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University of Alberta

Shrub Encroachment onto Floodplain Meadows within the Rocky Mountain Forest Reserve, Alberta

by

Angela Marie Burkinshaw ©

A thesis submitted to the faculty of Graduate Studies and Research in partial fulfillment

of the requirements for the degree of Master of Science

In

Rangeland and Wildlife Resources

Department of Agricultural, Food and Nutritional Science

Edmonton, Alberta

Spring, 2005

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ABSTRACT

Land managers are concerned with an apparent increase in the distribution and abundance of shrubs in valley bottoms of southwestern Alberta, Canada. This research used digitized aerial photograhps and GIS to quantify spatial grassland meadow loss within a portion of the Clearwater River drainage of the Rocky Mountain Forest Reserve, between 1958 and 1998. Additionally, a field study explored key ecological relationships between the encroaching shrub layer and herbaceaous understory. Grasslands decreased 49% leading to the loss of 1,482 AUMs of primary carrying capacity. Herbage production decreased non-linearly in 2002 from 6,629 $kg.ha^{-1}$ in grasslands (< 12% shrub cover) to 2,797 $kg.ha^{-1}$ in shrubland (> 33%), representing a 58% decline. Similar reductions were evident in native bunchgrass cover and density. These results suggest that to minimize conflict between wildlife and livestock, and maintain biodiversity, grassland restoration should be considered and shrub control be conducted prior to shrub canopy cover levels of about 20%.

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ACKNOWLEDGEMENTS

Great appreciation and thanks are extended to Dr. Edward Bork for his dedication and support throughout this degree. Thank you to Dr. P.H. Crown, Dr. R.J. Hudson, and Dr. E.H. Merrill for their comments and suggestions for planning and developing this research, and the time they spent reviewing this thesis. Also, thank you to Public Lands and Forest Division in particular Therese Morris and Ian Dunk for their GIS expertise, and Butch Shenfield for initially supporting my education leave, and Brian Laing and Barry Adams who supported me in finishing. As well, thanks to Alberta Conservation Association, in particular Cal McLeod and Kim Livingston, for their invaluable financial support and input into the development and collection of this research.

A special thanks to Brent Finnestad who assisted me in my data collection and to my sister and my parents, who also helped with data collection and for their encouragement and support. I'd like to express my gratitude towards my Husband Jon who not only helped me with field data collection and editing but who was always there for me and seen this through with me. I also greatly appreciate all the help Dr. Laki Goodewardene and Dr. Peter Blenis gave in regards to the statistical analysis of this research.

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1. Introduction

1.1 Background

Quantitative assessments of landscape change indicate a substantial increase in woody plants has occurred within grasslands over the last 50 to 300 years worldwide, including Africa (Kelly and Walker 1976, Jeltsch et al. 1994, Eckhardt et al. 2000, Roques et al. 2001), Australia (Walker and Gillison 1982, Harrington et al. 1984), and North and South America (Buffington and Herbal 1965, Blackburn and Tueller 1970, Bragg and Hulbert 1976, Knight et al. 1994, Mast et al. 1997, Driese et al. 1997, Fent and Richard 1999, Kitzberger and Veblen, 1999, Quinlan 1999, Bai et al. 2001). These physiognomic shifts have been attributed to several factors including climate change (Hastings and Turner 1965, Neilson 1986, Sturm et al. 2001), cattle grazing (Humphrey 1958, Duwiddie 1977, Jeltsch et al. 1997) and fire suppression (Cook et al. 1994, Kay et al. 1994, Bork et al. 1996, Kitzberger and Veblen 1999). Although encroachment of woody plants onto grasslands has been widely acknowledged, the rates, patterns, and dynamics of the process have seldom been quantified (Bragg and Hulbert 1976, Archer et al. 1988, Knight et al. 1994, Archer 1995).

In North America, fire suppression efforts during the past 75 years have favoured the establishment of shrub communities (Roughton 1972). Anthropogenic changes to the fire regime, mainly suppression of once frequent fires, has allowed for the expansion of gallery forests into the tallgrass prairie ecosystem (Kucera 1960, Blan 1970, Bragg and Hulbert 1976, Knight et al. 1994), grassland to shrubland conversion in the Rio Grande Plains of southern Texas (Archer et al. 1988), and the expansion of shrub and conifer in the Rocky Mountains (Johnston and Smoliak, 1968, Adams et al. 1992, Kay et al. 1994, Bork et al. 1996).

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In Alberta, the Rocky Mountain Forest Reserve (RMFR) covers much of the eastern slopes of the Rocky Mountains and is managed by Public Lands & Forest Division (PLFD), Sustainable Resource Development (SRD). This is a multiple use area with a wide variety of commercial uses occurring on the same land base, including domestic livestock grazing, timber harvesting, oil and gas development, commercial trail riding, guiding and outfitting. In addition, this area supports many different recreational activities such as random camping, fishing, horseback riding, hiking, mountain biking, and off-road vehicle use. A large portion of the RMFR lies within the Upper Foothills subregion, which is characterized by a mosaic of predominately closed canopy lodgepole pine forest found on undulating hillsides. Valley bottoms, and in some cases, south-facing slopes are occupied by variable amounts of grassland (Willoughby 2001). The meadows of grassland and shrubland associated with these valley bottoms are the main focus of this thesis.

Over the last several decades, land managers have been concerned with an apparent increase in both the spatial distribution and local abundance of shrubs occurring on valley bottoms along a number of major drainages in the forest reserve of southwestern Alberta (Johnston and Smoliak 1968). Many applied field projects have been undertaken in an attempt to reduce these shrubs. In particular, much attention has been directed towards the abundance of bog birch (*Betula glandulosa*. Michx.) in the Upper Foothills, as it is the dominant shrub associated with this region. Most published studies have addressed work undertaken in the region west of Rocky Mountain House. Mechanical clearing of bog birch has taken place on Ribbon Flats, located within the Panther Corners Forest Land Use Zone (MacCallum and Yakimchuk 1991). William de Groot studied the effects of fire severity and season of burn on bog birch growth dynamics on a small grassland meadow in west central Alberta (de Groot and Wein 1999). Darrell Smith and Edward Bork used prescribed burning to reduce bog birch on 7-Mile Flats along the Clearwater River (Bork et al. 1996). The overall objective of these projects has been to make more forage

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available for wildlife and livestock, as bog birch is not a particularly palatable shrub for ungulates and has a chemical defense mechanism that is initiated at the onset of browsing (de Groot 1998).

To date, the notion that shrub encroachment has occurred in these areas is based largely on published literature and repeat photography (Kay et al. 1994, Rhemtulla 1999) documenting what these areas once looked like compared to the present. In the past, Native Americans used fire in the area, particularly on south-facing slopes to manipulate vegetation and bison grazing patterns (White 2000). Burned areas attracted ungulates and fur-bearers, acted as fire guards against wildfire, and opened up valleys to establish villages (Lewis 1978). The fire return interval in the region varies from as frequently as less than 25 years, to infrequent light surface fires greater than 25 years apart or infrequent severe surface fires during drought (de Groot et al. 1999). Fire suppression policies over the last century are believed to be the major factor affecting the increase in shrubs (Bork et al. 1996).

Naturally occurring grasslands occur all over the world and are important from a conservation perspective by supporting both plant and animal biodiversity (Watkinson and Ormerod 2001). In 1949, the rangeland reference program was initiated by Alberta Forest Service to monitor trends in range condition in the RMFR (Hanson 1975). Vegetational species composition along permanent transects, established inside and outside livestock exclosures, have been monitored over the last thirty to forty years: several exclosures show an increase in shrub cover (Willoughby 2000). A 25% increase in shrubs over a 20-year period has occurred in the Ram River Exclosure while the Elk Creek Exclosure shows an increase of 15%. The general trend among exclosures is that species diversity is greater outside the exclosures where grazing occurs and shrub canopy cover is lower (Willoughby 2000), presumably due to the browsing of shrubs.

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Shrub establishment shades the understory and its associated competitive impacts may change vegetational species composition (Willoughby 2001). Changes in vegetation from open grassland meadows to shrub invaded communities significantly reduces the amount of forage available for elk and domestic livestock grazing (de Groot 1998), as well as its accessibility to herbivores by reducing the area of primary range. A decrease in available forage means recent efforts to increase the elk population in the area through Fish and Wildlife's relocation program (Jim Allen ASRD Wildlife Biologist, pers.comm. 2002) may be less feasible, as well as increase the risk of degrading the condition of remaining open meadows. In addition, total allowable carrying capacities set for livestock grazing by PLFD may have to be reduced in the future to align forage supply with year long forage demands. Perhaps most important, further shrub encroachment may continue to increase competition among herbivores for forage and create conflict between resource users. In addition to the loss of forage, shrub encroachment onto grasslands is considered responsible for altering the composition of rangelands (Adams et al. 1992).

This research addresses both the spatial extent of shrub expansion in a portion of the RMFR, as well as quantifies the synecological impacts of encroachment on understory (herbaceous) plant community characteristics, including species composition, diversity, and forage production. This project therefore has both a landscape and plant community focus, which will be addressed using geographic information system (GIS) and field studies, respectively. The fundamental question relating to the landscape component is: What is the area of grassland encroached by shrubs between 1958 and 1998 within a portion of the Clearwater River drainage of the RMFR? The second question addresses the ecological and practical implications of encroachment into these native plant communities. In particular, this research quantifies how shrub encroachment has changed the vegetational species composition, diversity and aboveground net primary production (ANPP) of the understory. Ultimately, from a land management perspective, it is important to

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understand how rangeland biodiversity, herbaceous production and associated carrying capacity of Upper Foothills meadow vegetation is changing as a result of ongoing shrub encroachment.

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2. Literature Review

2.1 Description of the Rocky Mountain Forest Reserve (RMFR)

The Rocky Mountain Forest Reserve in Alberta covers 23,188 km^2 (Government of Canada 1911a) and is located along the southeast slopes of the Rocky Mountains. It includes the Saskatchewan River basin of the Rocky Mountains, stretching from Waterton Park to the northern extent of the North Saskatchewan River basin. On the west side it adjoins either the BC border, Banff or Jasper National Park, whereas the east boundary is highly variable but generally captures the majority of the foothills (Figure 2-1). The RMFR is a classical example of land managed for multiple use, including oil and gas exploration, timber harvesting, livestock production, wildlife, important watersheds and recreational activities (Willoughby 2001). It's diversity and proximity to the National Parks make it an important area ecologically, consisting of a number of natural subregions including the Montane, Subalpine, Alpine, Upper and Lower Foothills.

The Upper Foothills natural subregion comprises the majority of the northern part of the RMFR and is found elevationally below the Subalpine and above the Lower Foothills (Figure 2-2). Elevation ranges at lower altitudes from 1000 to 1250m and at higher altitudes from 1200 to 1500m. The Upper Foothills subregion has a boreal climate modified by the Rocky Mountains. Average annual precipitation is 538 mm with over half falling in summer (340 mm). Temperatures average 11.5°C in the summer and –6.0°C in the winter, which are milder and less extreme than the Boreal Forest and Lower Foothills.

Vegetatively, this subregion is characterized by a closed canopy of lodgepole pine (*Pinus contorta* Loudon) on slopes and a mosaic of shrubland and grassland on the valley bottoms

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(Rayner 1984, Department of Environmental Protection 1994, Willoughby 2001). Deep snow accumulation and cold air drainage keep deciduous trees (Archibald 1982) and conifers from establishing on the valley floor. Grasslands and shrublands occur mainly as complexes found in larger fluvial systems where better drainage occurs compared to smaller, narrower systems with poor drainage (Archibald 1982). A variety of plant communities have been described in the region (Willoughby 2001), including 18 native grasslands, 12 native shrublands, 6 deciduous forest, 10 coniferous forest, and 11 cutblock community types, along with 11 communities arising from modifications due to grazing. The dominant shrubs are bog birch (*Betula glandulosa* Michx.) and willow (*Salix spp.*) while the dominant grasses include the bunchgrasses, tufted hairgrass [*Deschampsia cespitosa* (L.) Beauv.] and foothills rough fescue [*Festuca campestris* (Rybd.)].

The RMFR is public land managed under a multiple use mandate, and therefore, a wide variety of commercial and recreational activity occur. Currently, there is approximately 75,000 Animal Unit Months (AUMs) of permitted cattle grazing, with the majority of operations consisting of cow/calf pairs. Grazing occurs on allotments, which are contained grazing dispositions bordered by natural barriers such as topographic highs (i.e. ridgetops), rivers or creeks. All allotments are grazed by cattle with the exception of a few horse allotments associated with commercial trail riding operations. The RMFR is also a very important timber resource: two forest management agreement holders manage the timber over approximately half of the RMFR, although numerous timber quota holders exist that generally include the smaller, local logging and sawmill companies. Because of the abundance of wildlife such as elk, deer (white-tailed and mule), moose, cougar, grizzly bear, bighorn sheep, and mountain goat, there are a number of trappers as well as guiding and outfitting operations that hold permits in the western part of the RMFR and provide hunting opportunities.

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The abundance of wildlife combined with the beauty of the area attract many recreationalists seeking either commercial operations offering accommodations and activities such as horseback riding and rafting, or simply an opportunity to camp, generally adjacent to a creek or river. Throughout the RMFR, recreationalists take part in a wide variety of activities from low impact ones such as backpacking, catch and release fishing, and hiking, to greater impact activities such as equestrian and off-highway vehicle use. As a result of high impact activities, damage to riparian areas has occurred creating erosion problems (Figure 2-3), leading to the creation of Forest Land Use Zones (FLUZ) and Wilderness Areas that specify the activity that can occur on a particular area. For example, sensitive areas may be restricted to low impact activities, reducing conflict between recreational users as well as alleviating pressure on grassland meadows where the majority of activities occur.

2.1.1 History of Land Use in the RMFR

The RMFR was established in 1910 under the Dominion of Canada (Department of the Interior 1911a) as European settlement rapidly increased in the foothills of Alberta. Even before the reserve was created a variety of land use activities were occurring there, such as mineral exploration, localized timber harvesting, and in a few mountain valleys, cattle grazing. With the creation of the RMFR, grazing was prohibited until 1913 when it was recognized as a useful tool to reduce forage accumulation and assist in reducing fire hazard (Department of the Interior 1910, 1911b, 1912, 1914a). The area was administered federally until 1930 when the province inherited the responsibility (Government of Canada 1930, Government of Alberta 1931), and as a result, the Alberta Forest Service was created with two major responsibilities, timber management and forest protection (Murphy et al. 2002).

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The RMFR contains the headwaters to the heavily populated regions of Alberta, central Saskatchewan, and west-central Manitoba (Hanson 1973) (Figure 2-4). In 1948, the Eastern Rockies Forest Conservation Board (ERFCB) was appointed to oversee administration of the area for a 25 year period (Canada 1947, Alberta 1947, 1948). Reconnaissance watershed and range surveys initiated by the ERFCB revealed localized overgrazing (Weerstra and AFS 1990), therefore in 1949, the Rangeland Reference Area Program (RRAP) was initiated to assess range condition and monitor the long-term trends on rangelands in the RMFR (Hanson 1973). This program initiated comprehensive range surveys and assessments of the condition of the forage resource.

Forty-five exclosures were established in the RMFR as part of the Rangeland Reference Area Program (Willoughby 2000). These exclosures have one, 33-m transect inside and one outside (Willoughby 2000), along which the canopy cover of all vegetational species and more recently, production data, is collected every three years (ASRD 2002). This has allowed comparisons between grazed and ungrazed plant communities, and facilitated recognition of changes in plant species composition and canopy cover over the years due to livestock grazing. Although many studies, particularly ones assessing woody plant encroachment, report heavy grazing as the cause of shrub invasion (e.g. Dunwiddie 1977, Yool et al. 1997), data collected by Public Lands and Forests Division (PLFD) in the RMFR indicate shrub canopy cover inside the exclosures is generally higher than outside (Willoughby 2000), suggesting grazing has limited or slowed shrub encroachment.

2.1.2 Historical Fire Regime in the RMFR

The basis of current knowledge regarding historical fire regimes resides in dendro-chronology studies, although more specifics regarding the behaviours associated with fire can be

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found in the journals of the first explorers, fur traders, and the North West Mounted Police (NWMP). When colonel Sam Steele led the NWMP into western Canada in 1874 (Steel 1915), he noted, "Indians ... willfully set the prairies on fire [in the autumn] so that the bison would come to their part of the country to get the rich green grass which would follow in the spring". During an interview conducted by Dr. Henry T. Lewis in 1975, Louis Martel, a Beaver Indian in north-western Alberta, conveyed a clear understanding about fire: "Fires had to be controlled. You couldn't just start a fire anywhere, anytime. Fire can do a lot of harm or a lot of good. You have to know how to control it.... It has been a long time since my father and my uncles used to burn each spring. But we were told to stop. The Mounties arrested some people ... the country has changed from what it used to be -- brush and trees where there used to be lots of meadows, and not so many animals as before." (Lewis 1977).

Aboriginals significantly affected landscapes through their use of fire (Lewis 1978, Kay et al. 1994, White 2001), but their use appeared to be tempered by an understanding of the ecosystem, their coexistence with it, and the knowledge to constrain fires to the areas that they wanted to burn (Murphy et al. 2002). For example, they would burn stream and river margins to enable easier travel, promote succulent willow and aspen regrowth for moose and beaver, open up areas in which to establish camps, and to stimulate berry production (Lewis 1978, Kay et al. 1994, Murphy et al. 2002). Another common practice was to burn meadows to encourage the growth of grass and sedges, as well as burn stands of living trees to create future sources of dry wood for fires (Murphy et al. 2002). Much of this burning was done in the spring or fall when fire behavior was easier to predict and controlled by natural features or weather.

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2.1.3 Historical Abundance of Wildlife

Little is known about the influence of historic bison grazing in the Upper Foothills. However, archeological digs along the Red Deer River to the south and the Kootenay Plains to the North have uncovered bison bones (Workshop on Bison Reintroduction 2000). Using historic journals, Kay et al. (1994) compiled accounts of wildlife in order to determine the relative abundance of various fauna in the central Rocky Mountains prior to European settlement. They discovered from their synthesis that in the foothills of Alberta between 1792 and 1863, bison were the most commonly observed ungulate, followed by deer, then elk. During 26 trips recorded by explorers traveling the North Saskatchewan River crossing between 1792 and 1872, the most abundant animal was bighorn sheep, followed by bison, moose, and mountain goat. Bison were observed more than elk, with few deer encountered. Only one account claimed an abundance of wildlife in the foothills region.

2.1.4 Era of Fire Suppression

Following the settlement of western Canada, as immigrants began trying to make a living off the land and industrial activity gained momentum, the government of Alberta deemed it imperative to control the use of fire (Government of Alberta 2001). In 1832, the Council of Assiniboia instituted the first penalties for human-caused fires. By 1877, the council of the Northwest Territories, of which Alberta was a part of, passed an ordinance for the prevention of forest and prairie fires at its first session (Murphy 1985). In 1913, livestock grazing was implemented in the RMFR as a tool for controlling the build up of litter (i.e. fuel) and subsequently reducing the fire hazard (Department of Interior 1910, 1911b, 1912, 1914a). Fire suppression efforts increased in 1953 when a provincial branch of the government was

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mandated specifically with fire control, created to protect timber for an expanding forest industry and oil and gas development in the Rockies (Government of Alberta 2001).

Today, using leading edge technology from computer models to predict fires and highly sophisticated detection systems, fire suppression has become an everyday task for a number of people. Between the Provincial Forest Fire Centre in Edmonton, and 7 Fire Management Districts across the province, the province has access to a staff of 2,270 firefighters and leases airtankers, patrol aircraft, helicopters, heavy ground equipment, and catering camps (Government of Alberta 2001). Not only have their efforts in fire control reached a pinnacle never seen in Alberta, numerous fire awareness campaigns have been initiated including 427-FIRE, which encourages early detection, to urban interface programs like "fire smart" that promote awareness among country residents of the need to protect themselves from forest fire. At the same time, however, government agencies also recognize the importance of fire to maintain wildlife habitat and have undertaken a number of prescribed burns in conjunction with the Alberta Conservation Association to improve habitat. Despite this, many areas of importance are not burned due to the risk posed to infrastructure and forest management areas (FMAs).

2.2 Bog Birch

2.2.1 Description of Bog Birch

Bog birch (*Betula glandulosa* Michx.) is widely distributed throughout North America and is found within 30% of Canada's landscapes (de Groot 1998) (Figure 2-5). It's growth form is an ascending or spreading shrub up to 2 m high found in bogs, seepages, alpine slopes, and open subalpine forest as high altitudes N to Greenland & Baffin Island, W. to Alaska and Asia, S. to Nevada and California, and E. to Newfoundland (Moss 1959). Although bog birch reproduces by

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seed, its' primary reproduction is vegetatively through creeping roots, with its preferred habitat consisting of treeless, nutrient poor, moist and acidic soils (de Groot and Wein 1999). The lifespan of individual bog birch plants is estimated at 100 –150 years (Heinselman 1981). Bog birch has a chemical defense mechanism that evolved in response to browsing, and is responsible for its low palatability and tendency for herbivores to avoid it (de Groot 1998).

2.2.2 Ecology of Bog Birch

Bog birch is highly adapted to fire and the elimination of this species from suitable ecosites is unlikely as this would take a fire return interval of once every three years over a long period of time (de Groot 1998). The occurrence of 3 prescribed fires over a 10-yr period was not able to eliminate bog birch (Bork et al. 1996). Moreover, bog birch has a fire stimulated re-growth mechanism that gives the plant a competitive advantage over surrounding vegetation (de Groot and Wien 1999). Early spring burning will stimulate the largest growth response and biomass production two years after burning (de Groot and Wein 1999). The intensity of fire also affects bog birch regrowth, with low intensity fires producing the greatest regrowth (De Groot 1998). With severe fires the survival of bog birch decreases but creates conditions favorable for the establishment of new seedlings aided by wide seed dispersal and high seed production rates (de Groot and Wein 1999).

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2.3 Shrub Encroachment in the Foothills of Alberta

2.3.1 Extent of Shrub Encroachment

Historically, fire played a major role in structuring plant communities across North America (Bailey and Wroe 1974, Agee 1993, Agee 1994, Kay et al. 1994). Under fire protection, areas once dominated by grasslands have experienced increases in canopy cover by woody species (Bork et al. 1996 and Willoughby 2001). Eighty-five years ago floodplain meadows associated with major drainages in the RMFR of Alberta were either free of shrubs or if present, occurred at low canopy cover (Johnston and Smoliak 1968). Foothills landscapes have changed dramatically since Pre-European settlement (Adams et al. 1992). Peter Fidler, over-wintered in the foothills southwest of Calgary in 1792 and called the area of present day Highway 22X nearly devoid of woody cover (1792).

Shrub encroachment rates are occurring at approximately 0.75 to 5.5% per year along the southern portion of the east slopes, with higher rates on moister sites, the general succession tending to be grass to willows to aspen to conifer (Johnston and Smoliak 1968, O'Leary et al. 1989, Bailey and Conrad 1978). As a result forage production under closed canopy aspen stands yield about 448 $kg.ha^{-1}$ compared to adjacent foothills rough fescue grasslands yielding 1344 $kg.ha^{-1}$ (Johnston and Smoliak 1968). Brush expansion in Alberta's Central Parkland during a 59 yr period (1907 - 1966), increased 4.8 to 8.0%, resulting in a herbage production loss of 80 to 90% under aspen and willow canopies compared to adjacent plains rough fescue [*Festuca hallii* (Vasey) Piper] grassland (Bailey and Wroe 1974). Fescue grassland conversion to brush may be correlated with periods of drought with the highest rates of conversion taking place 1 to 2 yrs. after high mean temperatures and low precipitation (Bailey and Wroe 1974). Similarly, in the mixed grass prairie of Wyoming sagebrush-dominated communities are more likely to occur than grassland as summer precipitation decreases below 282 mm (Driese et al. 1997). Conversely, in

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the Cariboo/Chilcotin forest region of the British Columbia interior, open grassland has the highest probability of occupying drier and warmer south and west aspects, as well steeper slopes limit the amount of tree encroachment (Bai et al. 2004).

Repeat photography (1906 - 1986) of the Ya Ha Tinda, Kootenay Plains and Banff areas were used by Kay et al. (1994) to evaluate landscape changes after a century of National Park management. Although these areas are found in the Montane subregion, they appear to show trends similar to what has occurred in the adjacent Upper Foothills; a dramatic invasion of bog birch and willow onto grassland. Along with shrubs, conifer trees have also increased throughout the Rocky Mountains in southwestern Alberta. Rhemtulla et al. (2002) examined the changes in vegetation of Jasper Park over a 62 yr time period (1915-1977) using repeat photography and air photos and discovered a decrease in grassland, shrubland, juvenile forest and open forest and an increase in closed canopy coniferous forest. In the 1880s, smoke from widespread fires in western Canada darkened skies over London, England (Murphy et al. 2002). This example may have typified the devastating fire cycles that occurred at least once a century during the preceding millennia (Murphy et al. 2002). When forests surrounding the town of Hinton were first inventoried and aged in the 1950s, about one-third of the timber within them was found to have regenerated after the fires of the 1880s and 1890s (Murphy et al. 2002). In contrast, one-third of the timber originated after more recent fires, and one-third originated prior to the 1880s.

2.3.2 Impacts of Shrub Encroachment

For most of the 20th century, the central Rocky Mountain ecosystem has been under a fire exclusion policy, which has created an unnatural mean fire return interval and fire cycle (Rogeau 1999). The historic fire regime of much of the foothills and mountain regions has changed from high frequency, low intensity ground fires, which maintained a heterogeneous landscape

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dominated by grassland, shrubs and aspen, to one of high intensity crown fires due to the increase in conifer as a result of ongoing fire suppression (Kay et al. 1994).

Shrub canopy cover alters the microclimate in the understory, and reduces light, nutrient and water availability through competition, thereby affecting the ability of the understory to maintain growth, vigor, and ultimately, production (Hobbs and Mooney 1986). In the Upper Foothills subregion, shrubs shade the grassland and eventually, understory species composition shifts from tufted hairgrass/rough fescue to slender wheatgrass and sedges (Willoughby, 2001). Additionally, we can expect to see less forage available for wildlife and to accommodate domestic livestock grazing (de Groot 1998), with associated decreases in the value of habitat for wildlife (Adams et al. 1992). Plant communities that are less diverse are generally less resilient or likely to recover from disturbances such as fire and grazing (Tilman and Downing 1994).

2.2.3 Shrub Control Projects

Very few cost benefit analyses have been conducted on shrub control projects in Alberta. However, on two ranches with grasslands encroached by willow in the foothills fescue subregion, Adams et al. (1992), compared the costs per Animal Unit Month (AUM) of burning and grazing treatments to conventional shrub control methods (clearing and seeding). Costs associated with burning and grazing treatments at the two ranches were \$4.04/AUM (1684 AUMs) and \$11.61/AUM (762 AUMs) compared to \$29.40/AUM for conventional mechanical methods including clearing, piling, breaking, seedbed preparation, seeding, repiling brush and regrowth control (based on local rates in effect from 1985-1986). The major costs associated with the burning and grazing treatment was dozing the firebreaks and preparing the sites for burning.

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2.4 Detecting Landscape Changes in Vegetation

2.4.1 Defining Change Detection

Change detection is used to determine changes in land area cover between two imaging dates of a multitemporal data set (Lillesand and Kiefer 1994). It is a widely used process that has applications in quantifying activities such as urban-fringe development, deforestation, and desertification, as well as processes such as snow loss, flooding, and shifts in vegetation. Change detection can be used to quantify changes in vegetation over the landscape and can be applied to remotely-sensed data obtained from aerial photographs and satellite imagery. Although the data obtained from satellite imagery may be better for change detection depending on the object being studied, when a high resolution historical perspective is required, aerial photography may be the most appropriate source of data (Brown and Arbogast 1999). The basic premise for change detection of remotely-sensed data is that alterations in land cover must result in radiance value changes that are detectable using remote sensors (Singh 1989). This implies that changes in radiance due to land cover change must be large with respect to external factors influencing radiance, including atmospheric affects (for satellite imagery and aerial photographs) and differences in sun angle, soil moisture, or vegetation phenology throughout the season (Jenson 1983).

