody plant control, either through fire, chemical or mechanical means is used to create a mosaic of successional habitats that will maintain or promote species richness and diversity at the landscape level (Fullbright 1996).

Careful consideration must be given when evaluating ecosystem management and allowing fire or thinning to be used to control tree encroachment and forest ingrowth. Controlled burning tends to be less challenging for managers than deliberately fires burn because there is less risk of escape and a different perception associated with man-made fires (Fule et al. 2004). There is a natural tendency to fear fire because it is difficult to control (Wright and Bailey 1982). Careful measurements and understanding of ecological affects following burning must demonstrate the usefulness of burning as a tool in restoring fire-dependent ecosystems to gain public and administrative support (Fule et al. 2004). Disadvantages of thinning can include road construction, soil compaction, logging damage, and introduction of exotic species (Fule et al. 2004). The advantages and disadvantages of the proposed treatments must be weighed against the consequences of imposing no treatment. In the situation of ponderosa pine forests, forest ingrowth and tree encroachment occur because of mismanagement and inaction.

The characteristics of the sites used in these experiments are good examples of why different treatments are required to address tree encroachment and ingrowth. Before burning at Dew Drop, litter had accumulated, mainly pine needles, but there were few downed trees. Burning in similar sites allows relatively cool fires with little damage to tree crowns because mainly surface fuels burn. Coal Mine Pasture and Gladys Lake Pasture would not be ideal for a burning. These sites had large amounts of litter and downed trees. The forest was very dense and the risk of a crown fire would have been high. Thinning is a more appropriate treatment in areas where it is difficult to conduct and control a prescribed burn.

Treatments used to control tree encroachment and forest ingrowth are aimed at increasing ecosystem health and diversity by reducing the tree density and increasing the abundance of understory vegetation. Burning reduced the number of small trees within the ponderosa pine forest. Although species evenness and the Shannon-Weiner Diversity Index were reduced following burning, previous research (Fraas et al. 1991, Busse et al. 2000) suggested species evenness and species diversity would increase. Short-term responses to burning are not necessarily the same as the long-term responses (Reigel et al. 1995). For example, shrub and grass cover was reduced immediately following burning, but this cover recovered 3 years following burning in the present study. Reducing litter by burning also reduced fuel loads, which in turn may minimize the severity and damage of future fires. Trends in the data suggest burning increased standing crop of forbs and total understory standing crop in the grassland, but burning reduced graminoid standing crop in the forest. Although burning generally favours forbs over grasses (Daubenmire 1968, Antos et al. 1983), it is possible that other factors such as plant size and changes in growing conditions may have slowed the recovery of grasses in the present study.

Understory standing crop tended to increase after thinning. Enhanced species diversity in response to thinning was not observed in study even though previous studies suggested it would (Bailey et al. 1998, Thomas et al. 1999). In previous studies, plant responses to thinning were, however, evaluated 10 to 24 years following treatment. Therefore, the amount of time elapsed, combined with the effects of low precipitation, minimal soil disturbance during thinning and relative low tree density all contributed to the plant response to thinning in the present study. Data trends indicate total understory standing crop increased following thinning, which is in agreement with previous research (McConnel and Smith 1970, Uresk and Severson 1998, Brockway et al. 2002). Reducing the cover of trees, coupled with reduced litter depth, thinning reduces the fuel available for a fire.

The second experiment reported here evaluated the effects of tree canopy removal on a small scale by restricting the sampling area directly beneath the tree canopy. Increased humidity, decreased temperature and increased nutrient availability at the tree canopy edge could result in higher herbaceous vegetation (Fuhlendorf et al. 1997). Litter cover decreased from the tree stem toward the canopy edge in the present study. Decreased vegetation cover and increased litter beneath the tree canopy were expected. Differences in understory species diversity, species abundance and understory standing crop between tree species were expected because ponderosa pine and Douglas-fir trees have different crown structures. Plant response under both trees was similar in this study.

The influence of controlling trees on plant community composition and ecological processes depend on factors such as the treatment used, composition of the plant community before treatment, soil types, and rainfall before and after treatment (Fullbright 1996). Effects of fire on soil nutrients depend on the structure of the tree stand and fuel characteristics (Gillon et al. 1999). Effects of fire on litter reduction, soil nutrients, and nutrient composition of vegetation in Aleppo pine (*Pinus halpensis* Mill.) forests, could not be replicated even though fires were conducted using the same technique, in the same forest, on the same day, under the same weather conditions (Gillon et al. 1999). In the present study, burning and thinning removed smaller trees and opened the canopy. The cover of grasses was increased outside the canopy following burning. Changes in species richness and diversity are related to spatial and temporal scales (Fullbright 1996), therefore, burning or thinning may not increase biodiversity immediately. For example, controlling woody plants may reduce species richness and diversity at the patch level, but may increase these parameters at the community or landscape level (Scheiner 1992).

It is difficult to design efficient sampling plans for monitoring understory vegetation response to treatments such as fire and thinning (Abella and Covington 2004). Studying differences in soil water, soil nutrients, light intensity and light quality, would enable the identification and quantification of environmental factors controlling plant growth. Post-treatment sampling in the present study was relatively early compared to other studies in which the effects of treatments were evaluated 10 or more years later. Changes in understory vegetation between the controls and treatments may or may not be more evident later.

This research investigated the response of understory plants to burning and thinning in ponderosa pine and Douglas-fir ecosystems. Both treatments were aimed at increasing species diversity and plant productivity while attempting to mimic natural disturbances. Burning and thinning are effective in eliminating small trees, thus reducing forest ingrowth and encroached trees. Burning reduced litter in bunches of grasses, while thinning removed trees. Burning and thinning also reduced the amount of litter. The initial hypotheses that burning and thinning increase species diversity and increase total understory standing crop must be rejected. However, trends in the data indicate both treatments increased species diversity and total understory standing crop. This research demonstrated that uniform responses from natural ecosystems are unlikely and many factors influence the response of plant communities. Therefore, future studies and management should consider that burning and thinning will not elicit same responses observed in this project or any other project. A major component determining ecosystem response is time. Responses of plant communities in the present study suggest that a longer time between treatment and evaluation may reveal different responses. In conclusion, this study demonstrated that burning and thinning are effective for controlling trees that have encroached, forest ingrowth, reducing fuels, and generally increasing understory standing crop.

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