External factors must be taken into account in order to produce a quality map that accurately represents changes in land cover. An important step to minimize the amount of error associated with change detection procedures is to accurately register images from separate sampling dates to one another (Singh 1989). Accurate registration is crucial to ensure reliable change detection (Lillesand and Kiefer 1994), and several conditions should be followed to minimize the error associated with registration, including:

1) Acquire data using the same or similar sensors.

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2) Data should be recorded using the same spatial resolution and time of the day.

3) Use anniversary (similar) dates to minimize seasonal differences and sun angle.

Geo-registration to an accuracy of a quarter to a half of a pixel is often required for successful change detection, and the mapping error associated with mis-registration is large if registration is more than 1 pixel off (Bernstein et al. 1983, Jensen 1986 and Lillesand and Kiefer 1994). However, when co-registering aerial photographs, a Root Mean Square Error (RMSE) of greater than 1 pixel is not uncommon, which must be taken into consideration when interpreting the results of change detection (Brown and Arbogust 1999).

2.4.2 Change Detection Techniques

Many different techniques can be used to discriminate landscape-based changes between two dates of imaging, the most common include post-classification comparison, classification of multitemporal data sets, temporal image differencing, temporal image ratioing, and change vector analysis (Lillesand and Kiefer, 1994). The post-classification comparison method employs an algorithm to determine the number of pixels with a change in classification between two dates of imaging, that are independently registered and classified. The accuracy of this method depends on each of the independent classifications, and thus, the error associated with this procedure is compounded with each independent classification performed.

Classification of multitemporal data sets uses spectral pattern recognition to detect changes. This technique is performed on a combined data set for the two dates of interest and entails a supervised or unsupervised classification to categorize land cover classes. Temporal image differencing adjusts (i.e. subtracts) the digital spectral values of one date from those of another. Areas of change therefore result in either larger negative and positive values while areas of no

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change should very little difference (a value approaching zero). The image differencing technique is the most widely used technique for change detection and has been used in a variety of geographic environments (Singh 1989). A critical element is deciding where to place the threshold boundary between change and no change pixels using a histogram (Singh 1989). Some of the general difficulties associated with image differencing include a sensitivity to mis-registration error, the existence of mixed pixels and it does not consider ending locations of a pixel in the feature space (Riordan 1980). For example, a shrub pixel may have a digital value of 156 on one date and 120 on the second date, despite the substantial change the pixel still represents a shrub pixel.

Change vector analysis is a conceptual extension of image differencing except the process is different. The decision as to whether a 'change' has occurred is determined if the magnitude of the computed spectral change vector exceeds a specified threshold (Singh 1989). This method is computationally demanding because the data need to be geometrically corrected and merged, next transformation coefficients have to be developed, and lastly spectral/spatial clustering is done (Singh 1989). Temporal image ratioing rapidly computes the ratio of the data from two dates of imaging band by band on a pixel by pixel basis. In general, areas of significant change will have higher or lower ratio values, while areas of no change will have a ratio close to 1. The advantage of ratioing is that it tends to normalize the data for changes in external factors influencing pixel values such as shadows and sun angle.

Change detection is one of the major applications of digital satellite data (Singh 1989). He evaluated several techniques including image differencing, image regression, image ratioing, vegetation index differencing, principal component analysis, post-classification comparison, direct multidate classification, background subtraction, and change vector analysis, and concluded that these techniques, even when used in the same environment, yielded different

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results (i.e., different interpretations of land cover change). To compare the performance of various change detection techniques in different environments they must be evaluated quantitatively, although there is a lack of tried and tested procedures in specific environments (Singh 1989). There is also intrinsic difficulty and challenge associated with assessing the accuracy of change detection, and many studies do not present quantitative results (Congalton and Green 1999). Singh (1989) also concluded that digital change detection techniques need to be developed that require less precise registration or allow one to bypass registration altogether because accurate registration of images remains difficult. A lack of accurate ground control points decreases the accuracy of registration.

2.4.3 Examples of Applying Change Detection Methods

Mast et al. (1997) showed changes in vegetation cover and settlement within the ponderosa pine-grassland ecotone along the Colorado Front Range from 1938 to 1990. They combined aerial photographs, topographic maps, and digital elevation models to create a temporal data set on which to base change detection. A non-metric camera with only black/white panchromatic film was used in aerial photography before 1950. This means that the photography from before the 1950's had limited options for digital image processing and associated applications (Mast et al. 1997). However, this study did not discuss what the limited applications of this type of film and photography are. Mast et al. (1997) concluded a 2.5 m spatial sampling resolution was adequate for detecting changes in tree cover, and that this was commensurate with the information content and rectification accuracy of the aerial photography.

Mast et al. (1997) then determined the range of brightness values representing tree cover in the image processing software 'ERDAS Imagine'. A binary 'tree cover' versus 'no tree cover' mask was created from each digital photograph using density slicing (i.e., dividing the data into

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categories based on photographic tone). A Boolean operator, with conditional 'if/then' statements was used to separate the pixels that contained trees from those that did not. The binary map provided a visual interpretation of tree cover change. Areas of change in tree cover, locations, and total hectares of tree invasion were identified by image processing of the digitized air photos (taken in 1937, 1971, and 1988) and GIS analysis of topographic maps. The results demonstrated a 14% increase (2102 ha) in woodland where grassland formerly existed.

The most common method for detecting land cover changes since 1972 (the launch of the first Landsat, has been visual interpretation from multi-temporal MSS images (Yool 1997). Yool et al. (1997) evaluated vegetation changes in the Jornada del Muerto Basin of southern New Mexico, using Landsat MSS imagery from 1983 and 1992, coupled with a GIS. The premise for this research project was that drought and heavy grazing had changed the area from grassland towards shrubs. Data were evaluated for changes in vegetation using three different change detection techniques: image differencing from Landsat MSS red band digital counts, Euclidean distances to produce an image depicting change in red and near-infrared brightness values, and Standardized Principle Components Analysis. Yool et al. (1997) concluded that all three methods produced similar image outputs, and all successfully identified the spatial patterns of extreme rangeland vegetation change and associated environmental problem, although they also favoured the image differencing method because it was computationally the simplest.

In the grassland interior of British Columbia, in the Cariboo/Chilcotin Forest region, vegetation maps of the 1960's and 1990's were generated using aerial photographs and overlaid with GIS layers including aspect, slope, and elevation (Bai et al. 2004) to investigate the changes and the factors affecting the transition between grassland and forest. The maps were classified as open grassland, treed grassland, open forest, and closed forest, based on percent cover of conifers, ranging fro, 0-5, 5-15, 15-35, and >35%. Similar to the map created for the GAP Analysis

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program in Wyoming, by Driese et al. (1997), Bai et al. (2004) used probability indexes to determine the effects of slope, elevation, and aspect on the above vegetation classifications dynamics and forest expansion. Open grassland, regardless of aspect and slope, decreased going from 18, 245 ha in 1960 to 12, 646 ha in 1990, conversely treed grassland, open forest, and closed forest all increased in area.

Aspen encroachment in Central Alberta over 50 years has been quantified by analyzing the conversion of aerial photographs to binary images and included investigating precipitation and topographic variables as controlling factors (Fent and Richard 1999). Aerial photographs from 1950 and 1998 were scanned and georeferenced in ARCINFO and a textural classification using PCI software was employed to differentiate brush from grassland and supplemented with a grey-level thresholding technique in combination with digital gamma enhancements using Corel Photopaint. The final product was a binary image with white pixels representing brush and black pixels grass. Areas were derived by converting the image to a GRID. Brush, specifically aspen clones, increased in area 3-9 times, with higher expansion rates highly correlated with mean precipitation of 450mm or greater.

Change detection has important applications in mapping changes in vegetation cover over the landscape. The variety of remotely-sensed data (e.g., aerial photographs, topographic maps, satellite imagery, Digital Elevation Models), and the many different change detection techniques available provide resource managers with the tools to quantify changes in cover over many years. On the other hand, there are a number of factors that affect the accuracy of change detection and these must be minimized by the user to produce reliable results (Yool et al. 1997).

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Figure 2-1. Map showing the location of the Rocky Mountain Forest Reserve, Alberta.

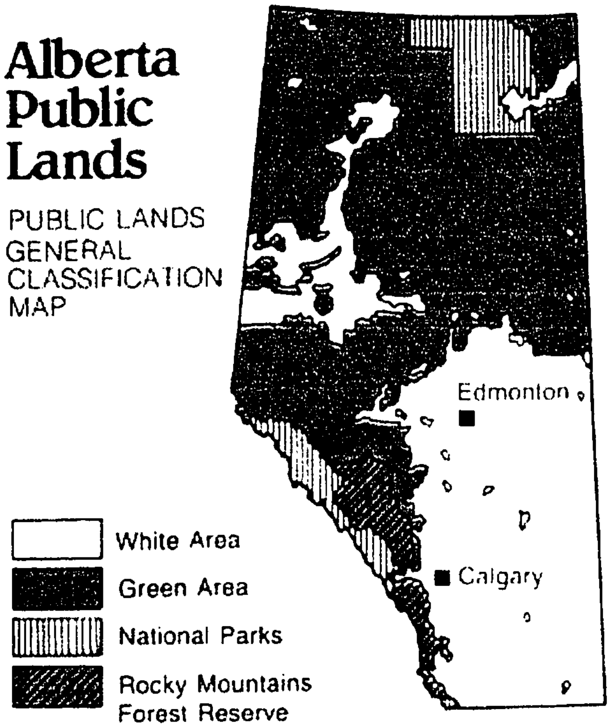

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Figure 2-2. Natural subregions map of Alberta. Note location of Upper Foothills Subregion in dark grey.

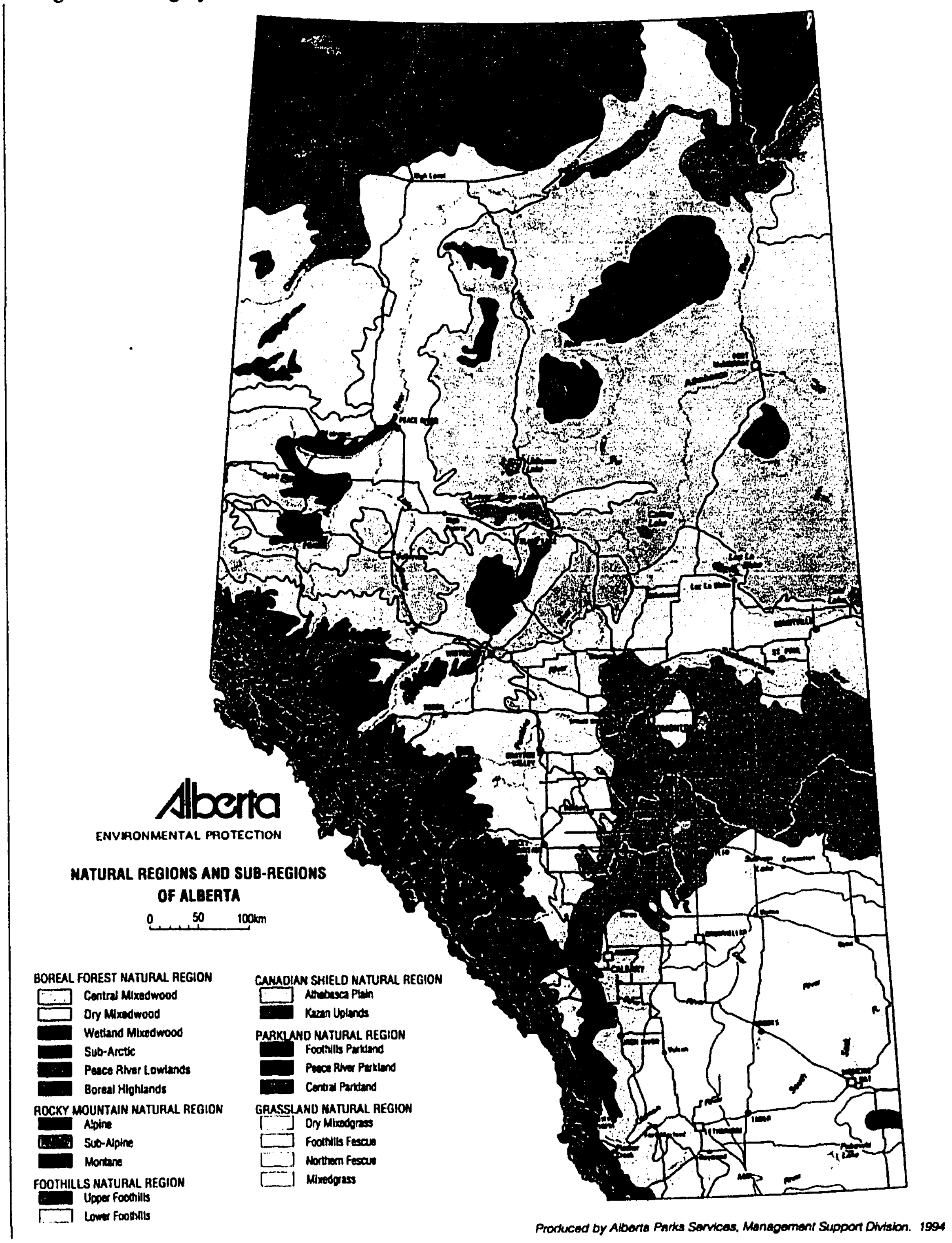

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Figure 2-3. Recreationalists camped along the Castle River (photo by Mike Alexander)

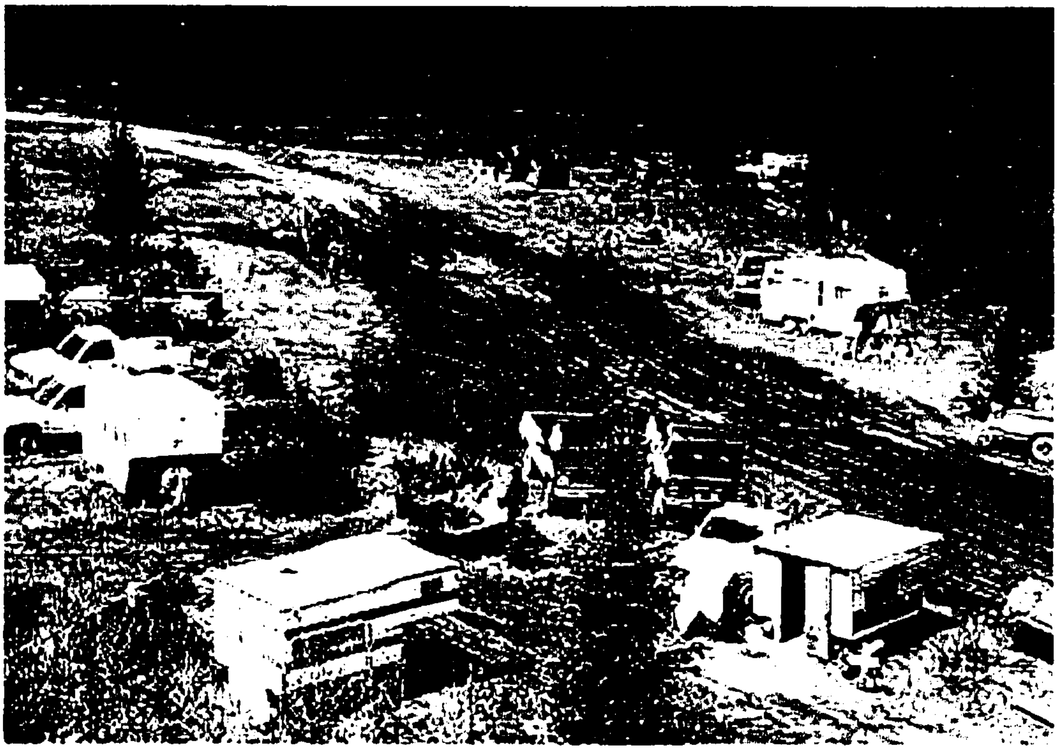

Figure 2-4. Saskatchewan River Basin

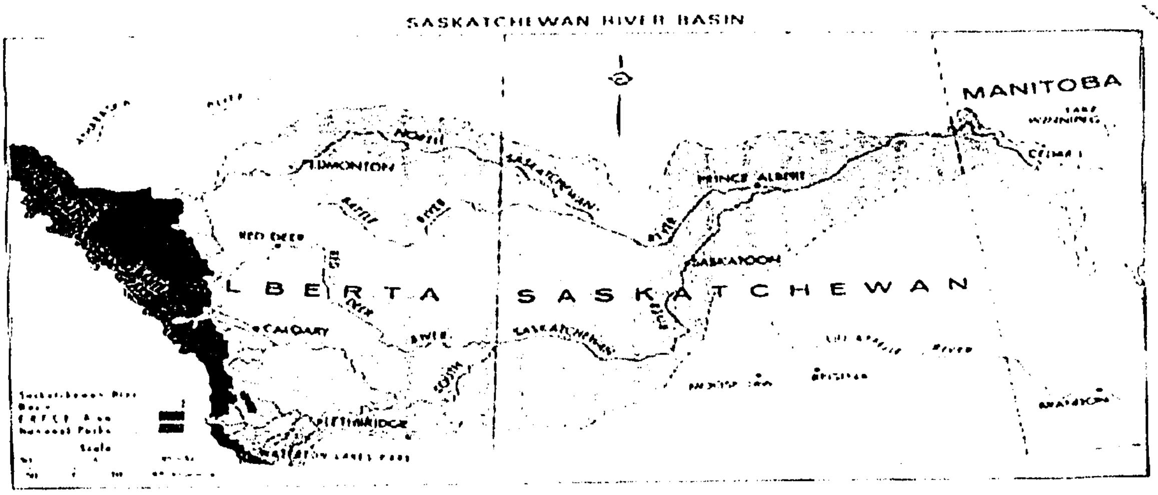

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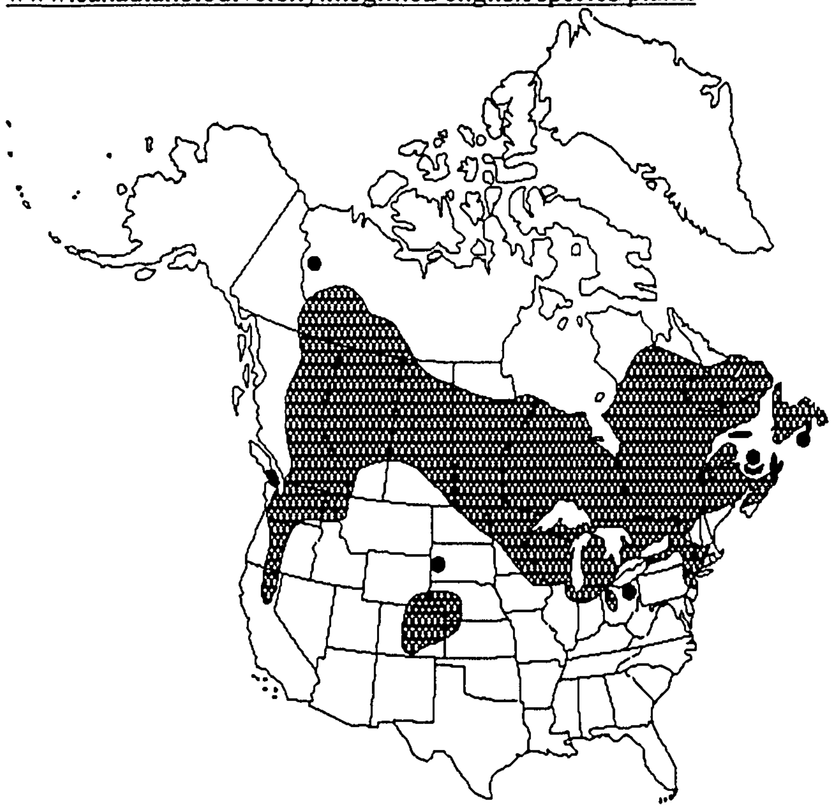
Figure 2-5. Distribution of bog birch in North America.
www.canadianbiodiversity.mcgill.ca/english/species/plants



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3. Quantifying Shrub Encroachment in the Foothills of SW Alberta

3.1 Introduction

Considerable interest has focused on recent landscape changes within grasslands of North America (England and DeVos 1968). European settlement of North America led to fire suppression (Murphy 1985, Adams et al. 1992) and has in turn, resulted in major landscape-scale changes in natural vegetation (Baker 1992, Agee 1994, Turner and Romme 1994). Many studies have documented shrub and forest expansion into grassland ecosystems (Bailey and Wroe 1974, Bragg and Hulbert 1976, Hobbs and Money 1986, Archer et al. 1988, Adams et al. 1992, Kay et al. 1994, Mast et al.1997, Fent and Richard 1999). Causes of this expansion include fire suppression (Knight et al. 1994, Cook et al. 1994, Kay et al. 1994, Bork et al. 1996), climate change (MacDonald 1989, Sturm et al. 2001), nitrogen deposition (Kochy and Wilson 2001), and cattle grazing (Dunwiddie 1977).

In the Rocky Mountain Forest Reserve (RMFR) of Alberta, floodplain grassland meadows have been recognized as an important habitat type for wildlife, a key source of biodiversity, and areas of primary summer range for livestock (Willoughby 2001). Similar to the western US (Cook et al. 1994), fire suppression in this portion of the Rocky Mountains of Alberta has favoured the establishment of mountain-shrub communities (Bork et al. 1996). Native Americans historically burned valley bottoms that were favourable areas to establish camps and as a method of habitat improvement to attract game (Lewis 1978, Kay et al. 1994). Frequent, low intensity burns maintained valley bottoms as herbaceous meadows (Willoughby 2001). Long-term rangeland references areas monitored in the region indicate shrub abundance has increased 15%

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regardless of the presence of livestock grazing (Willoughby 2000). Data collected by Public Lands and Forests Division (PLFD) in the RMFR indicate shrub canopy cover inside the exclosures is generally greater than outside (Willoughby 2000), suggesting grazing has limited or slowed shrub encroachment.

The maintenance of remaining native grasslands is not only important from the perspective of conserving biodiversity (Watkinson and Ormerod 2001) but are also an important source of forage for both cattle in the summer and elk in the winter (Adams et al. 1992, Bork et al. 1996, Willoughby 2001). Moreover, the loss of grassland has led to increased conflict regarding forage allocation within these areas. For example, declining grasslands are thought to result in reductions of total available forage, in turn necessitating either a reduction in livestock grazing, or the increased harvest of wildlife such as elk. As a result, land managers are concerned with both maintaining native grasslands as well as restoring those areas that have been lost due to shrub encroachment. Prescribed burning has been used sporadically to control shrubs and restore grassland (Bork et al. 1996) but remains difficult to undertake due to a number of reasons, including cost.

Effective restoration of grassland within floodplain meadows depends on knowledge about the historical abundance of this habitat type and the changes that have occurred since fire suppression began. Quantitative information on the landscape area affected by shrub encroachment can then be used by managers to modify existing management plans where land use conflicts (i.e. over-allocation of forage) may exist. Furthermore, this information can establish the urgency of grassland restoration in various watersheds and identify target abundances of habitats such as grassland and shrubland, which can then be met through specific actions such as prescribed burning or other restoration procedures.

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The detection of shrub encroachment and quantification of landscape changes from grassland to shrubland over time is crucial for ensuring that long-term land management objectives can be met. This process can be effectively accomplished using remotely-sensed data and GIS applications (Tueller 1989, Knight et al. 1994, Mast et al. 1997, Fent and Richard 1999, Eckhardt et al. 2000, Bai et al. 2001). The widespread availability of repeat aerial photography for much of the RMFR, dating back to as early as the 1950's, provides the opportunity to assess landscape changes in vegetation over time.

This study quantified the spatial extent of grassland loss (and therefore, shrub encroachment) over a 40 year period within grassland meadows located along portions of the Clearwater River in the Upper Foothills of the RMFR. This was accomplished using aerial photography from 1958, 1974, and 1998, as well as field data collected from 33 locations in 2001 and 2002.

3.2 Methods

3.2.1 Study Area

This research was conducted in a portion of the Clearwater Grazing Allotment approximately 80 km southwest of Rocky Mountain House, Alberta in the RMFR along the eastern slopes of the Rocky Mountains (Figure 3-1). The total study area is approximately 2,400 ha and focuses on the valley bottoms of Elk Creek and Radiant Creek, close to where they drain into the Clearwater River (Plate 3-1). Outwash plains and eskers were deposited here during glacial melt and glaciofluvial terraces were later deposited along the Clearwater River (Rayner 1984). Slopes throughout designated meadows in the study area vary from 1 to 9% (Rayner 1984) and the elevation ranges between 1,425 and 1,530 m. The study area receives an average

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of 538 mm of precipitation annually, of which over half (340 mm) falls in the summer. Temperatures average 11.5°C in the summer and –6.0°C in the winter, which are milder than the Boreal Forest and Lower Foothill subregions. The average number of frost free days is 75, with July generally frost free, and June and August receiving up to three days of freezing temperatures (Hanson 1973).

The study area is located within the Upper Foothills Natural Subregion (Strong and Leggat 1992). The widespread distribution and abundance of lodgepole pine (*Pinus contorta* Loudon) is the result of periodic fire events, which have prevented a successional shift to spruce dominated landscapes (Cormack 1953, Pettapiece 1971, Rayner 1984). Valley bottoms consist of a mosaic of shrubland and grassland (referred to in this study as meadows). Dominant grasses are foothills rough fescue (*Festuca campestris* Rydb.) and tufted hairgrass [*Deschampsia cespitosa* (L.) Beauv.], with the dominant shrubs bog birch (*Betula glandulosa* Michx.) and Barclay's willow (*Salix barclayi* Anderss). A number of different plant communities are found here ranging from wet sedge meadows at lower elevations to the bearberry/rough fescue type at the north end of the study area. The Elk Creek grazing exclosure is indicative of the latter plant community type. The Alberta Forest Service (AFS) constructed this exclosure in 1965 to monitor the long-term trend in range health under domestic livestock grazing (Weerstra and AFS 1990) (Appendix A).

Soils vary throughout the study area. Rayner (1984) found medium to course textured alluvial deposits consisting of Orthic Regosols or Rego Gleysols, with small pockets of medium to fine textured alluvium and mesic Organics. The latter soils are moderately well-drained to very poorly drained and include Mesisols. These sites support open deciduous shrubland, grasslands and fens in a wide flat valley of second or third order streams and can be well to poorly drained (Rayner 1984). Soils at the Elk Creek Exclosure have been classified as Orthic

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Eutric Brunisols with a clay loam texture overlying a lacustrine parent material (Weerstra and AFS 1990).

The study area was first grazed by cattle in the mid 1930's, and for the first 23 years use was not regulated or monitored. In 1953, the area was established as part of the Clearwater Grazing Allotment. The first range management plan was developed for the area in 1958 and the stocking rate set at 3,080 Animal Unit Months (AUMs). This number represented 50% of the maximum carrying capacity to accommodate a large elk population that used the valley bottoms during winter and spring calving (Weerstra and AFS 1990). In 1962 the allotment was divided into two areas in an effort to eliminate conflict between two permittees's. Ten years later, however, the range management plan was updated and the 2 allotments amalgamated. A second management plan was developed in 1972 and the new stocking rate set at 2,250 AUMs. In 1976, a rotational grazing system was implemented that required cattle be moved from one part of the allotment (distribution unit) to another as one herd. Within this rotational pattern, the study area was generally utilized throughout the summer and early fall, with occasional use to the end of October (Weerstra and AFS 1990). Due to poor management (i.e., inadequate cattle distribution and salting) the allowable use was reduced to 1,055 AUMs, but with the addition of a new permittee in 1980, stocking rates were allowed to increase on an annual basis, though never above the 1977 stocking rate of 2,803 AUMs. As part of the provincial policy for resource management in the eastern slopes, stocking rates were to be maintained at 1977 stocking levels (Alberta 1984). A third management plan in 1989 set the stocking rate for the entire allotment at 2,803 AUM. The specific carrying capacity for the Elk Creek and Idlewilde Creek distribution units was 690 and 412 AUMs, respectively (AFS 1989).

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3.2.2 Scientific Approach

A GIS framework was used to conduct all spatial and temporal analyses. Steps taken to complete the GIS portion of the study can be seen in Figure 3-2. Digital orthorectified photography of the study area taken in 1998 was initially converted to a grid, with a natural breaks classification used to find patterns inherent to the data. This classification was refined using a calibration - validation procedure employing field transects to more accurately determine the range of pixel values associated with both grassland and shrubland. This exercise also provided a measure of accuracy for detecting pixel values associated with the primary habitats of interest: grassland and shrubland. Once completed and the accuracy level determined, this same procedure for establishing pixel values associated with grassland habitat was applied to the 1974 and 1958 digital imagery. Validation of those data sets was not possible due to the lack of comprehensive field data for those dates other than photographs and transect data collected from the elk creek grazing exclosure (Appendix A). Finally, the calculated areas of grassland in 1958 and 1974 were compared against the area of grassland remaining in 1998.

3.2.2.1 Image Registration

Historical aerial photos were obtained of the study area west of Rocky Mountain House and selected based on their quality and scale. Photos taken prior to 1958 were not used because of poor quality and limited corroborating data, even anecdotal information, on the vegetation of the surrounding area before this date, either from Alberta Forest Service (AFS) staff or ranchers who grazed livestock in the area. The 1958, 1974 and 1998 photography was taken 15 August at a scale of 1:15,840, 1 September at a scale of 1:21,120, and 15 September, at a scale of 1:40,000 respectively. The 1998 aerial photography was orthorectified using provincial base map features (i.e., roads, rivers, etc.) and already available in digital format. The 1958 aerial photographs were

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scanned at different resolutions, which resulted in different pixel sizes (40 x 40 cm) and subsequently led to a different ground resolution (0.16 m^2) compared to the 1974 and 1998 photos, which had a ground resolution of 0.56 m^2. The two earlier air photos of the study area were co-registered to the 1998 orthophotography in Arcinfo using a minimum of 30 ground control points concentrated in the valley bottoms. Anthropogenic features that existed in 1958 and 1974 were used as ground control points, which included roads crossing either creeks, other roads or trails.

3.2.2.2 Setting the Spatial Extent

To reduce the amount of time spent processing images, digital imagery of the air photos was clipped in Arcview using the Grid Pig and Grid Analyst extensions. Digital images were initially clipped to an extent that contained the majority of the study area's meadows; geographic coordinates were retained in the clipping process. This imagery was then clipped further into five smaller sub-areas to better facilitate image processing, including the areas of Elk Creek, South Elk Creek, Radiant Creek, West Radiant, and Between Elk and Radiant Creeks. The same shape file was used to clip images of the study area from 1958, 1974, and 1998, thereby ensuring the analysis took place over the same spatial extent for all three years of imagery.

The geographic coordinates of 33, 30-m linear transects were overlayed on the digital images (Plate 3-1). These transects were used to collect the associated field vegetation data used in this study, specifically the density and canopy cover of shrubs and bunchgrasses. Specific XY coordinates (originally obtained as decimal degrees, minutes, seconds) and elevation (m) were obtained for each transect using a GPS (Global Positioning System) receiver. Meadows of interest were enlarged in Arcview to a scale where both shrubland and grassland could be examined (Plate 3-2). Since the georegistration varied between the 1974 and 1958 photography,

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transects overlayed on the digital imagery were spatially corrected for each year. This was completed using the measuring tool for distance and northing and easting directions in Arcview, compared to stationary objects such as trails, roads, creek bends and trees.

3.2.2.3 Field Sampling

The 33, 30-m transects used in the calibration/validation exercise were established in the summers of 2001 and 2002 (Appendix B). It was assumed that portions of the study area dominated by grassland in 2001 and 2002 were also dominated by grassland in 1998. This is a reasonable assumption because there has been no clearing of shrubs (and no fire) in the area during the interim 3 to 4 year period. Transects were established in pairs throughout the study area, with one in grassland and another in adjacent shrubland on the same ecosite. This was done to minimize site variability (i.e., moisture, nutrient, drainage, and soils) between the grassland and corresponding shrubland transect.

Field measurements recorded for the purpose of this study included the canopy cover and density of bunchgrasses and shrubs, as well as the height of shrubs. Canopy cover was determined using a 30-m long line intercept and density counts were conducted within a belted transect by counting individual shrubs within 1-m on either side of the 30-m transect (2x30-m) for a total area of 60 m^2. Other measurements were taken to provide information for the synecology component of this study (Chapter 4) such as above-ground net primary production, species diversity and richness, the density of bunchgrasses, and the canopy cover of individual forbs, grasses, lichens, mosses and shrubs.

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3.2.2.4 Image Classification

Classification is the process of aggregating data into homogeneous groups (i.e., vegetation classes) that have similar characteristics such as digital values. Prior to classification, all digital images were converted to grids to facilitate processing, after which the initial unsupervised classification took place (Plate 3-3). Although there are six possible classification methods in Arcview, the natural breaks procedure best suited the purpose of this study. This procedure looks for breaks that are inherent to the data and identifies breakpoints between classes using a statistical formula (Jenk's optimization) that minimizes the sum of the variance within each of the classes (Arcview 3.1 2001).

Once the type of classification to be used was determined the number of classification ranges was set to five: these classes were considered to comprise all the important land cover classes within the digital imagery investigated. The five classes included shadows and water as the darkest pixels and corresponding lowest digital value, and a class of roads and trails with the brightest pixels and greatest corresponding digital values. In addition, three vegetation classes were delineated including conifer, shrubland, and grassland.

To better capture the pixel values associated with the two key cover classes of interest, shrubland and grassland, these two classes were reclassified by conducting a calibration - validation exercise, which also assessed the accuracy of the previously mentioned unsupervised classification. This process involved determining the digital values of all pixels in the 1998 imagery along seven randomly selected transects (training sites), including three within primarily grassland habitat and four in areas dominated by shrubland. All transects used in the calibration were considered representative of the plant communities found throughout the study area.

As an initial step, a sensitivity analysis of the digital pixel values was conducted to evaluate potential variation in the sampling 'width' of the field transects. This was done because although individual pixel rows were 0.75 m wide, similar to the width of the actual field transects, the

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ability of the spatial analysis to accurately correlate field transects with the digital data was limited by the error of GPS determination of transect locations, as well as the registration error within the digital data. Three randomly selected transects were chosen and the digital value ranges examined across consecutive increases in pixel sampling width. Sampling width included either one row of pixels (n=41 pixels, 0.75 m x 30.75 m in length), or increments of one additional row up to 6 rows in total. The last row width therefore sampled an area with dimensions 4.5 m by 30.75 m.

The final step in calibration involved determining the number of pixels (out of 41) in each of the seven individual training transects occupied by either shrubland or grassland. Once the digital pixel value ranges associated with actual shrubland and grassland vegetation were clearly delineated within the seven training transects, the remainder of the digital grid for the study area was reclassified using the digital pixel value cutoff between these two classes. Accuracy of the initial calibration was assessed (i.e., validated) by testing the empirical relationship (Goodness of Fit) between the predicted (based on pixel values) and actual field vegetation data (shrub canopy cover) collected in 2002 for the remaining transects (n = 26) not used in the calibration. This procedure assumed that no interim increase in shrubs occurred between 1998 and 2002.

Finally, this procedure for classifying grassland and shrubland was applied to the 1958 and 1974 digital imagery. Although impossible to validate these classifications due to the absence of widespread field data, information about the Elk Creek exclosure (established in 1965) was used to supplement the classification of the two earlier sets of imagery. It was assumed that areas free of shrubland in 2002 around the exclosure were also free of shrubland in 1958 and 1974. This conclusion was supported by historical data from the Elk Creek exclosure monitoring site, which provided data on the basal frequency of shrubs throughout the period spanning the study (Weerstra and AFS 1990). As a result, the area immediately around the exclosure was used to determine pixel values associated with grassland for the earlier sampling times, and coupled with

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careful visual interpretation of the imagery, the classification was performed using the resulting pixel value cutoff.

3.2.2.5 Assessing Grassland Changes

To quantify changes in grassland between 1958 and 1974, and again from 1974 to 1998, the calculated area of grassland was contrasted between time periods for each sub-area of the study area. Since the aerial photography was scanned at different resolutions and therefore had different pixel sizes, total areas of grassland within each sub-area were calculated by multiplying pixel size by the number of pixels classified as grassland. Statistics obtained from the classification in each date of sampling provided the number of pixels as well as area (ha) associated with grassland. Given that the procedure used concentrated on classifying grassland and all of the study area was considered either grassland or shrubland (other types had been removed in the initial classification), only the area of grassland was formally examined and presented here. Moreover, changes in grassland were calculated only through simple post-classification comparison of the multi-temporal data sets from 1958, 1974 and 1998. This approach was used rather than true change detection because of the inability to directly spatially correlate individual pixels as small as 0.16 m^2 in size.

The calculated change in area associated with grassland was also corrected using the empirical relationship between predicted and actual field data from 2002 (Figure 3-3). For all transects, actual shrub canopy cover was over-estimated by an average of 15.4% (Table 3-2). Because grassland and shrubland classes had to total 100% on these transects, adjusted grassland areas could be derived by multiplying each area of predicted grassland by 1.15. Adjusted numbers were used to correct the total area of grassland for each of the study sub-regions and the combined study area, as well as the associated carrying capacity estimates derived below.

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The implications of a change in grassland from 1958 to 1998 is a potential loss in animal carrying capacity. This decline was quantified by determining grazing capacity based on forage production data collected in 2001 and 2002, standardized to the number of animal units. Grazing capacity was then multiplied by the area of calculated grassland for each of the sub-regions in 1958, 1974 and 1998. Although forage production values collected in these years were assumed to be representative of the 40 year time period with no degradation due to grazing or other factors, precipitation was below normal in 2001 (335.3 mm, or 56.1 mm below normal) and 2002 (414.6 mm, or 101.7 mm below normal). Thus, forage production values would likely be conservative compared to the 1958 - 1998 average production values. In any case, historical records of stocking levels for the area obtained from Public Lands and Forests Division indicate stocking levels have been relatively stable over time (Appendix C), enabling an objective assessment of current stocking levels in relation to forage availability for both cattle and wildlife.

3.3 Results

3.3.1 Grassland Abundance

Vegetation field data, collected in 2002, showed that shrubby cinquefoil was associated with a number of the predominant grassland transects. As a result, a threshold shrub cover level needed to be determined to discern "shrubland" from "grassland" habitat and distinguish the pixel values associated with each. Sixteen transects had under 12% shrub (willow and bog birch) canopy cover and 17 transects had between 34 and 92%, indicating a relatively distinct split between grassland (<12% shrub cover) and shrubland (>34% shrub cover) vegetation types. Therefore, transects with less than 12% shrub canopy cover were considered 'grassland' while transects with greater than 34% shrub canopy cover were considered 'shrubland'.

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Increasing the width of sampling from one to six rows led to a 15% increase in the range of pixel values for the grassland and shrubland land cover types (Table 3-1). It is not surprising that the range of pixel values associated with each transect increased when more rows were sampled because of the inclusion of more pixels, and therefore digital values, in a larger area. It was thus evident that enlarging the row width may have increased the likelihood of moving into another, adjacent vegetation type in this heterogeneous environment. However, mean pixel values for each transect generally remained stable regardless of the number of pixel rows examined (Table 3-1). Mean pixel values for transect RP30 and RP31 changed by less than one digital value for all additional rows of pixels used. For transect RP04, the mean pixel value changed by 3.5 digital values, probably due to the inclusion of portions of open grassland adjacent to the largely shrub covered transect. Based on these results, a narrow row width of only 1 pixel was used in all subsequent calibration and validation procedures.

Based on the 123 pixels (3 transects of 41 pixels each) used in the calibration procedure, the pixel values associated with the grassland transects ranged from 166 to 194 and the range of the four shrubland training transects was 123 to 165, in the 1998 digital imagery. The grassland training sites used for the 1974 calibration ranged from 156 to 210. The same number of training sites was used for the 1958 imagery and included the Elk Creek exclosure as part of the grassland training sites. For the 1958 imagery, the pixel values from the same training sites for grasslands ranged from 121 to 165. However, in 1974, the transect at the Elk Creek exclosure was not included because the pixel values associated with this area were generally low. This area of the photo was very dark and appeared to have been caused by under-exposure during collection of the film and not by any feature of the vegetation. Therefore, the pixel values associated with these transects were omitted.

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Results of the calibration - validation exercise demonstrated the accuracy of determining those pixels associated with grassland and shrubland in the orthophotography of 1998, compared to what was observed from the field data collected in 2002. The shrub canopy cover and density data collected in 2002 for each transect were used to verify the presence or absence of shrubs in the reclassified image (1998) produced from the training sites. The standard error in determining whether pixels were associated with shrubs was 3.5% and the mean difference between actual (field data) and predicted (remotely sensed) shrub cover 15.4% (Figure 3-3). Although the relationship between actual and predicted shrub cover had a relatively high Goodness-of-Fit (r^2=0.89), the error arose from a tendency to over-estimate shrub canopy cover (%) for all transects by 15.4%. Additionally, absolute predicted shrub canopy cover was more accurate when fewer shrubs were on the site. This finding may be attributed to the range of pixel values for shrubland being set too low in the calibration exercise or more likely, from pixel values associated with shadows being incorporated into the shrubland class, resulting in its overestimation.

3.3.2 Temporal Changes in Grassland Area

There was spatial variation among all five sub-regions of the study area (Elk, South Elk, Radiant, West Radiant, and Between Elk and Radiant) in the extent of decrease in grassland throughout the 40-yr time period (Table 3-2; Figure 3-4). Data from Elk Creek and West Radiant demonstrated the greatest overall unadjusted decrease in the area of grassland, but also had the most area of grassland in 1958 at the start of the monitoring period. Relative losses in grassland area among the five sub-regions varied from a low of 53% at Radiant to a high of 67% at South Elk (Table 3-2). When all sub-regions were combined, results showed the loss of grassland

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throughout the 40 yr time period for the whole study area was substantial. The unadjusted amount of the study area covered by grassland in 1958 was estimated at 1,313 ha, which by 1974 had declined to approximately 825 ha. In 1998, grassland had been further reduced to 549 ha (Table 3-2). These changes translate to an overall decrease in grassland from 1958 to 1998 of 764 ha or 58% (Table 3-2). Based on the results of the previous calibration - validation exercise, it is possible the area associated with shrubland was over-estimated by as much as 15.4%. Even after adjusting for this over-estimation, the decrease in grassland throughout the study area during the 40 year period was still 646 ha or 49% of the original area (Table 3-2), or approximately half the grassland area during this period.

Notably, while the increase in area associated with shrubland across the study area was marked, this change may not have captured the entire loss of grassland. The total area of grassland lost may also be attributed in part to conifer encroachment. Although not analyzed in this study, visual comparisons of the 1958 and 1998 photos indicated there appeared to be an increase in conifer trees within these floodplain meadows (Plate 3-2; Plate 3-5). However, the invasion of shrubs onto grassland appears to be the main cause for the latter's decline.

The decrease in animal carrying capacity from 1958 to 1998 demonstrates the implications associated with the loss of grassland (Table 3-3). In 1958, the estimated carrying capacity of grasslands (adjusted area) across the study area was approximately 2,157 Animal Unit Months (AUMs). In 1974, this carrying capacity dropped to 1,354 AUMs. From 1974 to 1998 the grassland carrying capacity decreased by 451 AUMs to a total of 903 AUMs. Over the total 40 yr time period the overall decrease in carrying capacity was 1,254 AUMs. This reduction is equivalent to 314 beef cows weighing 454 kg, grazing for four months, or 297 elk weighing 320 kg grazing for 6 months (i.e., during winter).

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3.4 Discussion

3.4.1. Implications of Grassland Change

By examining differences between air photos from 1958, 1974 and 1998, this study was able to quantify the loss of grassland meadows within this portion of the RMFR. Due to differences in geo-registration, the general trend across years of the multi-temporal data set was emphasized rather than absolute change detection. Even allowing for misclassification error (i.e., error associated with classifying grassland pixels as other vegetation classes) between historical and modern vegetation maps, the magnitude and inter-site consistencies in the change from grassland to shrubland are considerable between 1958 and 1998.

The justification for this study was a concern that shrub expansion may have occurred over the last four decades. Although the change in shrub cover could not be determined it is evident from visual interpretations that the 49% (adjusted) decrease in grassland area between 1958 and 1998 is primarily the result of increasing shrub cover (Table 3-2). Moreover, despite some variation in the actual amount of grassland across the study area, all areas were heavily affected. By combining data from these sub-areas the overall impact of shrub encroachment is appreciated on a broader landscape scale. The associated (adjusted) decline in carrying capacity of 1,482 AUMs across the study area reflects the impact that shrub encroachment has had on forage production.

The study area is situated such that the majority of primary rangeland is in the Elk Creek and Idlewilde distribution units (DUs), which are fields within the Clearwater (cattle) Grazing Allotment. Accurate records of livestock distribution and the grazing season within DUs were not kept prior to 1961, and AUMs could therefore not be calculated (Weerstra and AFS 1990). However, general information on grazing is available (Appendix C).

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In recent management plans developed by Alberta Sustainable Resource Development (ASRD), managers have made a commitment to maintain the 1977 stocking level of 2,803 AUMs for the entire Clearwater Allotment, and the 1,102 AUMs for the specific study area (Elk Creek DU) examined here. According to current estimates, however, the study area carrying capacity is already slightly less at approximately 1,067 AUMs. As a result, not only is the Elk Creek DU currently 33 AUMs below the stocking levels approved by the last management plan, when the 49% decrease in grassland since 1958 is considered, it appears this region may be susceptible to even greater forage losses. From 1958 to 1974 the area associated with grassland decreased by 37%. Over the next 24 years grassland decreased further by 33%. Given that this trend for shrub encroachment may well continue we can expect to see the remaining grassland habitat in the study area become further dominated by shrubland, leading to additional losses in carrying capacity.

Together with increased shrubs, there was a visible increase in conifers within the air photos examined, as they appear to be encroaching into shrubland. In the synecology study (Chapter 4, this volume), small pine and spruce trees were noted in several shrubland transects. Additionally, the disappearance of bunchgrasses and xeric-adapted species such as shrubby cinquefoil coincided with an increase in the density, canopy cover and height of taller shrubs such as willow and bog birch. These observations seem to support the notion that there has been widespread successional advancement across the study area, including bog birch and willow encroaching onto grasslands (Willoughby 2001). As taller shrubs become established, bunchgrasses and shrubby cinquefoil decline. Eventually, conifers are able to encroach into existing shrubland. It is quite likely that more of the remaining meadows will eventually be dominated by near continuous shrubland or even coniferous forest in the future, with meadow vegetation restricted to sites that are too dry or nutrient poor for woody species establishment.

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Worldwide, grasslands are considered one of the most threatened natural regions on earth (Henwood 2004). Although this particular study area is not in the grassland natural region of Alberta, it does have an important grassland component (Willoughby 2001) that contributes significantly to regional biodiversity and habitat management within the province. Loss or degradation of unique native habitat is widely recognized as the greatest threat to biodiversity (unpublished ASRD 2003). Although the area examined here is currently being grazed at conservative stocking levels by livestock, as shrubs encroach and other recreational activities increase, or as the number of elk in the area increase, the grassland component of these valley bottoms will be heavily impacted. Additionally, any management actions that increase forage use have the ability to increase grassland degradation, including increases in cattle stocking or elk. While cattle numbers have remained relatively constant since 1985, elk numbers have recently increased (Jim Allen, Fish & Wildlife pers. comm. 2001), in part due to ungulate relocation projects. The latter activity has the potential to place further pressure on the remaining forage resources and degrade grasslands. Perhaps most important, once woody plants dominate these meadows, the fire regime will change. For example, the low intensity, frequent fires created from abundant fine fuels that would have historically kept these meadows dominated by grasslands, have been replaced by a different (woody) fuel type that burns less often but with greater intensity (Wallace et al. 2003).

Current provincial management agencies are in the process of developing and adopting a strategy for conserving Alberta's biodiversity, which in turn, will set the course for minimizing the loss of biodiversity and associated economic, environmental, social, and political risks. Addressing biodiversity issues requires the effective integration of a wide variety of resource users and reflects a shared vision consistent with maintaining multiple uses on public lands. Land managers can contribute to this effort by working to control further encroachment of shrubs onto grassland, as well as restoring those grasslands already invaded.

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Prescribed burning of valley bottoms in the RMFR is one way to potentially reduce shrubs and restore the grassland component (Bork et al. 1996), thereby promoting biodiversity and restoring previous habitats. In addition to damaging shrubs, burning can promote seed set of key forage species and allows for a flush of early successional forbs (Bork et al. 2002). Despite the known historical importance of fire in the region, burning is not widely implemented at this time, presumably due to the risk associated with fire use. Moreover, repeated fires are necessary on a frequent interval to achieve effective control of the shrubs common in the region, particularly bog birch, which is well-adapted to and highly tolerant of fire (Bork et al. 1996).

Decisions on when and whether to burn are currently based on the potential economic loss of timber allocated to forest companies on the uplands adjacent to floodplain meadows. In contrast, it is difficult to attach a monetary value to grasslands that are contributing to regional biodiversity, wildlife habitat, and livestock grazing. Cost considerations are also a key determinant of the ability of PLFD to undertake controlled burns. Ultimately, despite the fact that fire may be the best and most natural tool for shrub control, the potential cost of dealing with an escaped prescribed burn often precludes the use of fire, particularly at the spatial and temporal scale needed to effectively reverse shrub encroachment throughout the entire RMFR. However, if the strategy of conserving biodiversity in Alberta is adopted more widely in the future and these ecological concerns are recognized as a priority, more resources could be made available and greater efforts directed into determining what the economic value of these floodplain grasslands may be through their support of livestock producers and wildlife populations, leading to their restoration.

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3.4.2 Evaluation of the GIS/Field Validation Approach

The quality of the photos used in this study influenced the identifiable pixel value ranges associated with shrubland and grassland habitats. Furthermore, aerial photography for 1958 and 1998 were of better quality than the 1974 photos. In the original 1974 aerial photography, shrub pixels tended to be similar in appearance to grassland pixels in the upper portion of the Elk Creek sub-area. Despite this, the 1974 imagery still resulted in areas classified as grassland consistent with (i.e. intermediate between) those evident in the 1958 and 1998 data, assuming shrub encroachment occurred at a constant rate throughout this period. Nevertheless, the exact quantitative decrease in grassland associated with the 1974 data relative to 1958 should be used cautiously and is better interpreted as part of an overall trend. Specifically, the exact rate of shrub increase associated with the 3 dates of imagery examined here may be a result of limited photo quality at the earlier dates, as well as actual vegetation changes, including those from grassland to shrubland.

During calibration with the field data, the pixel values associated with shrub were obvious along some transects (e.g., Plate 3-3). Although portions of transects dominated by bunchgrass vegetation were highly visible before and after the reclassification process during calibration, distinguishing whether individual pixel values were indicative of the presence of grass in transects containing between 12 and 60% shrub cover was difficult. Fortunately, all but 5 of the transects consisted of shrub cover over 60%, making the identification of the majority relatively easy. Conversely, intermediate communities containing moderate amounts of shrub cover were the most difficult to assess, but are arguably the most important as they represent those communities actively transitioning from open grasslands to closed shrubland.

When determining the values of pixels on digital images assembled from a mosaic of aerial photographs, there is a risk of having different values for the same habitat feature from different photos. This was evident even with the relatively high quality 1998 imagery. For example,

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where two photos were spliced together two different grassland values were evident on either side of the splice. In this case, the 1998 reclassification of grassland led to values ranging from 166 to 194. Fortunately, because 165/166 was used as the threshold between vegetation types, the inherent differences in grassland pixel values were still captured in the grassland class. However, this may not have been as reliable with the photos from the other dates. While the tones of the 1958 photo was relatively consistent throughout, the 1974 photo was not, as previously discussed.

This investigation utilized a passive comparison of the area of grassland obtained separately from each of the 3 years of photography. This was necessary because of the high spatial resolution of the scanned photographs used here (e.g., around 0.5 m^2), and the error (RMSE) associated with co-registration of the aerial photos. The RMSE of the 1958 and 1974 imagery ranged from 11.0 to 16.0 m relative to the 1998 date of sampling (Appendix D). True change detection can only be done if the images can be accurately registered to one another (Singh 1989), allowing individual changes in pixel value (and interpreted vegetation type) to be isolated. Because the study area had very few anthropogenic features in 1974 and even fewer in 1958, accurately registering the air photos proved difficult. One of the only reliable features used to co-register the photos was where bridges intersected the forestry trunk road. Difficulty in geo-registering features precluded a true change detection analysis, which would have had the advantage of mapping discrete localized areas of shrub expansion across the study area. Instead, the calculated areas of grassland found here can only be used to interpret vegetation dynamics at the scale of the sub-areas and the total study area. This method was considered relatively robust because each sub-area was surrounded by a matrix of identifiable coniferous forest. Thus, registration error would not affect the ability to assess overall changes in the area of isolated grassland within each sub-area.

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3.5 Conclusion and Management Recommendations

Despite the problems with determining pixel values associated with shrubland and grassland, the results of this study indicate significant areas of open grassland have been lost in this portion of the Clearwater Grazing Allotment within the RMFR between 1958 and 1998. Specifically, this research found a marked 49% (adjusted) decrease in total grassland area across all five sub-areas over the 40 years. This change in vegetation has also led to the estimated loss of 1,482 AUM of grazing capacity for cattle and elk, potentially intensifying competition between these species and threatening the long-term sustainability of native grasslands if ungulate numbers are not adjusted. Moreover, this trend towards a shrub dominated valley bottom is likely to continue with modern fire suppression efforts (de Groot 1998). Based on this study, projections could also be made to determine the future status of grasslands in this portion of the Clearwater River drainage.

The results of this study provide land managers with important information on habitat changes and associated ungulate foraging opportunities, as well as the need for implementing various land management strategies consistent with conserving grassland habitats. For example, the loss of grasslands is a major concern based on their importance as unique vegetation types contributing to regional diversity, as well as their importance for wildlife habitat and commercial grazing. In order to maintain existing grasslands as well as reverse the observed reduction in grassland with shrub encroachment, land managers will need to increase their use of prescribed burning or other shrub control practices, and carefully match forage demand in the affected areas from livestock and wildlife with forage availability.

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Table 3-1. Summary of pixel value ranges in successive row width intervals (m) along each of the 3 transects used as training data sets in the supervised classification.

Transect		Number of Rows, Pixels, and Row Width					
		1 Row	2 Rows	3 Rows	4 Rows	5 Rows	6 Rows
		41 Pixels	82 Pixels	123 Pixels	164 Pixels	205 Pixels	246 Pixels
		0.75 m	1.50 m	2.25 m	3.00 m	3.75 m	4.50 m
RP31	Range	142-176 (34)	141-176 (35)	138-176 (38)	138-176 (38)	138-177 (39)	138-177 (39)
	(x±SD)	156.4+6.9	156.7+7.4	155.9+7.8	155.5+7.7	155.5+7.7	156.0+7.7
RP30	Range	150-184 (34)	150-184 (34)	150-184 (34)	150-184 (34)	150-186 (36)	150-186 (36)
	(x±SD)	172.8+6.3	172.6+5.9	172.9+5.7	173.1+5.1	173.2+4.9	173.3+4.8
RP04	Range	107-161 (54)	91-161 (70)	91-161 (70)	91-161 (70)	91-162 (71)	91-166 (75)
	(x±SD)	135.7+15.5	132.6+15.1	132.2+14.4	133.1+14.5	133.5+14.8	134.1+14.3
Average change in range of pixel values (%) relative to 1 row			**+ 9**	**+ 11**	**+ 11**	**+ 14**	**+ 15**

Table 3-2. Change in spatial extent of grassland (ha) in 5 sub-areas from 1958 to 1998. Values adjusted by a mean difference of 15.4% based on validation exercise (Figure 3-3).

	Unadjusted Areas of Grassland (ha)						Adjusted Values (ha)
Sub-areas	**1958**	**1958 to 1974**	**1974**	**1974 to 1998**	**1998**	**1958 to 1998**	**1958 to 1998**
Elk	440	-192	248	-60	188	-252 (-57.3%)	-213 (-48.4%)
South Elk	134	-52	82	-38	44	-90 (-67.2%)	-76 (-56.7%)
Between Elk & Radiant	71	-6	65	-34	31	-40 (-56.3%)	-33 (-46.5%)
West Radiant	360	-116	244	-101	143	-217 (-60.3%)	-183 (-50.8%)
Radiant	309	-123	186	-42	144	-165 (-53.4%)	-139 (-45.0%)
Total	**1313**	-488	**825**	-276	**549**	-764 (-58.1%)	-646 (-49.2%)

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Table 3-3. Change in carrying capacity (AUMs) over the 40 yr study period. Carrying capacities are **adjusted** for the over-estimation of area associated with shrub cover compared to the unadjusted carrying capacity 1958 to 1998.

	Adjusted Carrying Capacity (AUMs)						Unadjusted CC (AUMs)
Sub-areas	**1958**	**1958 to 1974**	**1974**	**1974 to 1998**	**1998**	**1958 to 1998**	**1958 to1998**
Elk	722.1	-315.1	407.0	-98.5	308.6	-413.6 (-57.3%)	-488.9 (-67.7%)
South Elk	219.9	-85.3	134.6	-62.4	72.2	-147.7 (-67.2%)	-174.6 (-79.4%)
Between	116.5	-9.8	106.7	-55.8	50.9	-65.6 (-56.3%)	-77.6 (-66.6%)
West Radiant	590.8	-190.4	400.5	-165.8	234.7	-356.1 (-60.3%)	-421.0 (-60.3%)
Radiant	507.1	-201.9	305.3	-68.9	236.3	-270.8 (-53.4%)	-320.1 (-71.3%)
Total	**2156.6**	**-802.6**	**1354.0**	**-451.3**	**902.7**	**-1253.9 (-58.1%)**	**-1482.2 (-68.7%)**

Figure 3-1. Natural subregion map from Alberta Parks Service, Management Support Division 1994 showing the location of the study area, indicated by arrow in the Upper Foothills sub-region.

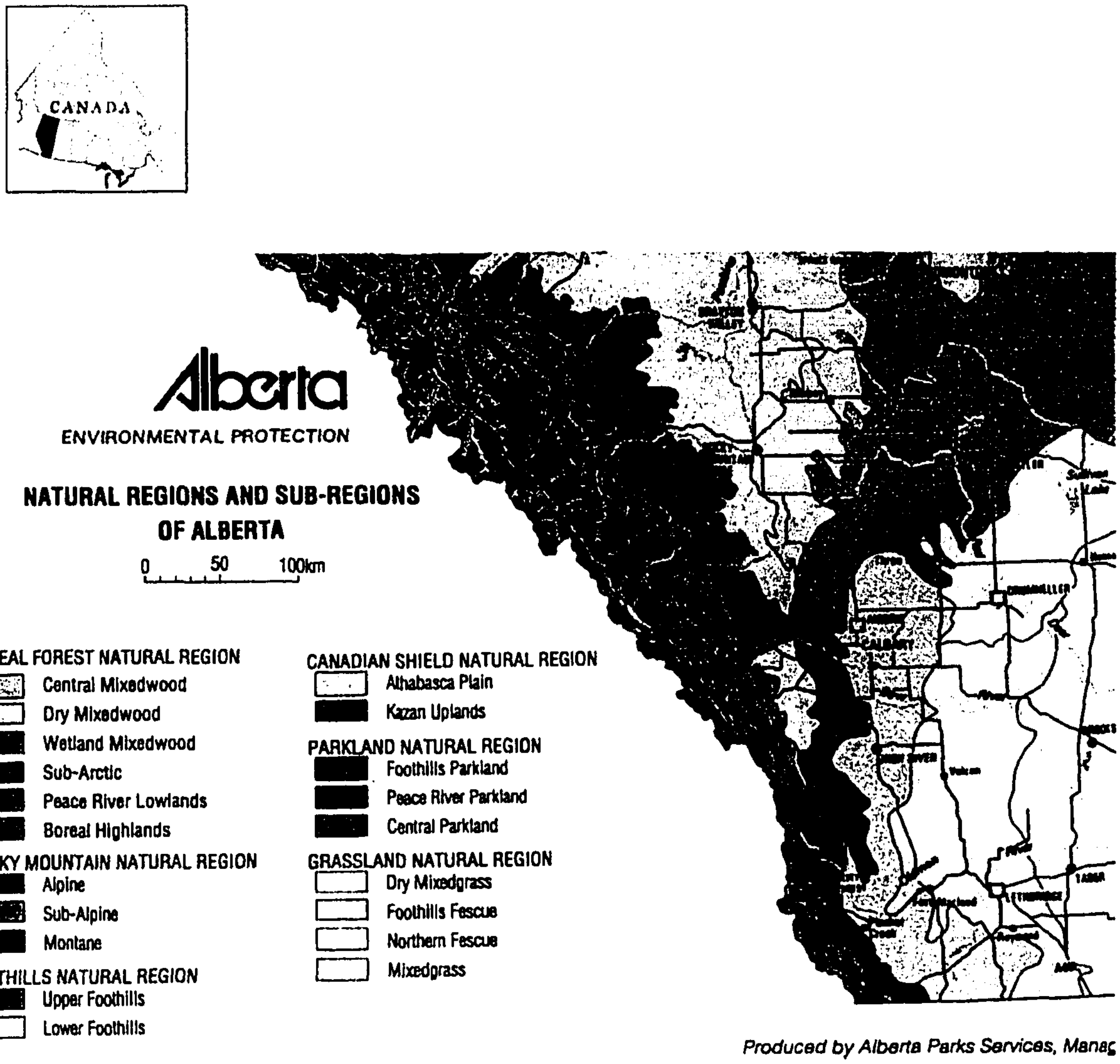

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Figure 3-2. Flowchart of GIS analysis used in the determining change in spatial extent of grassland.

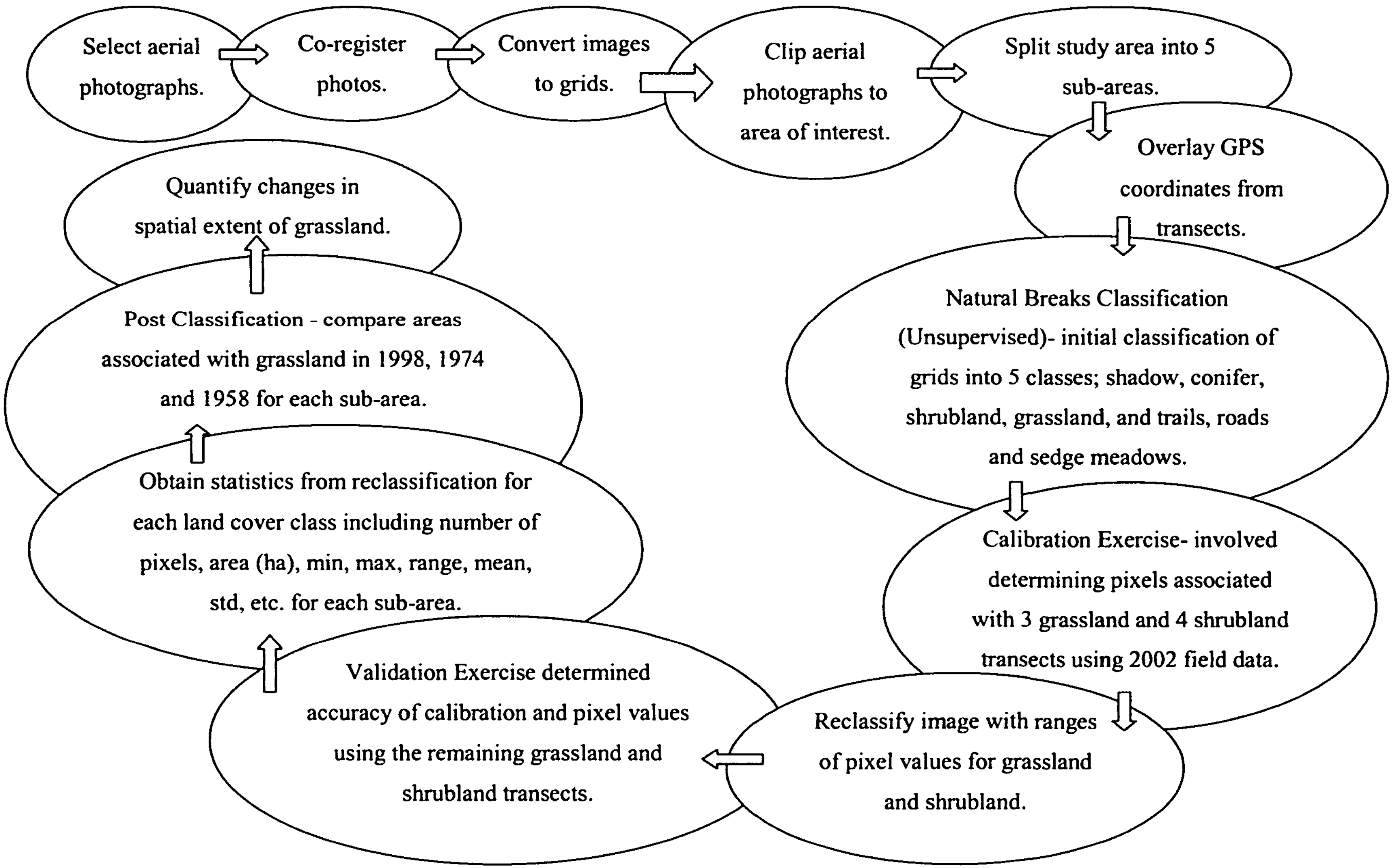

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Figure 3-3. Relationship between actual and predicted shrub canopy cover on the transects used for validation (SE = 3.5%, RMSE = 0.12, p < 0.0001) (n=26). Average difference between predicted and actual = 0.15. The dotted line represents a 1:1 slope.

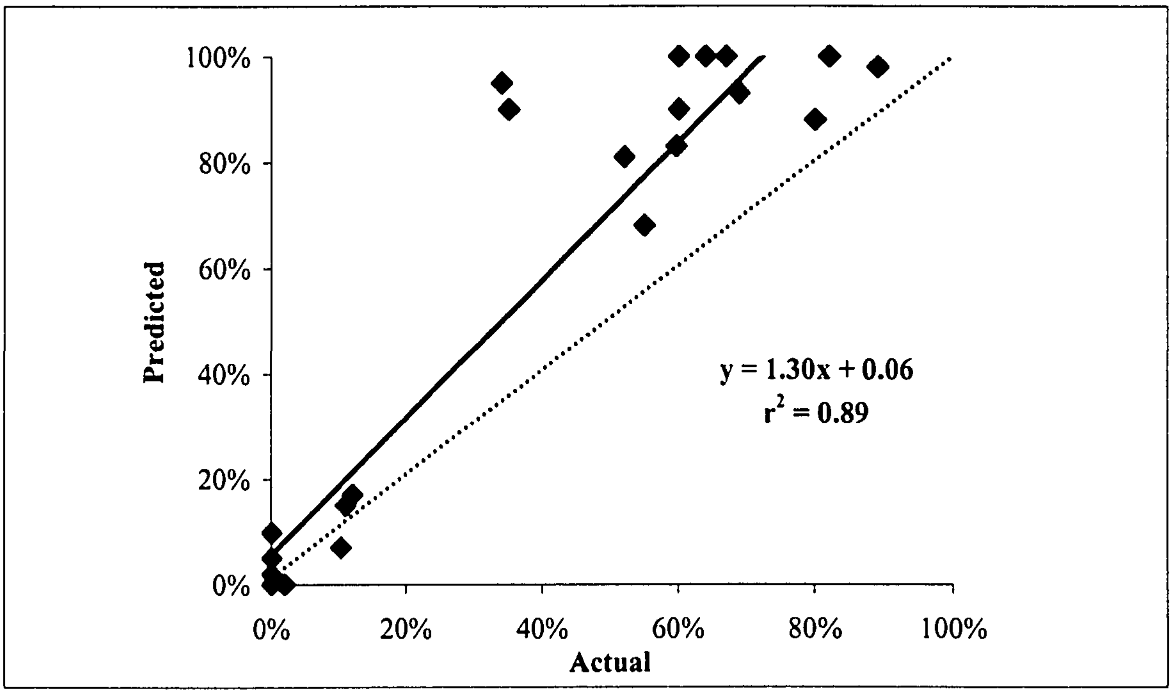

Figure 3-4. Area of grassland quantified in each of the 3 years (1958, 1974, and 1998) over a 40 yr period (1958 – 1998).

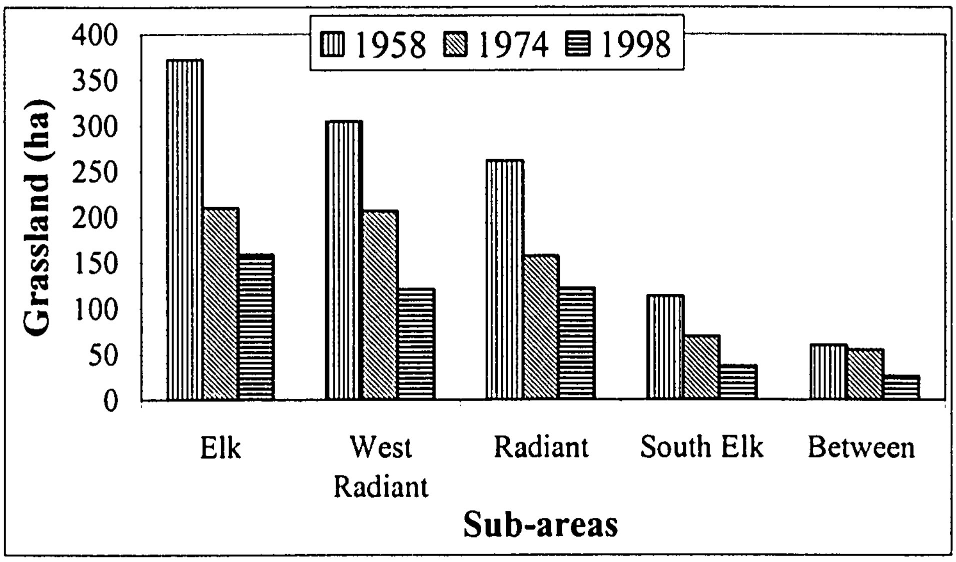

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Plate 3-1. Scanned black and white aerial photograph of study area in 1998 with transect locations collected by GPS.

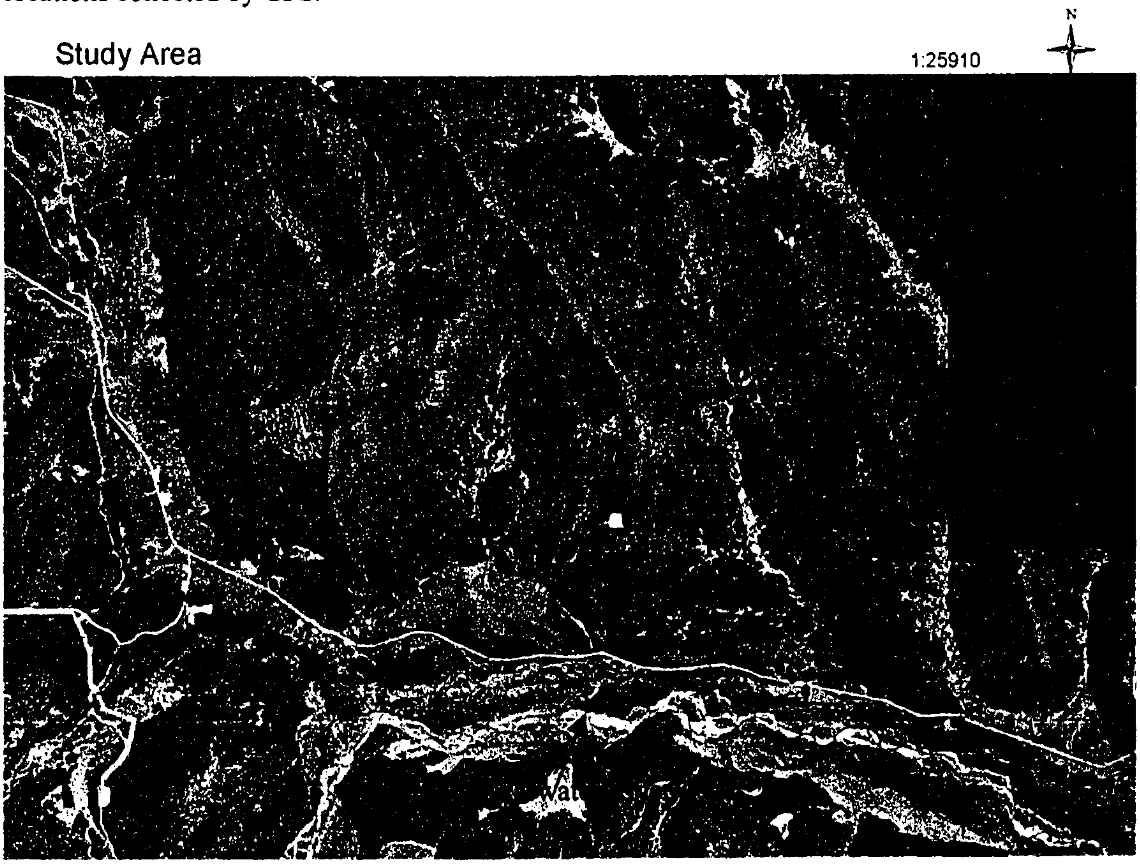

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Plate 3-2. Comparison of 6 field transects located within a portion of the Radiant Creek aerial photograph from 1958 and 1998 (bottom). Light areas represent grassland, darker grey areas shrubland and the very dark area conifer. Note Radiant Creek and the forestry trunk road running through the photos.

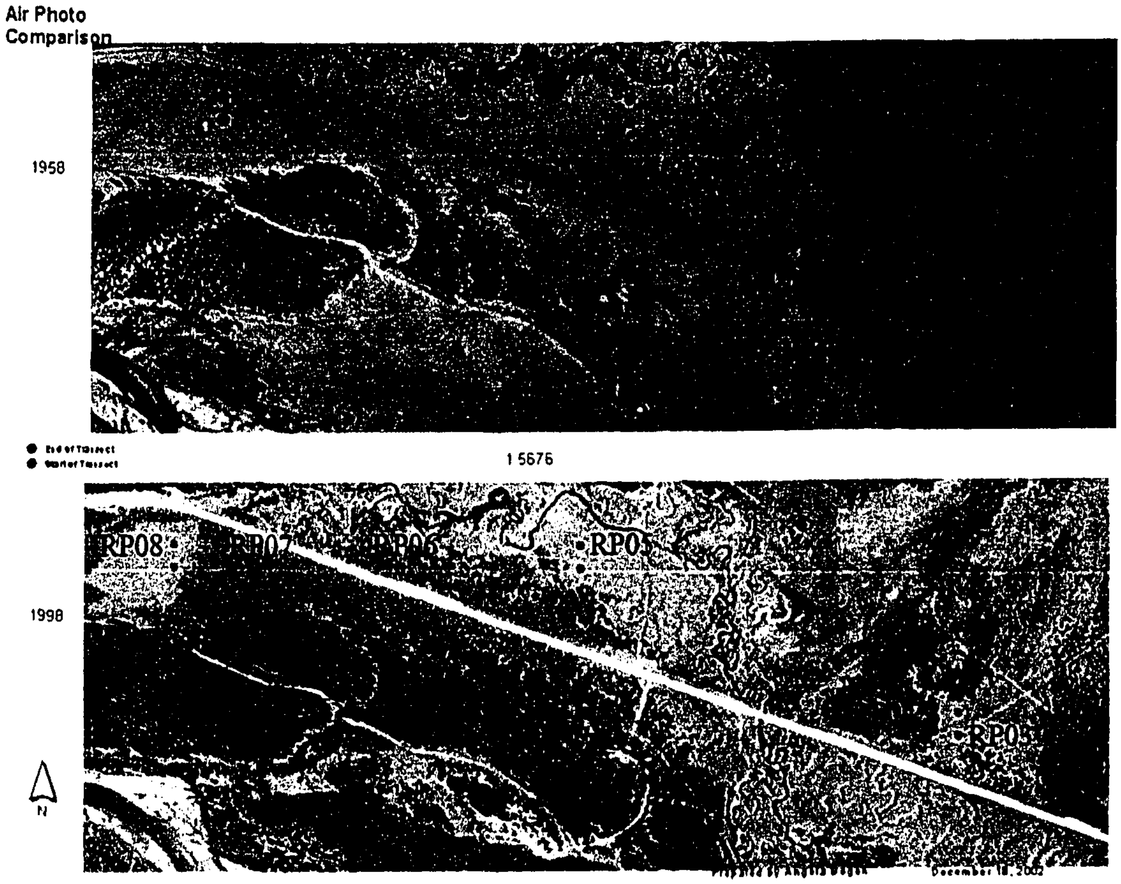

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Plate 3-3. Scanned aerial photographs (left) and grids (right) from 1998 (top) and 1958 (bottom) of Radiant Creek after reclassification, yellow represents grassland.

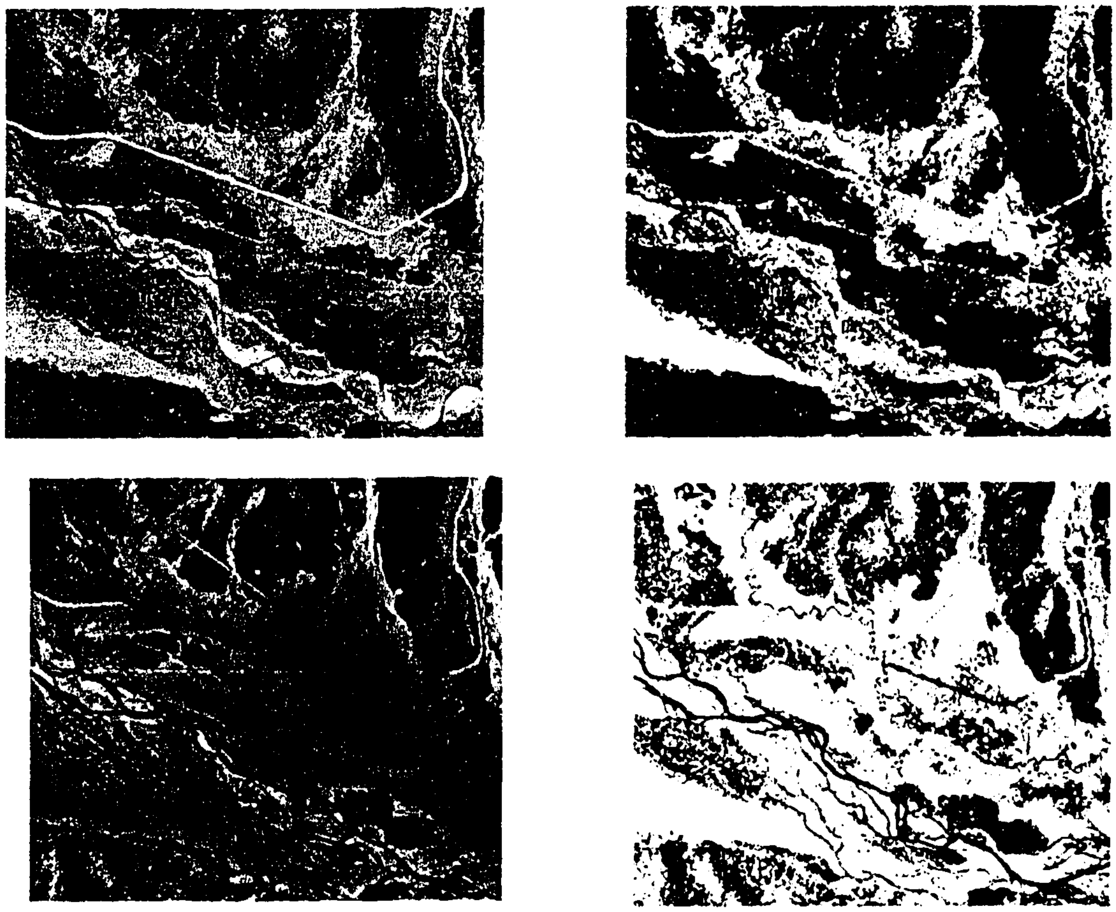

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Plate 3-4. Example of two field transects and the Elk Creek grazing exclosure overlayed on a grid within the Elk Creek sub-area from 1998 after supervised reclassification. Yellow pixels represent grassland and light green pixels shrubland.

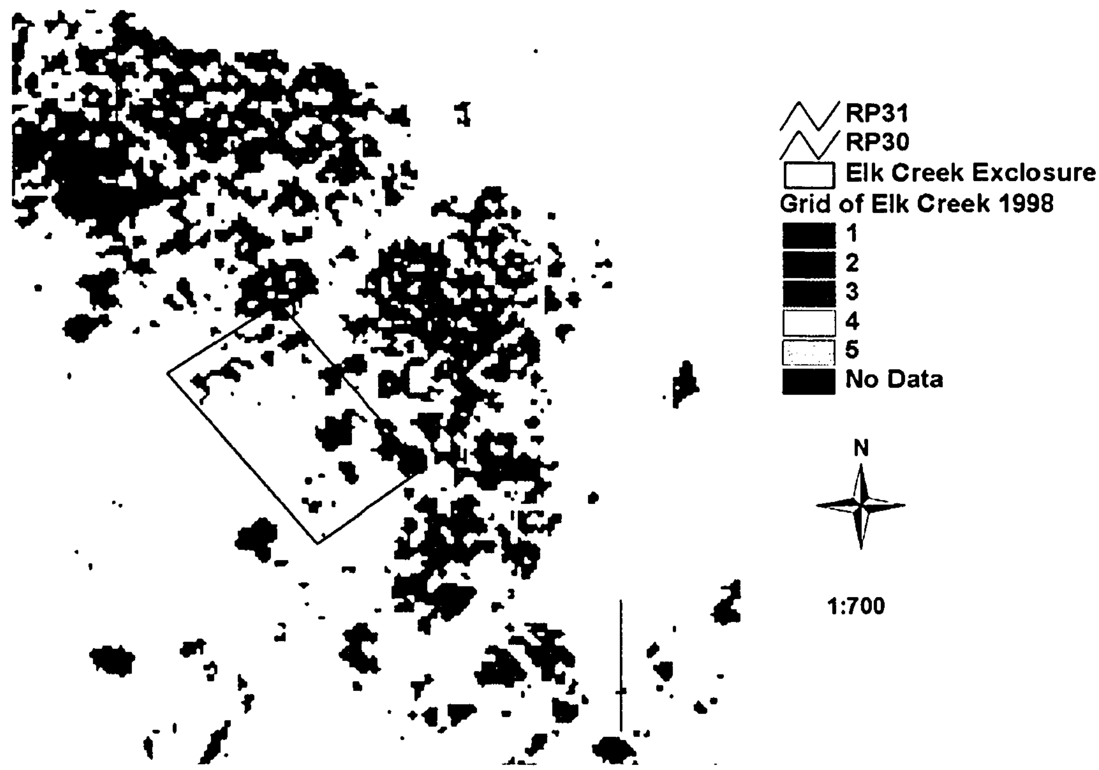

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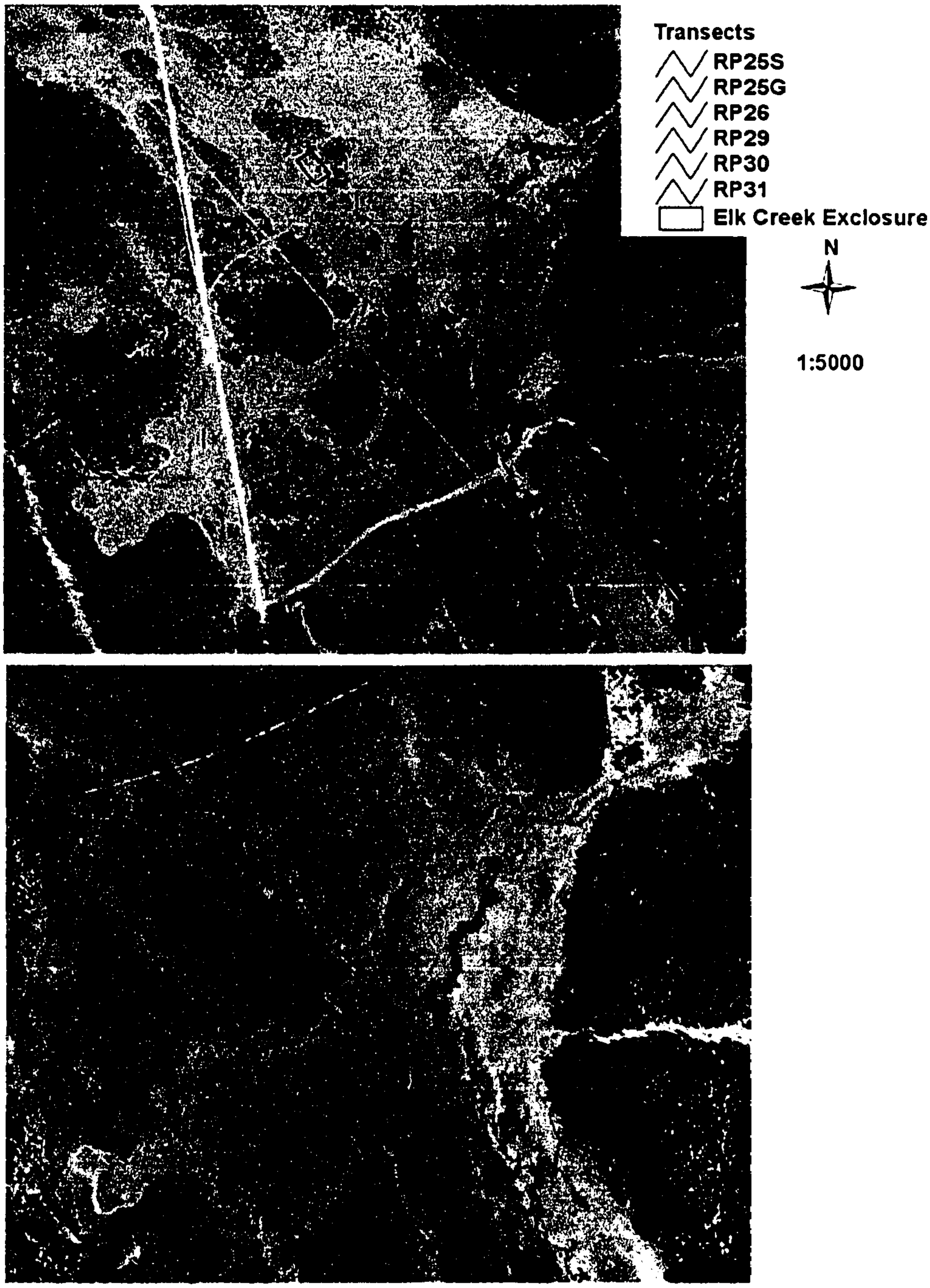
Plate 3-5. Location of field transects and the Elk Creek grazing exclosure overlayed on 1998 scanned aerial photography.



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4. Ecological Responses of Grasslands to Shrub Encroachment in the Rocky Mountain Forest Reserve

4.1 Introduction

Expansion of woody species is a concern for land managers on rangelands in North America (Adams et al. 1992, Bork et al. 1996, Bai et al. 2001). These increases have been attributed to many causes including climate change (Emmanuel et al. 1985, Sturm et al. 2001), atmospheric CO_2 enrichment (Archer et al. 1988), over-grazing (Humphrey 1958, Dunwiddie 1977, West 1993, Jeltsch et al 1997), nitrogen deposition (Kochy and Wilson 2001), and fire suppression (Cook et al. 1994, Kay et al. 1994, Bork et al 1996, Willoughby 2001).

In the Rocky Mountain Forest Reserve (RMFR) of southwestern Alberta, the invasion of floodplain meadows by shrubland has also been relatively widespread (Johnston and Smoliak, 1968, de Groot 1998, Willoughby 2001, Chapter 3 – this volume). Expansion of woody vegetation along the southeast slopes is occurring at approximately 0.75 to 5.5% per year, with higher rates on moister sites (Johnston and Smoliak 1968, Bailey and Conrad 1978, O'Leary et al. 1989). Historically, low intensity-high frequency fires maintained these grassland communities (Kay et al. 1994, Roguea 1999) but with fire suppression, they have become dominated by shrubs (Bork et al. 1996, Willoughby 2001). Regulations were passed as early as 1832 to suppress wildfire and were strengthened in 1953 when the Alberta government increased the resources dedicated to timber protection (Murphy 1985). For most of the 20th century, the central Rockies ecosystem has been under a fire exclusion policy and created unnatural mean fire return intervals and fire cycles (Rogeau 1999).

Invasion of shrubs and trees into grasslands is of particular concern due to the potential loss of biodiversity and forage availability for livestock and wildlife (Adams et al. 1992, de

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Groot 1998, Willoughby 2001). Following the initial exclusion of livestock from the RMFR, livestock grazing was re-introduced in 1913 as a tool for controlling litter build up and reducing the fire hazard (Department of Interior 1910, 1911b, 1912, 1914a). Changes in vegetation from open grassland meadows to shrub invaded communities, however, significantly reduces the amount of forage available for elk and domestic livestock grazing (de Groot 1998), as well as its accessibility to herbivores by reducing the area of primary range (Bork et al. 1996, Willoughby 2001). A decrease in available forage also leads to an increased risk of degrading the condition of remaining open meadows under fixed livestock stocking rates, stable or increasing wild ungulate populations and increasing feral horses. Shrub species responsible for the encroachment such as bog birch (*Betula glandulosa* Michx.) are low in forage quality and considered less preferable habitat (de Groot 1998). Moreover, land use conflicts may intensify with encroachment, particularly in areas where competing land uses for grasslands are high and this preferred habitat type is scarce. The inherent concentration of forage and water in the foothill valleys inevitably creates challenges for achieving proper livestock distribution and consequently, localized overgrazing has been observed. Rangeland management planners are increasingly under pressure to reduce forage allocations to livestock due to grassland loss and the need to provide sufficient forage for wild ungulates.

Along with changes in forage supply, encroachment alters the ecology of floodplain meadows, of which an understanding of the basic ecological impacts is required. Shrub encroachment has been associated with changes in herbaceous composition and a decline in species diversity (Willoughby 2001), reduced grass productivity (Stuart-Hill and Tainton 1989) and loss of habitat for wildlife (Adams et al. 1992). The loss of native bunchgrasses such as rough fescue (*Festuca campestris* Torr.) and tufted hairgrass [(*Deschampsia cespitosa* (L.) Beauv.] is a particular concern in the RMFR because they are valued as a primary source of forage (Johnston and Smoliak 1968). Moreover, increased grazing pressure on remaining

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grasslands increases the likelihood of their degradation, typified by the loss of native species and replacement with exotics, including weeds and tame forages (Adams et al. 2003).

This study was designed to examine the impact of shrub encroachment on the ecology of invaded native grassland community types in the RMFR. This was accomplished through a retrospective study contrasting the characteristics of open meadows (grasslands) with those of plant communities having various amounts of shrub cover, which in turn, is attributed to different degrees of shrub encroachment. Previous work (Chapter 3 - this volume) indicated the area in question had undergone a 49% decrease in grassland between 1958 and 1998, reinforcing the assumption that recent shrub expansion has occurred. Specific research objectives were to determine the empirical relationship between shrub overstory characteristics (density, cover, height, and stand age) and important understory characteristics such as vegetational species composition, diversity, bunchgrass abundance, and above-ground herbaceous primary production. This information will enable the implementation of improved long-term management strategies that deal with changes to the rangeland resource that ensure its sustainability.

4.2 Methods

4.2.1 Study Area

This research was conducted during 2001 and 2002 within floodplain meadows along the Clearwater River, a major tributary of the North Saskatchewan River within the northern portion of the RMFR in southwestern Alberta (Fig. 4-1). The climate here is substantially modified by the Rocky Mountains with cool summer temperatures and ameliorated winter temperatures expressing the cordilleran influence (Archibald 1982). Temperatures average 11.5°C in the

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summer and –6.0°C in the winter, which are milder and less extreme than the boreal forest and lower foothill subregions. Although the month of July is generally frost-free, June and August can receive up to three days of freezing temperatures (Poliquin 1968).

Actual precipitation collected by automated weather stations at the Clearwater Ranger Station, 13 km south of the study area, and Ram Falls, 11 km north of the study area, from 2001 to 2002, were substantially less than the long-term norm (Table 4-1). In addition, the 48-yr average (as of 2002) winter precipitation (October 31 to April 1) collected from a Sacramento gauge located in the study area, was 120 mm. In 2000 and 2001, annual precipitation collected from the automated weather stations was lower than the normals from Nordegg (Table 4-1), the nearest weather station, by 56.1 and 101.7 mm, respectively.

The research was conducted on the majority of primary range within the Elk Creek and Idlewilde Distribution Units (DUs) of the Clearwater Grazing Allotment (Figure 4-2). Based on the 1989 range management plan for the Clearwater Allotment, the permitted stocking rate for these two DUs is 690 and 412 Animal Units Months (AUMs), as compared to the calculated carrying capacity of 739 and 442 AUMs for Elk Creek and Idlewilde, respectively. The grazing season is from June 29 to July 18th for Idlewilde and July 19th to August 20th for Elk Creek. Wildlife surveys conducted on Elk and Radiant Creek (which are in the Idlewilde DU) during the winter of 2002 indicate 54 elk (Fish and Wildlife unpublished survey report). The greatest number of elk counted in the area was 85 in 1995 (Jim Allen ASRD, Wildlife Biologist, personal communication). Elk have little impact on forage in the area in the summer as they generally move away from the flats and up into the hills for calving and rearing. Deer and moose are frequent browsers and a small herd of feral horses roam the study area, the latter preferring the revegetated areas of pipelines and seismic lines. There are a number of recreational opportunities in the area, the most popular being random camping, which in turn, results in off-highway vehicle

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(OHV) use and equestrian activity, as well as lighter impacts such as fishing and hiking. The surrounding coniferous slopes, mainly comprised of lodgepole pine (*Pinus contorta* Loudon) are actively logged as part of a 100 year tenured forest management agreement with Weyerhaeuser. Oil and gas reserves in the area have led to the development of roads, pipelines, seismic lines, and other facilities to extract this resource.

Within the study area many different plant communities can be found and have been described (Willoughby 2001). These vary from wet sedge meadow to bog birch/rough fescue/bearberry [*Arctostaphylos uvi-ursi* (L.) Spreng.] plant community types found at the Elk Creek cattle exclosure. The two predominant grassland communities that comprise most of the study area and the primary range in question are rough fescue-tufted hairgrass and tufted hairgrass-sedge (*Carex* spp.) types. The former community is better drained while the latter is found at slightly lower elevations (Willoughby 2001). In the absence of fire, both plant communities eventually become dominated by bog birch and willow (*Salix myrtillifolia* Anderss.) (Willoughby 2001) in this region. With localized overgrazing, rough fescue and tufted hairgrass have decreased in abundance while Kentucky bluegrass (*Poa pratensis* L.) and dandelion (*Taraxacum officinale* Weber) have invaded (Willoughby 2001). Other shrub dominated communities within the study area include willow-bog birch/tufted hairgrass and willow-bog birch/rough fescue, both of which represent a higher nutrient and moisture regime (Willoughby 2001).

According to the Provincial Forest Fire Centre, only four fires, each less than 0.1 ha, have occurred in the study area since 1958, and the last major fire within the area was in the late 1800's (Rogeau 1999). No range improvement projects have been undertaken on this portion of the allotment from 1958 to 1998 (Alberta Forest Service 1989) except for a 2 ha brush mowing project (Bob Lenton, ASRD, Forest Officer, pers. comm.) at the south end of the study area. A

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larger brush mowing project was undertaken west of the study area on Idlewilde flats. Approximately 8 km south of the study area at 7-Mile Flats, a series of prescribed burns have been undertaken to reduce bog birch (Bork et al. 1996).

4.2.2 Experimental Design

Initially, 30-m transects were randomly placed in the spring of 2001 within shrub communities throughout the study area ranging from 0 to 65% shrub cover in an attempt to deliberately assess the leading edge of shrub encroachment. However, there were very few areas representing shrub canopy levels between 20 and 40%, and therefore, in 2002 the transects were established in selected plant communities (randomized internally) to range from open grassland to closed shrubland. In the spring of 2002, 33, 30-m transects were randomly positioned (for the most part) as pairs throughout approximately 11 km of valley bottom in the study area (Plate 4-1): 21 of these transects remained from 2001. Transects were placed to minimize variation in ecosite characteristics between shrub encroached and adjacent un-encroached sites. Although elevations varied slightly from the south to north end of the study area (1450 m to 1530 m), aspects were generally level with no slope to minimize site variability. The location of all transects was determined using a Garmin 12XL GPS (Appendix B).

Along each transect the canopy cover (%) of all grass and forb species, as well as moss and lichen, were estimated (Daubenmire 1958) within 15, 0.1m^2 quadrats positioned every 2 m for a total sampling area equaling 1.5 m^2. A 1 m^2 quadrat was nested overtop each of the smaller quadrats to estimate shrub canopy cover for a total sampling area of 15 m^2 per transect. The minimum sampling area for the above measurements was determined by preliminary analysis of response data averages with incremental sample sizes (Appendix E). From the vegetation inventory data, vascular species richness was calculated by counting the total number of species

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in all 0.1 m^2 quadrats (no/1.5 m^2) per transect and species diversity was determined using the Shannon-Wiener diversity index ($H=P_i\log[P_i]$) (Bonham 1989). The density of dominant native bunchgrass, including rough fescue and tufted hairgrass, was determined for each community by counting these plants individually in a 60 m^2 belted plot (2-m x 30-m) centered along each transect. The density (no./60 m^2) and height (cm) of individual shrubs [bog birch, willow, shrubby cinquefoil (*Potentilla fruticosa* L.)] and trees [lodgepole pine and white spruce (*Picea glauca* (Moench) Voss)] were recorded along the same belted plots. Woody species considered to be 'encroaching' were bog birch, willow, lodgepole pine and white spruce. Tree encroachment was minimal throughout the selected plant communities and insufficient for analysis.

Above-ground current annual grass and forb production at each transect was estimated for the plant community by sampling a 0.5 m^2 quadrat within each of 2, 1.5 m^2 range production cages used to prevent grazing. Plant biomass in all quadrats was harvested to ground level in mid August after peak growth was reached and before advanced senescence could occur. Samples were sorted into grasses and forbs, subsequently oven-dried at 30°C to constant mass and weighed (g/m^2). Production clippings taken within 0.5 m^2 quadrats are standard methodology for the Public Lands and Forests Division (AFS, 1992). However, preliminary data were also used once again to ensure the appropriate area was being sampled to obtain representative production and canopy cover estimates for each plant community (Appendix E).

Shrubs were aged along each belted transect by sampling three bog birch and three willow plants, where present. Shrubs representative of the plant community were selected based on their size along the transect in an effort to get a mean age indicative of the shrubs within each community. Shrubs were cut off just above the ground, left at least one month to dry, sanded with No. 300 grit paper and then rings were counted using a hand lens (Stokes and Smiley 1968, Thierry Varem-Sanders, Canada Forest Service, pers. comm.). Sanding the wood burnishes the cellulose in the late wood and does a better job of bringing out the rings than dyes (Thierry

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Varem-Sanders, Canada Forest Service, pers. comm.).

Adjacent to each transect, a 1-m deep soil pit was dug and soil morphological information was collected on the identity of horizons as well as their depth and thickness. Other information collected included manual texturing of horizons and the assessment of soil structure, consistency, rooting depths and other characteristics. The presence of calcium carbonate through HCl tests and mottles or gleying were also recorded (Appendix F).

4.3 Statistical Analyses

Although all data were checked for normality prior to analysis, the presence of non-normal data were not transformed because regression analysis is robust to non-normal data (P. Blenis, pers.comm. 2003) and it is difficult to interpret regressions using transformed data in an ecologically meaningful way. For non-linear relationships a curvilinear regression was performed. All differences were considered significant at $P<0.01$, unless otherwise indicated. Quantitative regression analyses (Proc Reg., SAS 2004) were conducted on the data to assess relationships between the understory and shrub overstory. Given the retrospective nature of this investigation, varying levels of *in-situ* shrub encroachment (i.e., overstory characteristics) and their impacts on the understory were considered the treatments at each site. Experimental units were the transects representative of individual plant communities, with each representing one data point in the regression procedure.

The understory variables selected for monitoring were species diversity, bunchgrass canopy cover and density, as well as herbage production, because they are important indicators of overall biodiversity and rangeland health, the condition and demographic status of the bunchgrass population, and the amount of available forage, respectively. Varying degrees of shrub canopy cover closure can lead to changes in understory vegetational composition (Willoughby 2001),

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reduced bunchgrass cover (Adams et al. 1992) and production declines (Stuart-Hill and Tainton 1989, Willoughby 2001), and was therefore considered the primary independent variable in this investigation. However, shrub density, height and age were also assessed for their use in predicting understory responses. Overstory-understory relationships were assessed empirically using Goodness-of-Fit (r^2) criteria; and helped establish which overstory variables accounted for the most variation in (or had the greatest influence on) the key understory variables. A correlation matrix was used to assess redundancy between variables. Data were assessed separately for 2001 and 2002 because of potential confoundment introduced by changes in a few transect locations between years.

To determine if shrub height reflected shrub age, the relationship between these two variables was explored by regressing mean shrub height against mean shrub age (willow and bog birch) from each transect. This was done because height is a relatively easy parameter to estimate and if height was correlated with age, height could then be used as a surrogate to assess the time elapsed since shrub encroachment. In addition, soil Ah and LFH thickness were regressed against mean shrub canopy cover for each transect in an attempt to discern whether there was a shift in the diagnostic grassland mineral horizon (Ah) toward the development of a surficial organic horizon (LFH) with increasing shrub abundance.

4.4 Results and Discussion

4.4.1 Understory Production and Carrying Capacity

The plots sampled in 2001 showed that total herbage (grass plus forb) production exhibited a significant ($P<0.01$) negative curvilinear relationship ($r^2=0.63$) with increasing shrub canopy

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cover (Figure 4-3, A). One year later in 2002 (Figure 4-3, B), a similar but stronger negative relationship existed (r^2=0.79; P<0.01). During 2002, herbage production decreased from a mean of 6,629 $kg.ha^{-1}$ in the grassland transects (< 12% shrub cover) to a mean of 2,797 $kg.ha^{-1}$ in shrubland (> 33%), representing an 58% decline. Herbage production in the absence of shrubs (i.e. 0% cover) varied greatly, between 9,964 and 4,152 $kg.ha^{-1}$. Given that the field sampling strategy used was designed to maintain relatively similar ecosite conditions among all plots, the variation in production was attributed to variability in the composition and condition of plant communities throughout the study area. Finally, herbage production collected in 2001 was generally lower (x= 2,865 $kg.ha^{-1}$) than one year later in 2002 (x= 4,655 $kg.ha^{-1}$), likely due to lower preceeding and growing season moisture (Table 4-1).

Similar to total herbage, grass production alone exhibited a significant negative curvilinear relationship (r^2=0.55; P<0.01) with increasing shrub canopy cover in both 2001 and 2002 (Figure 4-4). However, these relationships were slightly weaker than those observed for total herbage. This result suggests the forb component of the plots investigated were also negatively related to increasing shrub cover, leading to total herbage responses being a superior indicator of shrub encroachment impacts on understory production.

Indicators used to estimate the amount of forage associated with shrub invaded communities may be a useful tool for land managers to quickly and easily make decisions regarding the sustainability of the range resource. Results of this study indicated that mean shrub canopy cover (r^2=0.79) was a superior overstory variable compared to shrub density (r^2=0.52) and height (r^2=0.42) in accounting for the most variation in herbage production (Table 4-2). Given that land managers can quantify shrub canopy cover (i.e. through ocular assessments) more easily than understory biomass, this information can be used to readily estimate forage production from the relationship between these variables ($y=6401.80e^{-0.0122x}$). Shrub canopy cover alters the

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microclimate in the understory and reduces light, nutrient and water availability through competition, thereby affecting the ability of the understory to maintain growth, vigor, and ultimately production (Hobbs and Mooney 1986).

Herbage-shrub relationships in 2002 (Figure 4-3, B) appear to exhibit a threshold in shrub cover >11% and <34%. Below 12%, production is variable (presumably from ecosite variation) but relatively high (greater than 4,000 kg.ha^{-1}). In contrast, above 33% shrub cover, maximum herbage production is less than this value, reflecting the overriding negative influence of shrubs regardless of ecosite potential. Moreover, the non-linear relationship between understory production and shrub cover indicates that greater production losses occur with small increases in shrub when the plant community is relatively open grassland with less than 12% shrub cover. Conversely, once shrub cover exceeds 34%, reductions in herbage occur at a lower rate with progressively greater shrub cover. For example, at 10, 40 and 70% shrub cover, estimated reductions in herbage production were 69, 48, and 33 kg ha^{-1} for each additional 1% increase in shrub cover. The gap in sites between 12 and 33% shrub cover may be the result of what is occurring below ground and not do simply to their location. The initial sampling strategy attempted to locate transects in 20 to 40% shrub cover but were modified because sites throughout the study area either had a well established shrub canopy or not.

Among the independent variables examined comparing different shrub species, mean total shrub canopy cover (the combined average of willow and bog birch) explained the most variation (r^2=0.79) in herbage production (Table 4-3) and was greater than either shrub species assessed alone. This result indicates that both bog birch and willow impacted understory production. However, among the different species of shrubs examined individually (Table 4-3), bog birch cover had the most significant (P<0.01) relationship with herbage production (r^2=0.64). Although willow was also significantly (P<0.01) associated with herbage production, this variable

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explained less than a third of the variation (r^2=0.24) in production (Table 4-3). The relationship between shrubby cinquefoil and production was not explored because they were highly correlated.

Bog birch is a dominant shrub within these floodplain meadows (Table 4-4), and therefore might be expected to have the most influence on the understory. The 2002 vegetation data showed that the proportion of all shrub cover consisting of bog birch was 70%, with willow comprising the remaining 30% (Table 4-4). While this trend may explain in part why bog birch had the most significant impact on herbage production, these results nevertheless reinforce the importance of bog birch as a determinant of understory herbaceous production.

4.4.2 Components of Diversity and Stability of Grassland Vegetation

Species diversity data collected in 2001 did not exhibit a significant relationship with mean shrub canopy cover, density or height (P=0.65) (Figure 4-5, A): species diversity reached a maximum of 2.86 under 75% shrub cover. In 2002, species diversity increased as shrub canopy cover increased from 0 to 80% (P<0.01). The sample size for the 2002 data was 33, increased from 21 the previous year, which may have helped establish the significant relationship found in 2002.

Overall, species diversity in 2002 increased as shrub canopy cover (P<0.01) increased from 0% to 80%. While the species diversity within transects with no shrubs was 2.37, this value increased to as high as 3.0 under 64% shrub cover. The increase in species diversity was likely due to the shrub canopy creating more fine spatial-scale variation in microclimate. This would have increased overall understory vegetational species diversity by allowing shade tolerant plants to establish directly under encroaching shrubs while the interspaces between shrubs were still

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occupied by grassland species. A slight "leveling-off" in diversity as shrub canopy cover further increased (>80%) may be interpreted and could potentially be due to the disappearance of grassland species as they were out-competed by shade tolerant species for resources (i.e., lack of light in the understory (Willoughby 2001). Generally, the formation of a closed shrub canopy previously dominated by grassland has a negative effect on the abundance of herbaceous grassland species, as it greatly reduces the amount of light reaching the understory and presumably alters most microclimatic features (Hobbs and Mooney 1985).

The results of this study indicate that shrubs were positively associated with species diversity within these floodplain meadows. However, further investigation should be undertaken to determine if prolonged shading of the understory would result in a significant decline in species diversity, and the specific shrub canopy cover level at which diversity declines due to excessive shading. Species diversity is an important component of an ecosystem, and is an important attribute for which the RMFR may be managed for under the proposed biodiversity monitoring strategy (ASRD 2003). Plant communities that are less diverse are generally less resilient or likely to recover from disturbances such as fire and grazing (Tilman and Downing 1994). There was no significant positive or negative relationship between species richness and the overstory variables (except height; $P<0.01$, $r^2=0.2$) in either year of the study and may suggest that when shrub canopy increases, plants adapted to high light begin to disappear and are replaced by more shade tolerant ones (Knowles et al. 1999). It was noted during data collection that rough fescue plants in well-established shrub stands were found only at the base of some shrubs and were reduced in basal area and generally appeared less robust and healthy.

Two other key understory variables, bunchgrass density and canopy cover, exhibited declines ($P<0.01$) in relation to increasing overstory canopy cover in both 2001 and 2002 (Figures 4-6 and 4-7). Bunchgrass density and cover varied widely among communities sampled with low

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shrub cover, which may be a result of varied grazing levels or localized overgrazing. Heavy grazing or grazing during the growing season can decrease the abundance of key bunchgrasses and reduce their vigour (Willms et al. 1988). These two variables are important as indicators of the health of the bunchgrass population, which is the primary forage source for domestic livestock and elk in the area. The variation/range in bunchgrass density could also be attributed to different plant communities containing lower densities of bunchgrasses at the north end of the study area, and could be the result of many factors including variation in the duration and timing of defoliation. The decline in bunchgrass cover and density in well established shrub stands may indicate intolerance to more intense and prolonged shading from shrubs. The most immediate effect of an open overstory is increased light and associated root gaps in the soil, which may increase water and nutrient availability.

The overstory variable that appeared to have the greatest influence on species diversity, bunchgrass density and cover, was consistently shrub canopy cover (r^2=0.28, r^2=0.47, and r^2=0.71, respectively: Table 4-2). However, shrub density and height were also significantly associated with these understory variables, indicating all three could be used as tools by land managers to monitor the health of the bunchgrass population as shrub encroachment occurs. For example, it may be easier and quicker to determine shrub density through field spot counts than estimate canopy cover.

The soil information collected at most of the plots within the study area, under open grassland and shrub dominated sites, were difficult to interpret, because the horizon directly below the AH or LFH exhibited several unique characteristics. Therefore, the soils were not classified and the B horizon referred to in the soil profiles may be interpreted as another A horizon (Appendix F). At many transects (RP17, RP18, RP24, RP25G, RP25S, RP26, RP30, RP31) this horizon was an orangey to yellowish color and at times was difficult to identify

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because of its limited size. This may be due to the soil development likely resulting from periodic flooding of these valley bottoms in the past. The influence of a high water table in the development of these floodplain meadows was also evident in the soils documented, as an organic mat at the top of the soil horizons seemed to be associated with some of the wetter grassland sites and the presence of mottling and gleying was noted in many soils (i.e., gleying: RP03, RP08, RP09, RP12, RP22, and RP27; orange mottles: RP02, RP04, RP07, RP16, RP18, RP23, RP25G, RP25S, RP29, and RP30). Charcoal deposits were also noted in the horizon directly below the AH and LFH in the soil pits at RP03 (45 cm depth), RP05 (22 cm), and RP17 (16 cm), suggesting the occurrence of periodic historical fire events between major floodplain accretion events. Another interesting note is the orange coloured ashy platy layer at several locations (RP01, RP03, RP08, RP09, RP10G, and RP14). This appears to be an eluviated layer but may have been deposited by periodic flooding instead of originating on site. Carbonates were detected in some of the soil pits (RPRP10G, RP19G, RP26, and RP31) but within most of the sites carbonates were not reached even after digging to 1 m depth.

The thickness of the LFH and Ah horizons had opposite relationships to increasing mean shrub canopy cover (Figure 4-8): LFH thickness generally increased and Ah thickness decreased. Between the two, more variation in LFH (r^2=0.51;P<0.0001) than Ah (r^2=0.27; P>0.01) thickness was explained by mean shrub canopy cover. Although the variation in Ah may be attributed to the length of time of soil development following the last major deposition of sediment with flooding, the variation in LFH depth is more likely the result of direct shrub encroachment and the deposition of more recalcitrant leaf and woody biomass. In general, a shrub cover greater than 10% appeared to result in a decrease in Ah depth.

When the empirical relationship between mean shrub height and shrub age was explored, relatively little variation in age was explained by shrub height and the relationship was bordering

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on significance (r^2=0.18; P=0.05). As mean shrub height increased linearly so did mean shrub age: however, shrub age did not increase beyond 15 yrs potentially indicating a limited life span of above ground stems (Figure 4-9). It was initially thought that this positive relationship would provide a useful tool (e.g. shrub height) for quantifying the time period over which encroachment had occurred. However, because shrub stems grow and die back regularly, stem demographics may be a representation of individual stem development rather than actual shrub age. Both bog birch and willow have below ground creeping roots (de Groot 1998) and these produce the individual ramets visible above ground. Indeed, based on the data collected here the mean shrub age at each transect ranged from only 6 to 15 yrs. Given that both air photos and verbal accounts indicate shrubs have been dominating at least some of these areas for much longer than that, the individual age of above ground ramets are unlikely to represent true shrub ages and thus, the length of time the understory has been subject to shrub encroachment.

One last relationship explored was the presence of shrubby cinquefoil in well-established bog birch and willow stands. This was done because shrubby cinquefoil is commonly associated with bunchgrasses throughout southwest Alberta, particularly rough fescue (Willoughby 2001). There was no significant relationship between shrubby cinquefoil density, height or canopy cover, and mean shrub (willow and bog birch) canopy cover, density or height. However, it was noted that in the majority of grassland transects shrubby cinquefoil was quite well-established, while in stands dominated by bog birch and willow, the cinquefoil shrubs were not as large or vigorous (healthy), similar to that observed for rough fescue. Field observations therefore suggest that this species was shade intolerant and that tall shrubs could eventually out-compete the shorter-statured shrubby cinquefoil along with the bunchgrasses.

The canopy cover and density data collected in 2001 and 2002 showed very little evidence of tree encroachment onto these floodplain meadows. However, the areas sampled were in the

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middle of the valley bottoms and as noted from the air photos used in the study mapping shrub expansion since 1958 (Chapter 3, this volume), a considerable amount of tree encroachment had occurred on the lower slopes of the adjacent foothills. More specifically, lodgepole pine had encroached along the edges of the valleys and expanded on certain level areas throughout the valley floor. It may therefore be reasonable to predict that the shrubs encroaching onto these floodplain meadows will eventually be overtaken on suitable ecosites by the larger, later seral lodgepole pine.

4.5 Conclusions

Retrospective assessment of the relationship between the shrub overstory and the herbaceous understory within floodplain meadows of the Alberta RMFR indicated that the production of herbage was negatively associated with increasing shrub abundance including cover, density, and height. Reductions in bunchgrass density and cover from species such as tufted hairgrass and rough fescue were also associated with increasing overstory variables. Species diversity was positively associated with shrubs however a "leveling-off" in diversity may be interpreted at shrub cover levels above 70%. Among shrubs, bog birch had the greatest apparent influence on the understory variables due to its prevalence throughout these meadows. Similarly, mean shrub cover rather than shrub density or height generally had the greatest influence on the understory.

These results highlight the potential problems associated with the absence of fire from these ecosystems, which includes the potential for increased conflict between domestic livestock grazing and elk for forage as shrub expansion continues. It also provides land managers with tools to help assess the health of this important forage resource and ensure its sustainability under increasing land use impacts and changes in plant communities. Further analysis to investigate the affects of not only shrub expansion but tree encroachment may provide more information on the

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total impact of fire suppression on these floodplain meadows. The combination of both shrub and tree encroachment is likely to have a more serious impact at a greater spatial level. An alternative perspective of shrub and tree encroachment may be a shift in management strategies. An increase in lodgepole pine would benefit the timber companies making it even more difficult to promote projects (i.e.; burning or mowing) that enhance and ensure the survival of the grasslands that elk and domestic livestock rely on.

The negative relationship between herbage production and shrubs implies that the carrying capacity associated with the study area will decrease. The carrying capacity of 2,157 AUM's in 1958 calculated using production values collected from this study (Chapter 3, this volume) was followed by a 58% decrease over 40 yrs, much of which can be attributed to an increase in shrubs. The implications of further encroachment will have dramatic impacts on the health of the grassland communities or the allowable stocking rates for domestic livestock. Land managers should implement more rigorous means to reduce shrub cover throughout these floodplain meadows and ideally strive to maintain shrub cover between 12 and 33%, which as indicated by this study, appears to be the level for sustaining a significant attribute of these grasslands: forage production for domestic livestock and wildlife.

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Table 4-1. Average annual precipitation (mm) data from 2000 to 2002 collected from automated weather stations at the Clearwater Ranger Station (CRS), Ram Falls (RF), and Nordegg.

Location	2000	2001	2002	30 Yr Mean
CRS & RF	404.3	335.3	414.6	--
Nordegg	--	391.4	516.3	596.3

Table 4-2. Comparison of overstory shrub cover, density and height for predicting various understory characteristics in foothills meadows of the RMFR.

Understory Response Variable	Year	Overstory (Independent) Shrub Variables		
		Shrub Cover	Shrub Density	Shrub Height
Species Diversity	2001	N/S	N/S	N/S
	2002	$r^2 = 0.27$	$r^2 = 0.22$	$r^2 = 0.23$
		$P = 0.002$	$P = 0.006$	$P = 0.004$
Bunchgrass Canopy Cover	2001	$r^2 = 0.48$	$r^2 = 0.31$	$r^2 = 0.20$
		$P = 0.0005$	$P = 0.0085$	$P = 0.04$
	2002	$r^2 = 0.71$	$r^2 = 0.23$	$r^2 = 0.52$
		$P < 0.0001$	$P = 0.004$	$P < 0.0001$
Bunchgrass Density	2001	$r^2 = 0.34$	$r^2 = 0.21$	N/S
		$P = 0.0069$	$P = 0.0344$	
	2002	$r^2 = 0.47$	$r^2 = 0.37$	$r^2 = 0.40$
		$P < 0.0001$	$P = 0.0002$	$P < 0.0001$
Grass Production	2001	$r^2 = 0.55$	$r^2 = 0.40$	$r^2 = 0.24$
		$P < 0.0001$	$P = 0.0021$	$P = 0.0242$
	2002	$r^2 = 0.82$	$r^2 = 0.55$	$r^2 = 0.39$
		$P < 0.0001$	$P < 0.0001$	$P < 0.0001$
Herbage Production	2001	$r^2 = 0.65$	$r^2 = 0.46$	$r^2 = 0.30$
		$P < 0.0001$	$P = 0.0007$	$P = 0.0009$
	2002	$r^2 = 0.79$	$r^2 = 0.62$	$r^2 = 0.42$
		$P < 0.0001$	$P < 0.0001$	$P < 0.0001$

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Table 4-3. Summary of the most influential shrub overstory cover characteristics in relation to the understory characteristics assessed.

Understory (Response) Variables	Overstory Canopy Cover (Independent Variable)			
	Total Shrubs	Bog Birch	Willow	Shrubby Cinquefoil
Species Diversity	$r^2 = 0.28$ $P = 0.002$	$r^2 = 0.19$ $P = 0.01$	$r^2 = 0.17$ $P = 0.02$	N/S
Bunchgrass Canopy Cover	$r^2 = 0.71$ $P < 0.0001$	$r^2 = 0.58$ $P < 0.0001$	$r^2 = 0.32$ $P = 0.0007$	N/S
Bunchgrass Density	$r^2 = 0.47$ $P < 0.0001$	$r^2 = 0.44$ $P < 0.0001$	$r^2 = 0.16$ $P = 0.02$	N/S
Grass Production	$r^2 = 0.71$ $P < 0.0001$	$r^2 = 0.59$ $P < 0.0001$	$r^2 = 0.30$ $P = 0.0009$	N/S
Herbage Production	$r^2 = 0.52$ $P < 0.0001$	$r^2 = 0.64$ $P < 0.0001$	$r^2 = 0.24$ $P < 0.01$	N/S

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Table 4-4. Abundance of bog birch and willow on transects across the study area.

	Mean Actual Shrub Cover (%)			Proportion of Total	
Transects	Both Shrubs	Bog Birch	Willow	Bog Birch	Willow
RP01	0.0	0.0	0.0	0.0	0.0
RP02	68.9	47.6	33.3	0.7	0.3
RP03	0.0	0.0	0.0	0.0	0.0
RP04	81.9	42.7	41.2	0.5	0.5
RP05	0.0	0.0	0.0	0.0	0.0
RP06	54.5	17.1	38.2	0.3	0.7
RP07	59.6	24.3	41.5	0.4	0.6
RP08	0.0	0.0	0.0	0.0	0.0
RP09	0.0	0.0	0.0	0.0	0.0
RP10G	0.0	0.0	0.0	0.0	0.0
RP10S	82.4	62.3	35.2	0.8	0.2
RP11	64.2	47.3	29.2	0.7	0.3
RP12	0.0	0.0	0.0	0.0	0.0
RP13	91.5	15.9	86.7	0.2	0.8
RP14	0.0	0.0	0.0	0.0	0.0
RP15	10.8	10.8	0.0	1.0	0.0
RP16	80.3	79.8	9.8	1.0	0.0
RP17	68.9	58.1	20.5	0.8	0.2
RP18	0.0	0.0	0.0	0.0	0.0
RP19G	0.0	0.0	0.0	0.0	0.0
RP19S	80.8	78.0	6.7	1.0	0.0
RP21	0.0	0.0	0.0	0.0	0.0
RP22	51.6	50.4	2.3	1.0	0.0
RP23	11.0	11.0	0.0	1.0	0.0
RP24	67.1	55.0	1.9	0.8	0.2
RP25G	0.0	0.0	0.0	0.0	0.0
RP25S	59.33	59.33	0.0	1.0	0.0
RP26	60.1	60.1	0.0	1.0	0.0
RP27	89.3	72.9	27.0	0.8	0.0
RP28	1.8	1.8	0.0	1.0	0.0
RP29	34.4	34.4	0.0	1.0	0.0
RP30	10.3	10.3	0.0	1.0	0.0
RP31	35.1	35.1	0.0	1.0	0.0
Mean	**55.7%**	**41.6%**	**29.8%**	**0.7**	**0.3**

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Figure 4-1. Natural subregion map from Alberta Parks Service, Management Support Division (1994) showing the location of the study area, indicated by the arrow.

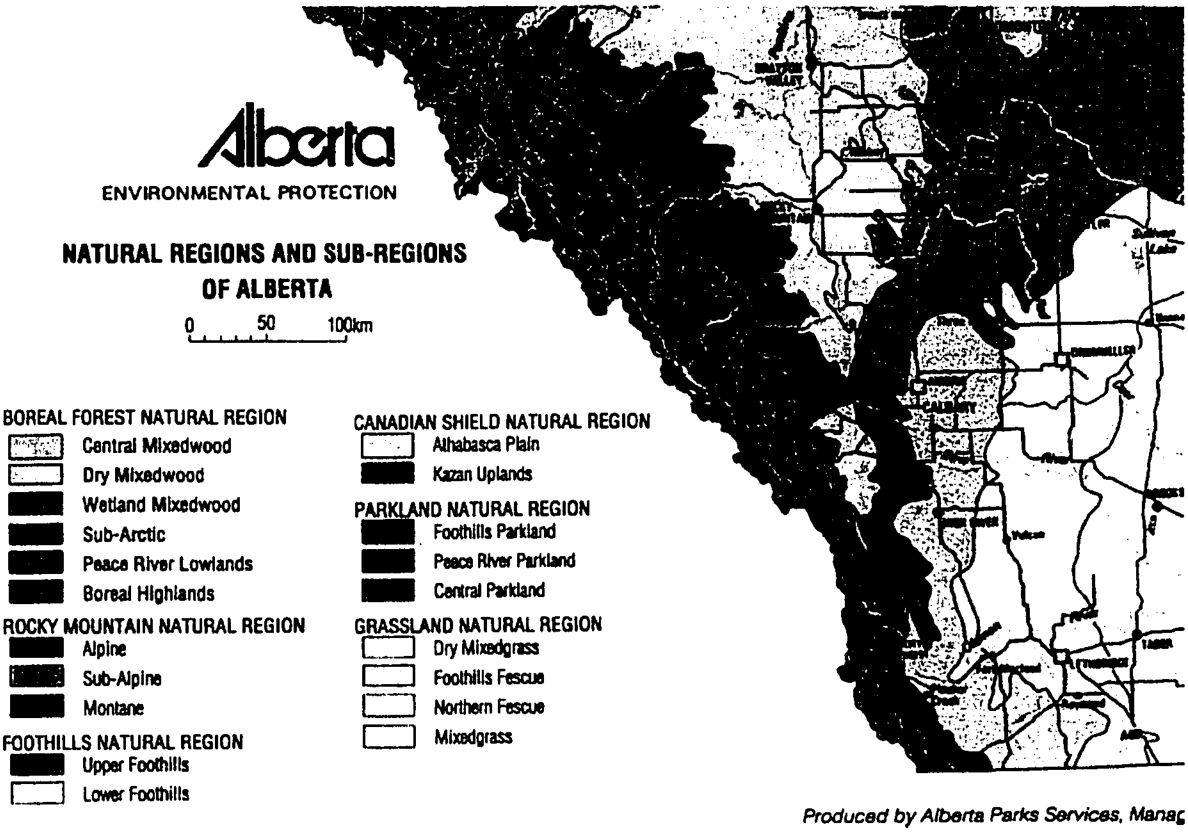

Figure 4-2. Regional map of the Grazing Allotments (gray) located west of Rocky Mountain House, Alberta. The Clearwater Allotment is No. 82.

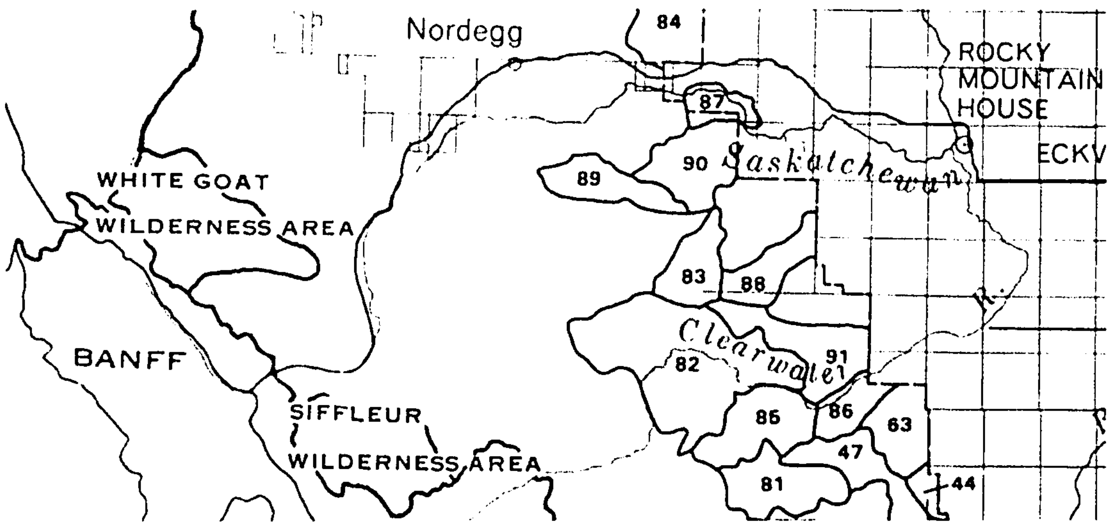

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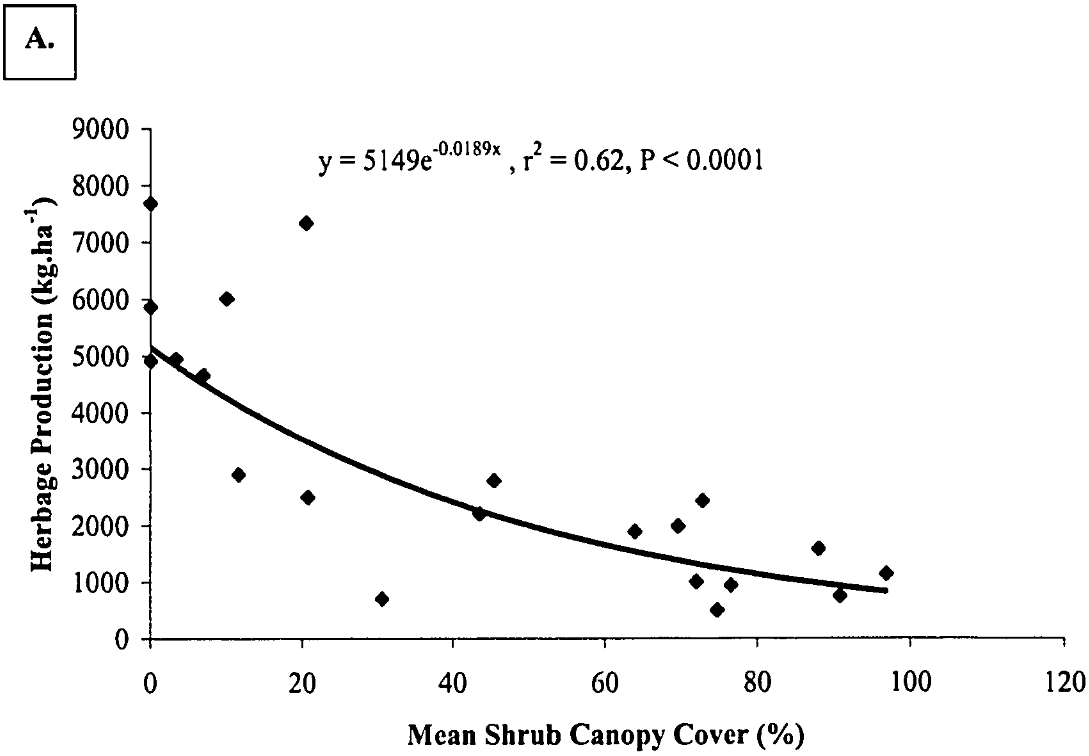
Figure 4-3. Relationship between total herbage (grass and forb) production and mean shrub cover (%) in each of (A) 2001 (n=21) and (B) 2002 (n=33).



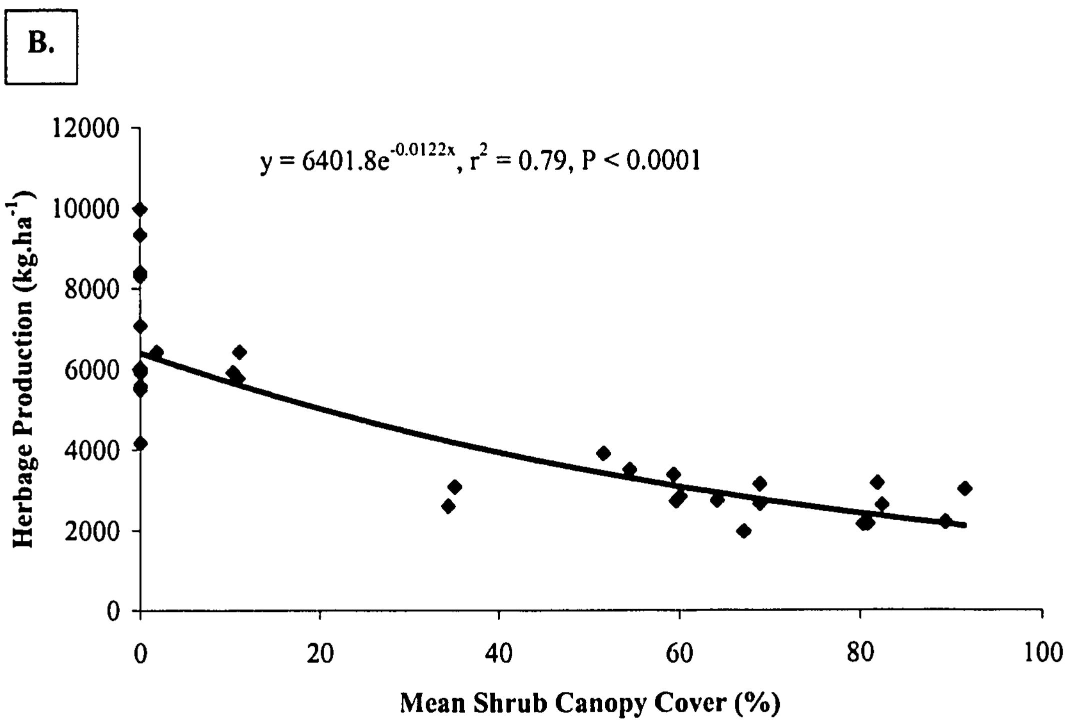


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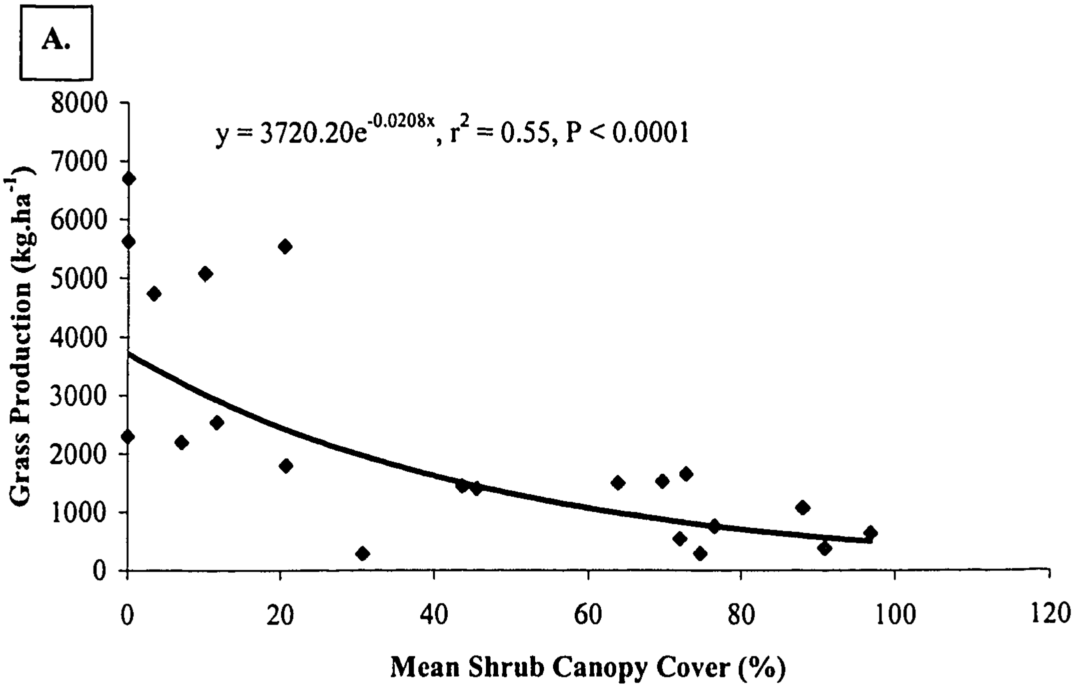
Figure 4-4. Relationship between grass production and shrub cover in each of (A) 2001 (n=21) and (B) 2002 (n=33).



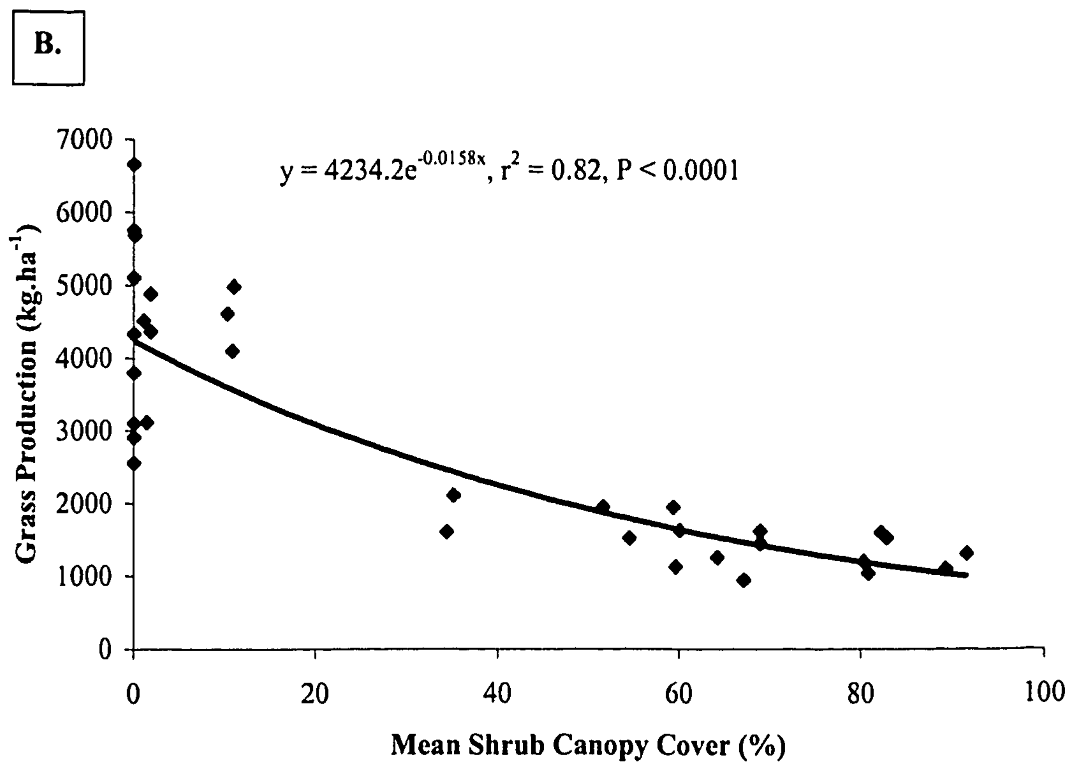


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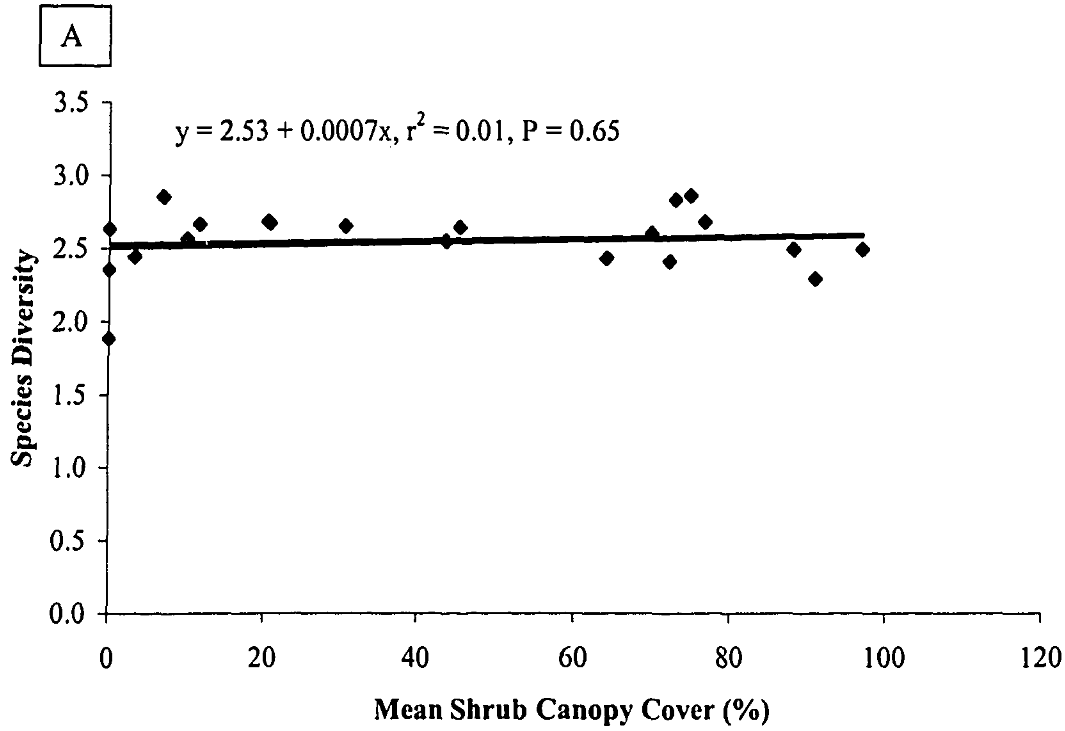
Figure 4-5. Relationship between species diversity and shrub cover for each of (A) 2001 (n=21) and (B) 2002 (n=33).



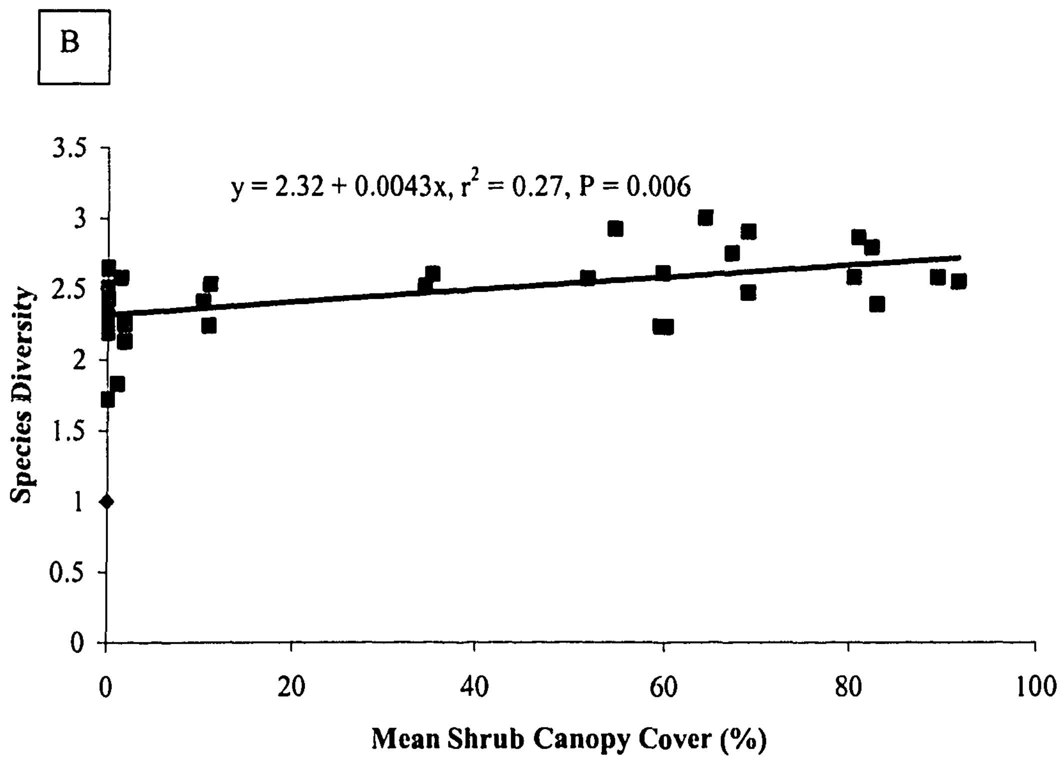


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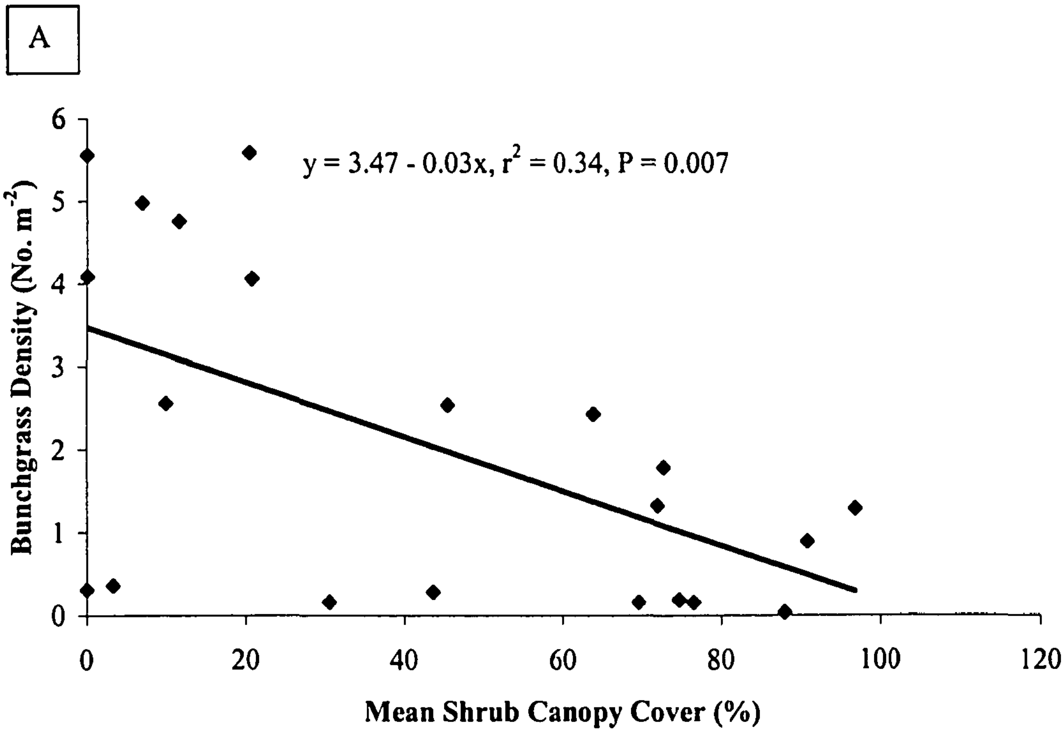
Figure 4-6. Relationship between bunchgrass density and mean shrub cover for each of (A) 2001 (n=21) and (B) 2002 (n=33).



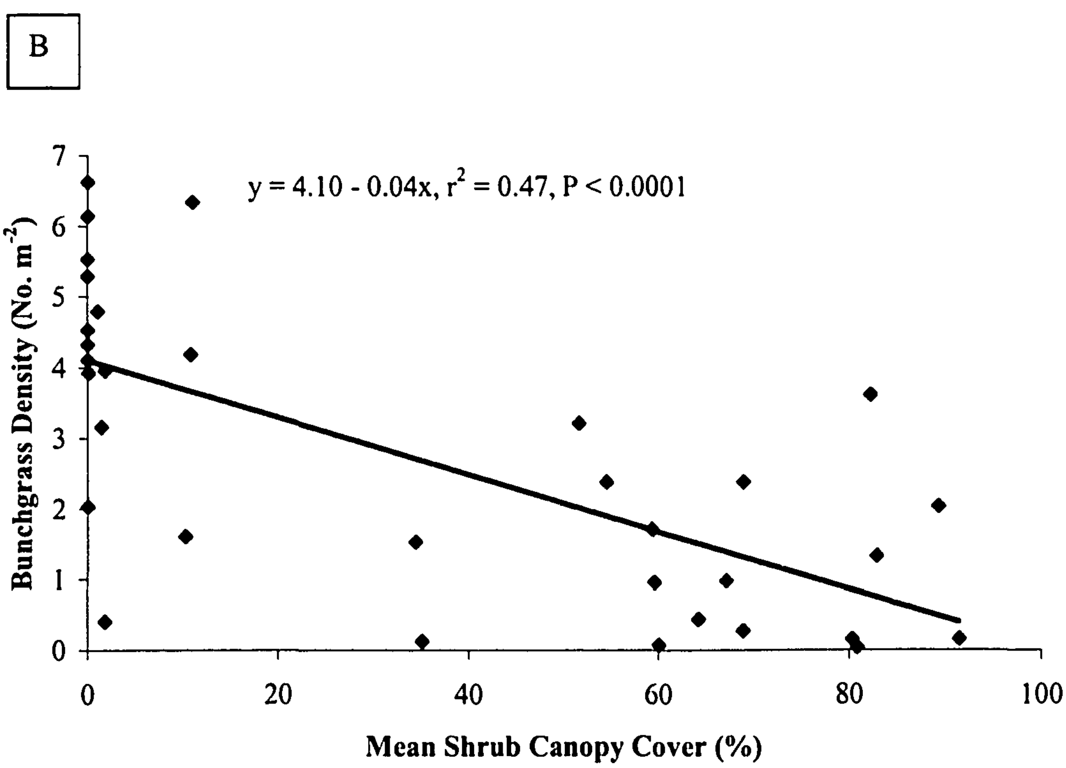


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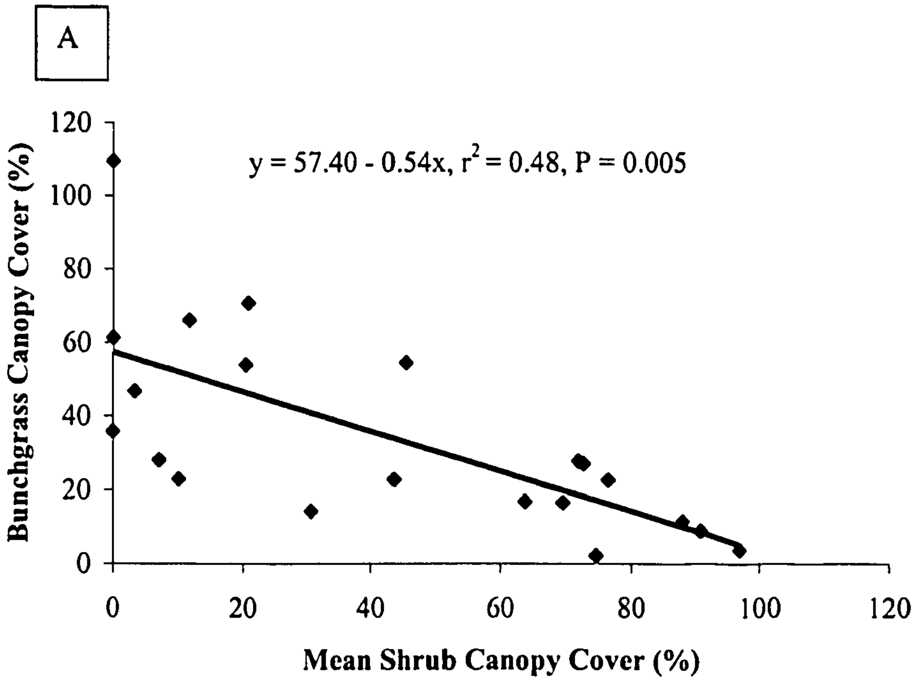
Figure 4-7. Relationship between bunchgrass canopy cover and mean shrub cover for each of (A) 2001 (n=21) and (B) 2002 (n=33).



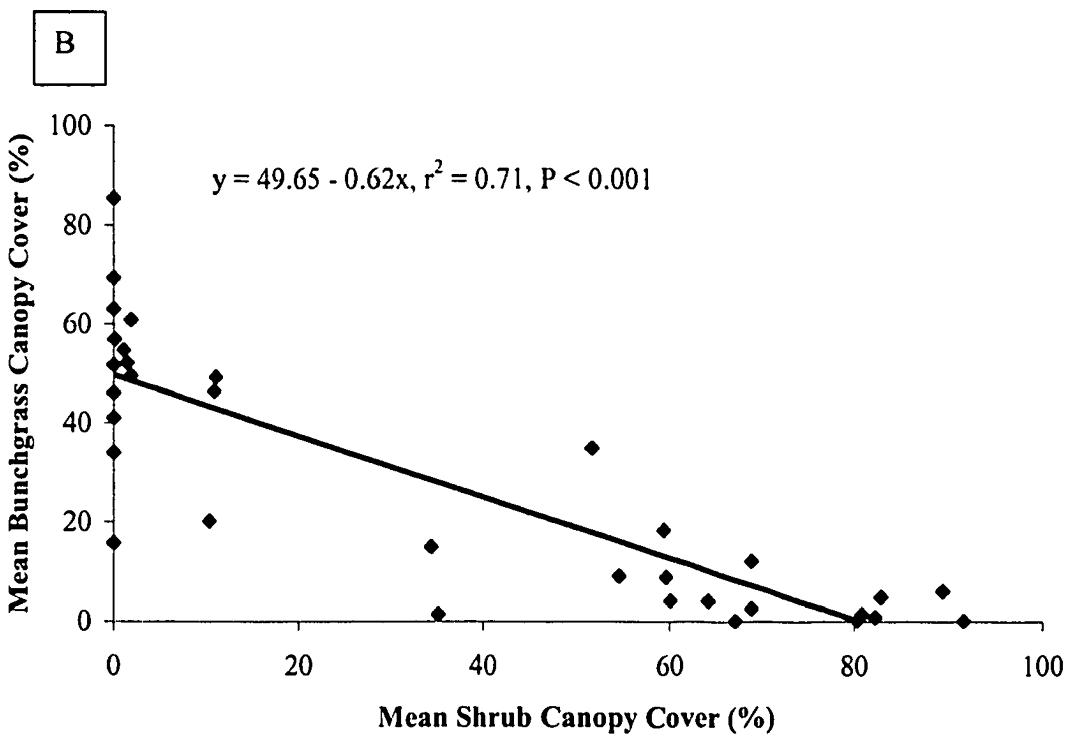


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Figure 4-8. Relationship between mean shrub canopy cover and the thickness of Ah (dashed) and LFH (solid) soil horizons (n=33).

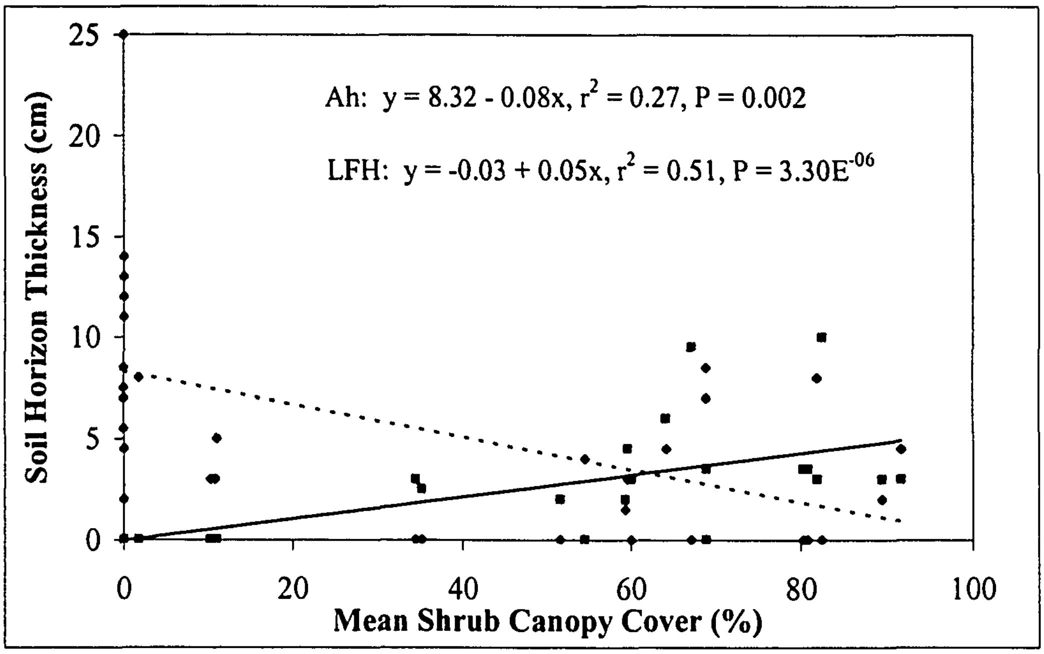

Figure 4-9. Relationship between mean shrub height (cm) and mean shrub age on individual transects (n=22).

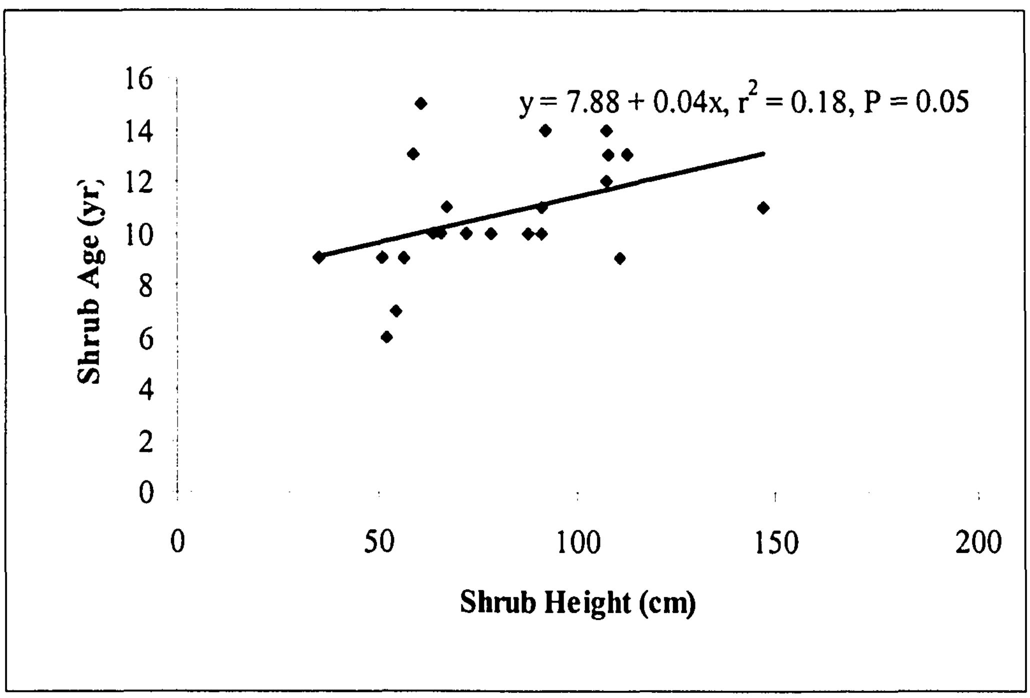

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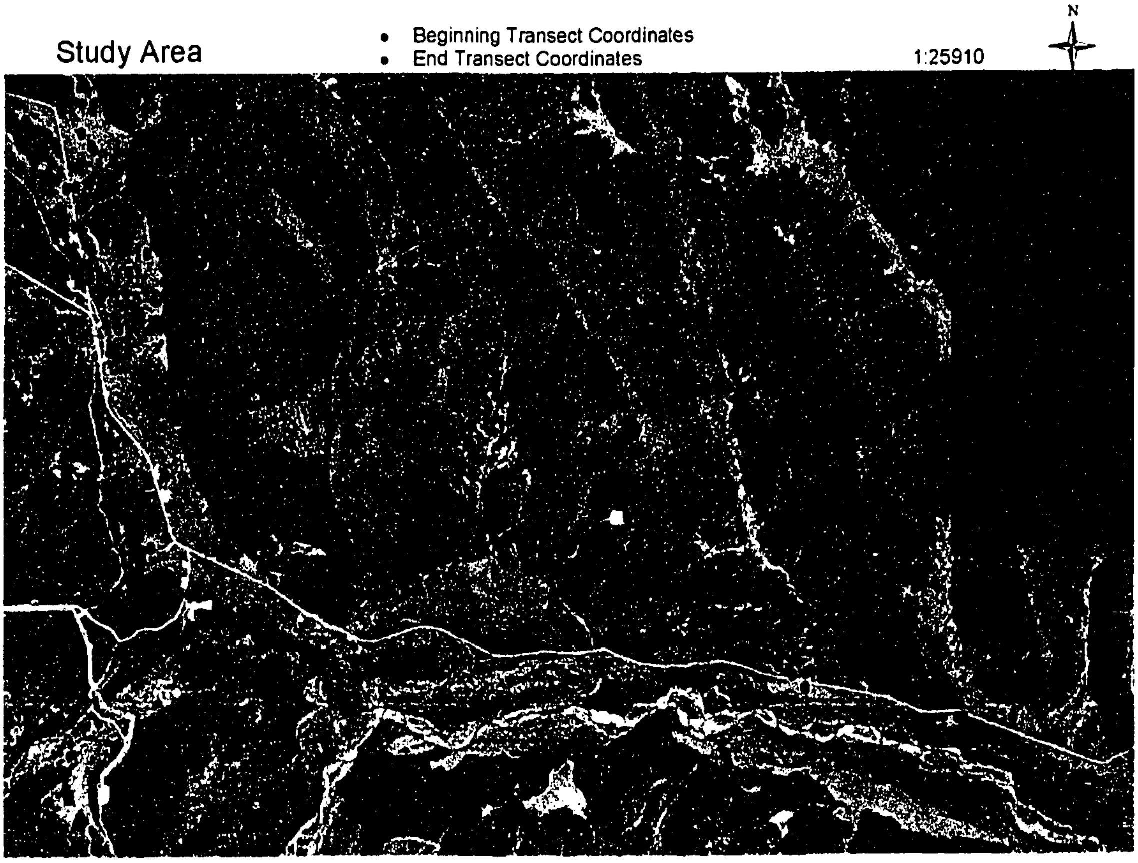
Plate 4-1. Study area and location on transects over approximately 11 km.



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5. Synthesis

Almost a century of fire suppression in the RMFR of Alberta has altered the mean fire return interval (Rogeau 1999) leading to an increase in woody vegetation (Kay et al. 1994, Bork et al. 1996, Rhemtulla et al. 2002, Willoughby 2001) and an associated reduction in grassland. The consequences of this change in the landscape have been an altering of the soil carbon sink (Gill and Burke 1999, Jackson et al. 2002), modification to the fuel source and changes in fire regime when prescribed burns are initiated (Wallace et al. 2003), as well as reduced herbage and corresponding carrying capacities for domestic livestock and wild ungulates such as elk (Adams et al. 1992, Willoughby 2001).

Quantifying the extent of woody plant encroachment is essential for implementing decisions able to effectively address land management needs consistent with the conservation of remaining native grassland in the RMFR. The landscape-based GIS study documented in Chapter 3 was successful in quantifying the amount of loss in grassland area and subsequent reduction in carrying capacity over a 40 yr period from 1958 to 1998. Results indicated a 49% decrease in total grassland area corresponding to 646 ha, across 5 sub-areas located in the valley bottoms of a portion of the Clearwater River basin. Aside from the concern of reduced landscape diversity with grassland reduction, a major implication of this decrease is the associated reduction in ungulate carrying capacity. The latter implicated an estimated loss of 1,482 AUM of grazing for domestic livestock and elk. This trend may increase the degree of competition between domestic livestock grazing in summer and elk that use the same areas in winter, thereby jeopardizing the health of remaining grasslands along these valley bottoms.

In Chapter 4, the synecological impact of shrub encroachment into meadow grasslands was investigated, with an emphasis on overstory-understory relationships. Herbage biomass

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production, which is one of the most important variables of rangeland communities (Willms and Jefferson 1993), was quantified under various levels of shrub abundance. Results from 2002 indicated that herbage levels tended to decline from an absolute high of 9,964 $kg.ha^{-1}$ with no shrub cover, to 1,948 $kg.ha^{-1}$ under 67% cover, representing an 80% decline. A non-linear relationship between herbage and shrubs indicated greater production losses occurred below 20% shrub cover and a lower rate of production loss when shrub cover exceeded 30%. These results suggest that proactive shrub control measures should be implemented either prior to, or at the onset of initial shrub invasion, in order to conserve grassland vegetation. Notably, a similar threshold was identified in a study on redberry juniper (*Juniperus pinchottii* Sudw.) invaded rangeland in western Texas (Uekert et al. 2001).

An important consideration for understanding the dynamics of shrub encroachment within the southeast slopes of western Alberta is identifying the key species responsible. The vegetation data collected indicated that the proportion of all shrub cover consisting of bog birch was 70%, with willow comprising the remaining 30%. This study found that the abundance of bog birch, particularly canopy cover, explained most of the variation in the understory variables, including species diversity, herbage production, and bunchgrass abundance. Not only is it important to understand the ecology of bog birch when implementing shrub control techniques in an economically feasible manner but provides crucial information on changes in habitat values for wildlife. Since bog birch has a chemical mechanism that deters browsing (de Groot 1998) its importance for wildlife may be far less than that of grassland.

Current research indicates that bog birch has a competitive advantage over other plants in the warmer growing conditions created after fire, and resprouting is enhanced by a fire-stimulated height growth response (de Groot and Wein 1999). Additional needs for research based on the above and the outcome of this study include a need for better information on methods of

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integrated shrub/bog birch control and studies documenting the resilience/recovery of invaded grassland following shrub removal.

While the potential benefits of controlling and/or reversing shrub encroachment on floodplain meadows are numerous, long-term planning is likely necessary by land managers to effectively deal with this problem. Without effective control measures the grassland component of these valley bottoms will be lost, or at a minimum, severely reduced in area. Although the Clearwater Grazing Allotment has been conservatively grazed since at least the last management plan was implemented in 1989 productivity is still decreasing. According to this study the carrying capacity for the study area has been reached with further shrub encroachment the health of these grasslands will be jeopardized if stocking rates are not changed. This marks a critical time for land managers to make key management decisions not only in this particular area but in many valley bottoms throughout the RMFR comprising the majority of primary range for cattle, feral horses, and wild ungulate.

Using the empirical relationships discovered in this study, indicators such as shrub cover can be readily used to assess forage production and the health of bunchgrasses in the understory. If land manager finds shrub height and density easier to estimate these variables could also be used. Targeting shrub populations, in particular bog birch, for control to maintain canopy cover levels between 12 and 33% will maintain sustainable productive grasslands. Moreover, maintaining a mosaic of grasslands and shrublands in river floodplains of the RMFR will add to the diversity of this landscape and continue to emulate the values many have come to rely on and enjoy.

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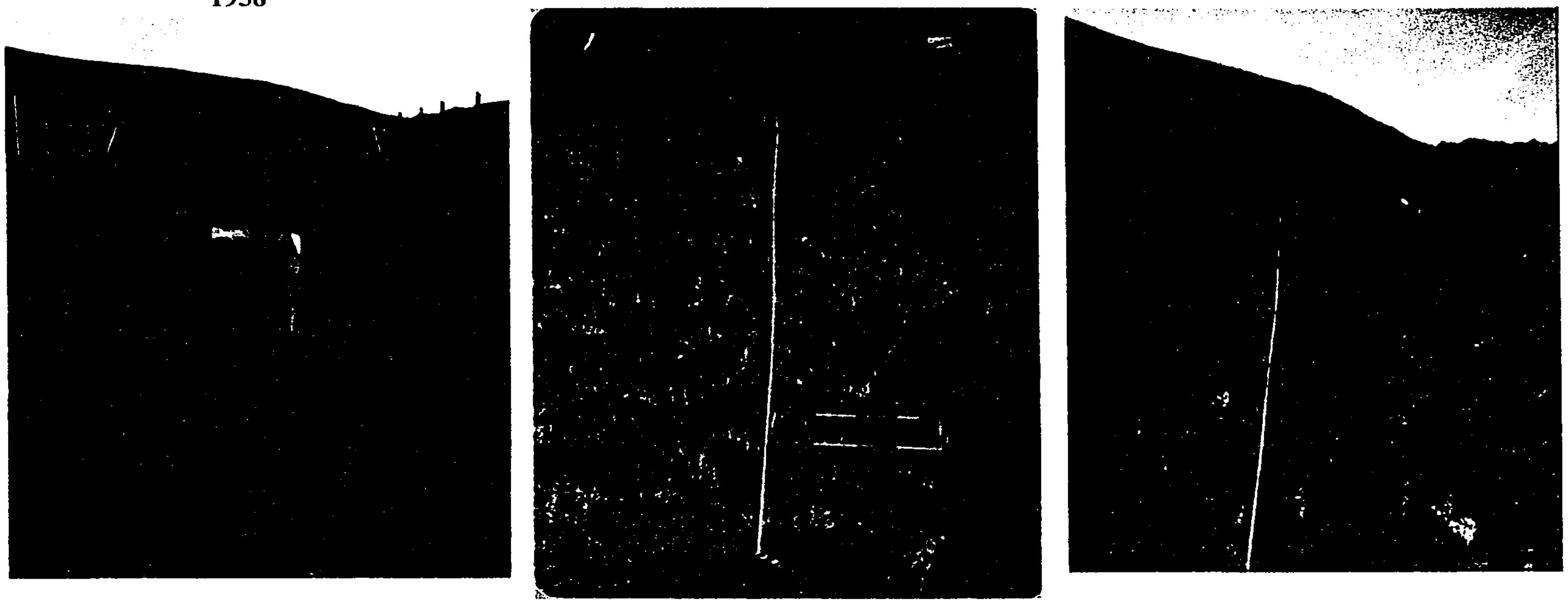
Appendix A: Photo Comparisons

Figure A-1. Comparison of photos taken at the Elk Creek cattle exclosure at 3 different times.

1958 1984 2003



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APPENDIX B: Transect Locations

Table B-1. GPS locations of the 33, 30-m transects in the study area. The first sixteen transects are grassland and the remaining seventeen are shrubland. The bold training transects represent those used in the calibration exercise while the remaining transects were used for the validation procedure.

Grassland Transects			Shrubland Transects		
Transects	**Easting**	**Northing**	**Transects**	**Easting**	**Northing**
RP0102	599522	5766890	RP0202	599452	5766890
RP0302	599503	5767139	RP0402	599472	5767145
RP0502	599002	5767359	RP0602	598777	5767342
RP0802	598458	5767362	RP0702	598513	5767365
RP0902	597009	5767654	**RP10S02**	593797	5767847
RP10G02	593116	5768114	RP1102	596890	5767783
RP1202	596761	5767864	**RP1302**	597065	5767815
RP1402	597020	5767797	RP1602	590531	5768960
RP1502	590611	5768991	**RP1702**	591917	5768603
RP1802	591809	5768680	**RP19S02**	590514	5768812
RP19G02	590525	5768870	RP2202	590268	5771086
RP2102	590296	5771056	RP2402	590361	5771403
RP2302	590181	5771542	RP25S02	589874	5772400
RP25G02	589779	5772382	RP2602	590016	5772451
RP2802	590389	5771589	RP2702	590570	5771759
RP3002	590024	5772784	RP2902	590055	5772726
			RP3102	589924	5772901

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APPENDIX C: History of Livestock Use

Table C-1. Summary of actual range use from 1947 to 1998 of the Clearwater Allotment and the Elk Creek Distribution Unit (DU), the latter of which is one of the main DUs in the allotment. "Season of Use" refers to the grazing period when the Elk Creek DU was utilized by livestock.

Date	Allotment Stocking Rate (AUMs)	Season of Use	Elk Creek[1] Stocking Rate (AUMs)	Season of Use
1947-1952	non use			
1953	78	not recorded	not recorded	not recorded
1954	240	not recorded	not recorded	not recorded
1955	340	not recorded	not recorded	not recorded
1956	1335	not recorded	not recorded	not recorded
1957	1345	not recorded	not recorded	not recorded
1958	2001	not recorded	not recorded	not recorded
1959	1540	not recorded	not recorded	not recorded
1960	566	not recorded	not recorded	not recorded
1961	491	June 14 - Nov.30	207	Aug. 15 - Oct. 12
1962	1296	June 15 - Oct.22	not recorded	not recorded
1963	1289	May 30 - Oct.31	not recorded	not recorded
1964	1700	July 12 - Oct.31	1015	July 12 - 0ct.31
1965	1671	May 29 - Oct.31	713	May 29 - Jun 20/ July 5 - Sept.5
1966	601	June 2 - Oct.5	183	Aug.29 - Oct.5
1967	1395	June 14 - Nov.10	345	Sept. 5 - Oct.10
1968	913	June 13 - Nov.1	424	July 21 - Oct. 5
1969	974	June 18 - Oct. 24	523	June 18 - July 18/ Oct.12 - Oct.24
1970	766	June 18 - Oct. 8	766	June 18 - Oct.8
1971	803	June 11 - Sept. 30	537	July 11 - Sept. 30
1972	909	June 15 - Oct.31	574	July 16 - Sept.30
1973	963	June 16 - Oct.29	657	July 16 - Oct 13
1974	968	June 18 - Oct.29	656	July 16 - Oct. 13
1975	1827	June 17 - Oct.28	355	July 8 - Sept.30
1976	1937	June 16 - Dec.12	613	July 7 - Aug.16/ Aug. 26 - Sept. 30
1977	1799	June 14 – Nov. 4	400	Sept 8 - Oct. 7

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Table C-1. (cont.) Summary of actual range use from 1947 to 1998 of the Clearwater Allotment and the Elk Creek Distribution Unit (DU), the latter of which is one of the main DUs in the allotment. "Season of Use" refers to the grazing period when the Elk Creek DU was utilized by livestock.

1978	831	June 18 – Nov. 2	non use	
1979	1055	June 14 – Nov. 2	413	July 15 - Aug. 4
1980	1351	June 14 – Nov.2	195	Aug. 1 - Aug. 20
1981	1376	June 14 - Oct. 23	230	July 22 - Aug. 15
1982	1401	June 14 - Oct. 30	306	July 17 - Aug 15
1983	1561	June 16 - Oct. 31	386	Aug. 1 - Sept. 5
1984	1628	June 23 - Oct.31	628	July 17 - Sept. 7
1985	1631	June 15 - Oct. 31	366	Aug. 9 - Sept. 9
1986	1631	June 13 - Oct. 31	363	Aug. 6 - Sept. 6
1987	1668	June 14 - Oct. 15	369	July 18 - Sept. 4
1988	1752	June 15 - Oct. 31	690	July 19 - Aug. 20
1989	1858	June 15 - Oct. 31	690	July 19 - Aug. 20
1990	2124	June 15 - Oct. 31	690	July 19 - Aug. 20
1991	1958	June 15 - Oct. 31	690	July 19 - Aug. 20
1992	1934	June 15 - Oct. 31	690	July 19 - Aug. 20
1993	1825	June 15 - Oct. 31	690	July 19 - Aug. 20
1994	1746	June 15 - Oct. 31	690	July 19 - Aug. 20
1995	1750	June 15 - Oct. 31	690	July 19 - Aug. 20
1996	1859	June 15 - Oct. 31	690	July 19 - Aug. 20
1997	1884	June 15 - Oct. 31	690	July 19 - Aug. 20
1998	1888	June 15 - Oct. 31	690	July 19 - Aug. 20

[1] For the following years the stocking rate applies to Elk Creek and another DU:

1970 Main Elk and Main Idlewilde

1971 - 1974 Main and Upper Elk

1979 Main and South Idlewilde and Main Elk

1983 – 1986 Main and Upper Elk

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APPENDIX D: Photo Registration Accuracy

Table D-1. Summary of RMSE values for the 1958 and 1974 imagery. RMSE values are based on comparison to the corrected 1998 imagery.

1974 Photography		1958 Photography	
Location	Distance (m)	Location	Distance (m)
Elk	7	0734_060r	9
Elk	7	0734_060r	10
Elk	7	0734_060r	3
Elk	7	0734_060r	10
Elk	7	0734_060r	8
Elk	7	0734_060r	0
Elk	7	0734_060r	0
Elk	8	0734_060r	5
South Elk	8	0734_169r	10
South Elk	13	0734_169r	16
South Elk	15	0734_169r	5
South Elk	21	0734_169r	5
South Elk	25	0734_169r	11
South Elk	31	0734_169r	5
Between Elk and	5	0734_169r	0
Between Elk and	9	0735_174r	12
Between Elk and	8	0735_174r	5
Between Elk and	4	0735_174r	6
Between Elk and	28	0735_174r	7
Between Elk and	9	0735_174r	9
Between Elk and	10	0735_174r	2
Between Elk and	24	0735_176r	7
Between Elk and	13	0735_176r	0
Radiant	24	0735_176r	10
Radiant	12	0735_176r	14
Radiant	17	0735_176r	26
Radiant	19	0735_176r	31
Radiant	23	0735_176r	16
Radiant	12	0735_176r	19
Radiant	37	0735_178r	22
Radiant	45	0735_178r	15
West Radiant	15	0735_178r	16
West Radiant	20	0735_178r	16
West Radiant	25	0735_178r	15
West Radiant	17	0735_178r	21
West Radiant	29	0735_178r	20
Total RMSE	**15.97**	0735_178r	0
		0735_178r	18
		0735_178r	24
		Total	**10.97**

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Appendix E: Preliminary Analysis to Determine Sample Sizes

Table E-1. Summary of mean species canopy cover associated with sampling either 5, 10, or 15 plots on each of 3 transects.

	Transect 24			Transect 30			Transect 15		
Species	**5 Plots**	**10 Plots**	**15 Plots**	**5 plots**	**10 plots**	**15 plots**	**5 plots**	**10 plots**	**15 plots**
Agropyron trachycaulum (Link)	11.0	13.8	12.9	0.6	3.0	2.9	5.0	7.0	6.2
Carex praegracilis W.Boot	19.0	13.6	12.1	16.0	14.9	12.4	3.0	2.0	1.7
Danthonia californica Boland	18.0	13.3	16.1	-	-	-	-	-	-
Deschampsia cespitosa (L.) Beauv.	0.0	0.5	0.3	50.0	47.0	47.3	25.0	29.5	30.7
Elymus innovatus Beal	-	-	-	1.4	2.4	2.7	-	-	-
Festuca scabrella Torr.	23.0	25.0	22.3	-	-	-	9.0	8.5	10.3
Hierochloe odorata (L.) Beauv.	-	-	-	0.4	0.2	0.3	-	-	-
Juncus balticus Willd.	0.4	0.7	0.7				0.3	0.5	0.0
Luzula parviflora (Ehrh.) Desv.	3.0	0.5	1.0	-	-	-	-	-	-
Poa alpine L.	-	-	-	0.0	0.2	0.1	-	-	-
Poa pratensis L	0.0	0.3	0.2	2.4	2.9	2.1	7.0	10.8	11.2
Achillea millefolium L.	4.6	4.4	3.9	5.0	4.6	5.0	3.4	2.0	2.7
Agoseris glauca (Pursh) Raf.	0.0	0.2	0.3	-	-	-	-	-	-
Arctostaphylos uva-ursi (L.) Spreng.	17.6	10.5	15.9	-	-	-	-	-	-
Aster laevis L.	3.4	0.8	1.9	23.4	15.0	16.0	14.6	13.3	14.2
Cerastium arvense L.	0.8	0.7	0.9	4.0	0.0	0.1	2.4	2.2	2.2
Crepis runcinata (James) T.& G.	4.0	1.8	1.9	-	-	-	-	-	-
Fragaria virginiana Duchesne	7.4	5.8	6.2	-	-	-	15.0	14.8	12.9
Galium boreale L.	0.8	0.6	0.9	-	-	-	1.6	0.5	0.7
Gentiana prostrate Haenke	0.6	0.6	0.6	-	-	-	-	-	-
Geum triflorum Pursh	1.0	2.5	3.0	0.0	1.0	0.7	3.4	3.0	3.5
Hedysarum alpinum L.	0.6	0.8	0.5	-	-	-	-	-	-
Lathyrus ochroleucus Hook.	1.0	1.1	0.7	-	-	-	-	-	-

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Table E-1. (cont.) Summary of mean species canopy cover associated with sampling either 5, 10, or 15 plots on each of 3 transects.

	Transect 24			Transect 30			Transect 15		
Species	**5 Plots**	**10 Plots**	**15 Plots**	**5 plots**	**10 plots**	**15 plots**	**5 plots**	**10 plots**	**15 plots**
Lathyrus ochroleucus Hook.	1.0	1.1	0.7	-	-	-	-	-	-
Linnaea borealis L.	0.4	0.2	0.1	-	-	-	-	-	-
Penstemon procerus Dougl. *ex* Grah.	2.0	1.1	1.4	0.6	0.7	0.7	-	-	-
Polygonum viviparum L.	1.8	1.0	1.1	1.4	0.7	1.0	-	-	-
Potentilla gracilis Dougl. *ex* Hook.	0.6	1.3	0.9	15.8	19.9	20.1	5.9	7.5	4.6
Rubus arcticus L.	-	-	-	-	-	-	0.6	1.2	1.5
Rumex acetosa L.	-	-	-	3.0	1.8	1.9	2.4	2.3	2.0
Smilacina stellata (L.) Desf.	-	-	-	-	-	-	0.6	0.8	0.7
Solidago spathulata DC.	1.0	0.5	0.6	-	-	-	2.0	2.0	1.9
Stellaria longipes Goldie	0.8	0.2	0.3	0.6	0.2	0.5	1.8	2.5	2.6
Taraxacum officinale Weber	-	-	-	0.0	1.2	1.1	28.0	16.5	17.0
Thalictrum occidentale A. Gray	0.0	0.2	0.1	-	-	-	3.4	3.4	2.5
Vaccinium cespitosum Michx.	20.0	12.8	14.2	-	-	-	-	-	-
Valeriana septentrionalis Rydb.	-	-	-	1.2	0.6	0.4	1.0	1.5	1.3
Vicia americana Muhl.	-	-	-	0.0	1.0	0.7	1.0	1.1	0.7
Viola adunca J.E. Smith	1.4	1.5	1.5	0.0	0.7	0.5	7.6	4.8	3.8
Zizia aptera (A. Gray) Fern.	1.0	0.2	0.5	-	-	-	-	-	-
Betula glandulosa Michx.	29.0	32.0	30.0	-	-	-	18.4	14.4	13.6
Potentilla fruticosa L.	0.6	0.8	0.5	0.0	0.4	0.3	0.0	0.7	0.7
Salix myrtillifolia Anderss.				-	-	-	0.6	1.4	1.40.5
Moss & Lichen	35.0	29.8	26.2	0.0	1.5	1.0	0.0	0.3	0.5

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Table E-3. Summary of plant density measurements (No./m^2) along 3, 10-m intervals of the 30-m line transect for each of 2 transects.

		Transect Length		
Component	**Transect**	**0 – 10 m**	**0 – 20 m**	**0 – 30 m**
Bunchgrasses	17	1.3	0.95	1.32
	16	1.75	3.03	2.53
Bog Birch	17	1.2	1.23	1
	16	1.11	0.56	0.62
Salix	17	0.75	0.43	0.32
	16	0.45	0.35	0.45

Figure E-1. Summary of shrub canopy cover measurements (%) along different distances within each of the four transects.

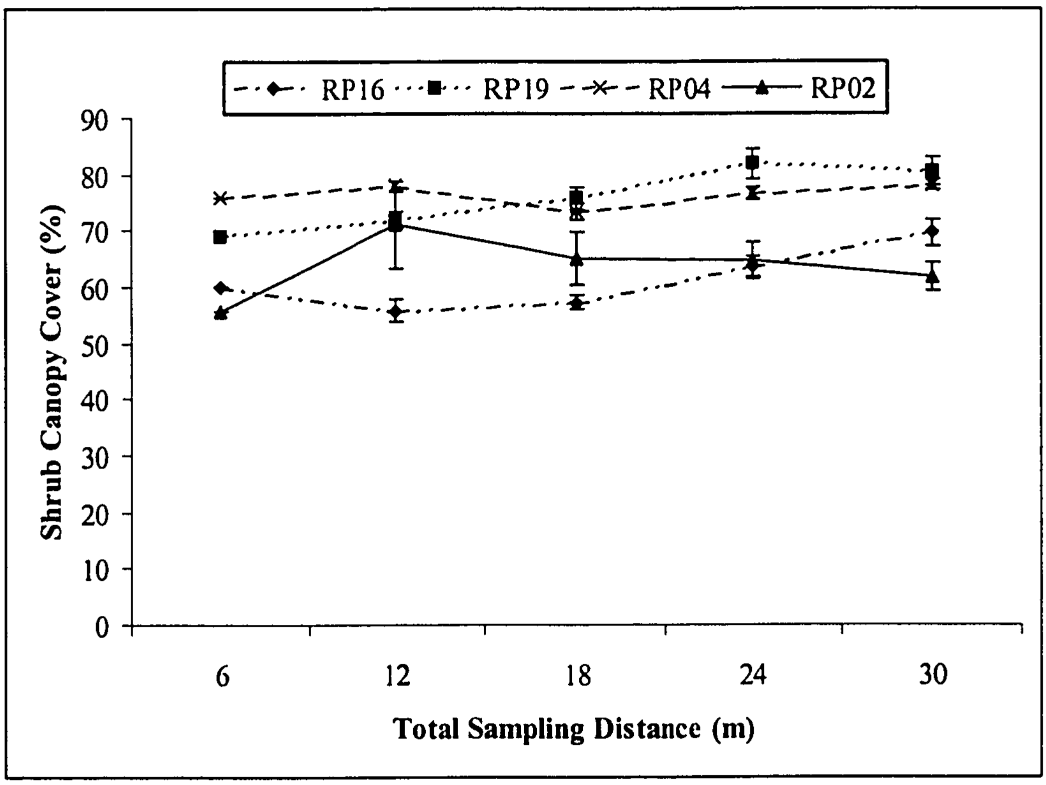

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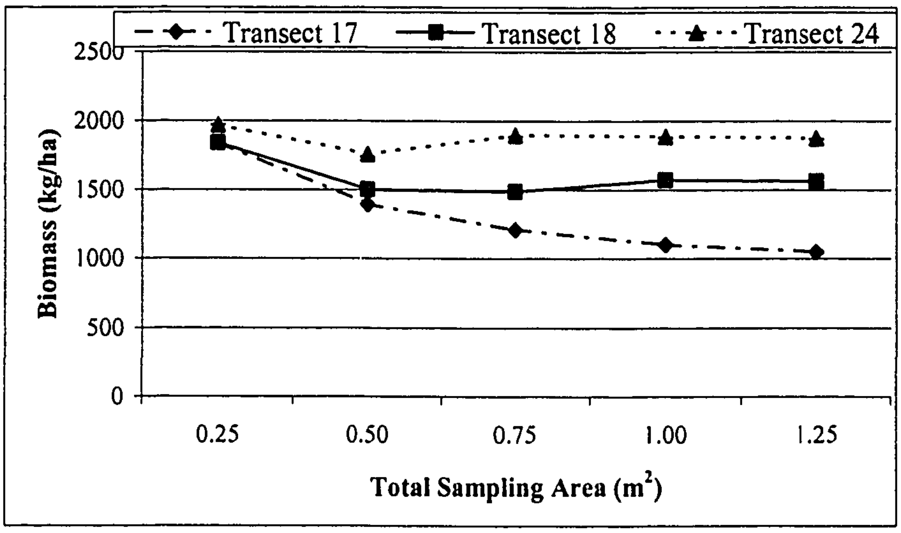
Figure E-2. Running averages of total herbage (grass and forb) production clips (n=5) for Transects 17, 18 and 24.



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Appendix F: Description of Soil Profiles

Table F-1. Summary of soil descriptions collected from soil pits dug adjacent to each transect throughout the study area.

Transects	Horizon	Depth (cm)	Description
RP01	Ah	0 -25	Dark brown (7.5 YR 3/2, moist) clay loam; fine, granular; abundant fine, random inped roots to 30 cm; wavy gradual boundary;25 cm thick.
	Ahe	25 - 28/30	Ashy orange colour;platy;clear wavy boundary; 3-5 cm thick
	Bm	30 +	brown (10 YR 5/3 dry) loamy clay; angular blocky;few fine inped roots to 80cm; gradual wavy boundary;
RP02	LFH	4/3 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 3-4 cm thick.
	Ah	0 -6/8	Dark brown (7.5 YR 3/2, moist) loamy; fine, granular; abundant fine, random inped roots; wavy gradual boundary;6-8cm thick.
	Bm1gj	6/8 - 44/46	Brown (10 YR 5/3 dry) loamy; medium subangular blocky;few,medium,prominent (orange) mottles;many fine inped roots to 27cm; gradual wavy boundary;36-40cm thick
	Bm2	44/46 - 64+	Brown (10 YR 5/3 dry) clay loam; medium to coarse subangular blocky;
RP03	Ah	0 -14	Dark brown (7.5 YR 3/2, moist) clay loam; fine, granular; abundant fine, random inped roots; wavy gradual boundary;14 cm thick.
	Bm	14 - 34	Dark brown (10 YR 4/3, moist) clay loam;angular blocky; many fine random inped roots to 20 cm; gradual wavy boundary;22 cm thick
	II Ae	34 - 34/44	Ashy orange colour; platy; abrupt broken boundary; 0 - 10 cm thick
	Bmg	34/44+	Grey black clay loam; angular blocky; blue common, medium prominent mottles at 56 cm. Dark black splotches appear to be like charcoal at 45 cm
RP04	LFH	3 -0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 3 cm thick.
	Ah	0 - 4/12	Dark brown (7.5 YR 3/2, moist) loamy; fine, granular; abundant fine, random inped roots; wavy gradual boundary;4-12 cm thick.
	Bmgj	4/12 - 66+	Brown (10 YR 5/3 dry) clay loam; medium subangular blocky;common,medium,prominent (orange) mottles;many fine inped roots to 45cm, most common to 15 cm and many vertical exped roots to 50 cm; gradual wavy boundary;54-62cm thick

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RP05	LFH	6 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass; 6 cm thick.
	Bm	0 - 22	Dark brown (10 YR 4/3, moist) silty loam;dark black splotches may be charcoal;fine subangular blocky;abundant, fine, random, inped roots; gradual wavy boundary;22 cm thick.
	II Ah	22 - 37	Dark brown (7.5 YR 3/2, moist) silty loam;moderate, fine, granular;soft; abundant fine, random inped roots; wavy gradual boundary;15 cm thick.
	III Ae	37 - 40	Ashy white sandy; platy; clear smooth boundary; 3 cm thick
	Bmtj	40 - +	Dark brown (10 YR 4/3, moist) clay loam;subangular blocky; common medium prominent mottles;few, fine, random, inped roots
RP06	LFH	4/3 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 3-4 cm thick.
	Ah	0 - 3/5	Dark brown (7.5 YR 3/2, moist) loamy; fine, granular; abundant fine, random inped roots; wavy gradual boundary;3-5 cm thick.
	Bm1gj	3/5 - 54	Brown (10 YR 5/3 dry) loamy; medium subangular blocky;common,medium,prominent (orange) mottles;many fine inped roots to 48cm, most common to 40 cm and few oblique vertical exped roots to 57 cm; gradual wavy boundary;49-51cm thick
	Bm2	54 - 75+	Brown (10 YR 5/3 dry) clay loam; medium subangular blocky
RP07	LFH	7/2 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 2-7 cm thick.
	Ah	0 - 2/4	Dark brown (7.5 YR 3/2, moist) loamy; fine, granular; abundant fine, random inped roots; wavy gradual boundary;2-4 cm thick.
	Bm1gj	2/4 - 44/48	Gray brown loam; medium to coarse subangular blocky;few, medium, prominent mottles (orange) near top of horizon;abundant, fine,random,inped roots to 58 cm most common to 42 cm;gradual smooth boundary;46-40 cm thick
	Bm2	44/48 - 80+	Gray to dark gray clay loam; medium to course subangular blocky; few oblique vertical (shrub) exped roots to 56 cm.
RP08	LFH	12 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass; 12 cm thick.
	Bmgj	12 - 70	Dark brown (10 YR 4/3, moist) clay loam;fine subangular blocky; common medium prominent mottles;abundant, fine, random, inped roots above 40 cm; gradual wavy boundary;58 cm thick.
	Ae	70 - 75/80	Orange sandy;platy;clear smooth boundary; 5-10 cm thick

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	Bmtjgj	75/80+	Gray black clay; clay skins evident; course angular blocky to medium granular;blue/gray medium, common, distinct mottles
RP09	Ah	0 - 8/14	Dark brown (7.5 YR 3/2, moist) silty loam;moderate, fine, granular;soft; abundant fine, random inped roots; wavy gradual boundary;8 to 14 cm thick.
	Bm	10 - 50	Dark brown (10 YR 4/3, moist) silty loam;medium subangular blocky; few, fine to medium, random, inped roots; 40 cm thick.
	II Ae	50 - 54	Ashy orange platy; very few fine to medium inped roots; abrupt broken boundary;2-6 cm thick
	Bmtgjtj	54 - 100+	Clay;mdium prismlike to fine prismlike;few medium distint blue-gray mottles;very few fine inped roots as far down as 85 cm most in upper 83
RP10S	LFH	10 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 10 cm thick.
	Bm	0 - 10/15	Brown (10 YR 5/3 dry) sandy loam; medium subangular blocky;many fine inped roots to 20cm and few vertical oblique exped roots to 30 cm;gradual wavy boundary; 10 - 15 cm thick
	C	20/25 +	Gray black clay loam;amorphous
RP10G	Ah	0-7	Dark brown (7.5 YR 3/2, moist) loam;moderate, fine, granular;soft; abundant fine, random inped roots; wavy gradual boundary;7 to 10 cm thick.
	Ahe	7 - 7/17	Orange tinged coloured; fine platy; fine, random inped roots; discontinous boundary;0 to 10 cm thick.
	Bm	7/17 - 60+	Dark brown (10 YR 4/3, moist) loamy clay;angular blocky to sub-angular blocky; fine, random inped roots; strong effervescence at 75 cm;55 to 60 cm thick.
RP11	LFH	6 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 4-7cm thick.
	Ah	0-4/5	Dark brown (7.5 YR 3/2, moist) loam; fine granular;gradual smooth boundary; abundant fine inped roots;4-5 cm thick.
	Bm1	4/5-44/52	Dark brown (10 YR 4/3, moist) loam;medium angular blocky; common,medium,faint mottles;many fine inped roots to 80 cm and few oblique vertical exped roots to 30 cm; gradual smooth boundary;41-48 cm thick
	Bm2	44/52-78+	Dark brown (10 YR 4/3, moist) and brown (10 YR 5/3 dry) clayloam; medium angular blocky;abundant fine inped roots to 46 cm most common to 36 cm and few vertical oblique exped roots to 34 cm

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RP12	Ah	0 - 5/9	Dark brown (7.5 YR 3/2, moist) silty loam; fine granular;gradual smooth boundary; abundant fine inped roots;5-9 cm thick.
	Bmgj	9 - 65	Dark brown (10 YR 4/3, moist) silty loam;angular blocky; large, light red sandy patches in upper 50 cm; dark black patches that look like Ah in upper 50 as well;many fine inped roots to 80 cm and few oblique vertical exped roots to 30 cm; gradual smooth boundary;50 cm thick
	Cgj	65+	Dark gray clay loam; amorphous
RP13	LFH	3-0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 3cm thick.
	Ah	0-4/5	Dark brown (7.5 YR 3/2, moist) silty loam; fine granular;gradual smooth boundary; abundant fine inped roots;4-5 cm thick.
	Bm1gj	4/5-44/48	Brown (10 YR 5/3 dry) loamy; medium subangular blocky;faint,medium,prominent (orange) mottles;many fine inped roots to 48cm, most common to 30 cm and few oblique vertical exped roots to 36 cm; gradual wavy boundary;41-43cm thick.
	Bm2	44/48-64+	Dark brown (10 YR 4/3, moist) clayloam; medium to coarse subangular blocky; 20+ cm thick.
RP14	Ah	0-8/9	Dark brown (7.5 YR 3/2, moist) silty loam; fine granular;gradual smooth boundary; abundant fine inped roots;8-9 cm thick.
	Bm	8/9 - 48/49	Dark brown (10 YR 4/3, moist) silty loam;fine to medium subangular blocky; many fine inped roots; gradual smooth boundary;40 cm thick
	II Ae	50 - 53	Orange ashy colour platey; abrupt,discontinous boundary; 2-3 cm thick
	Bm	53+	Dark brown (10 YR 4/3, moist) clay loam;fine to medium subangular blocky;few fibrous roots
RP15	Ah	0 - 2/5	Dark brown (7.5 YR 3/2, moist) loam; fine granular;gradual wavy boundary; 2-5 cm thick.
	Bm1gj	2/5 - 34/43	Brown (10 YR 5/3, dry) loam; medium subangular blocky;many medium distinct mottles (red); gradual wavy boundary;29 to 41 cm thick
	Bm2gj	34/43 - 70+	Gray brown clay loam; few medium faint mottles; many fine fibrous roots to 55 cm, most common to 28 and few courser fibrous roots.
RP16	LFH	5/2 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 2-5cm thick.

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	Bml	0 - 34/39	Dark brown (10 YR 4/3, moist) loam;fine to medium subangular blocky;few faint mottles; many fine inped roots to 55 cm and many oblique exped roots to 35 cm; wavy gradual boundary;34 - 39 cm thick
	Bm2	34/39 - 55+	Brown (10 YR 5/3, dry) sandy loam; medium to coarse subangular blocky;few medium distinct(orange) mottles; gravelly;gradual wavy boundary;56 to 62 cm thick
RP17	Ah	0-7	Dark brown (7.5 YR 3/2, moist); silty loam; fine, granular;random inped roots; 7 – 14 cm thick.
	Ahe	7 -10	Ashy orange; Silty-loam; fine, granular; random inped roots; 0 – 2 cm thick.
	Aej	10 - 12	Ashy white; silty-loam; medium granular; random inped roots; abrupt, irregular boundary; 0 – 2 cm thick.
	Bfj	13-20	Orange brown; clay-loam; medium sub-angular to fine blocky; random, inped roots; gravelly; 15 –20 cm thick (Charcoal prominent at 16 cm)
	Bm	21-49	Brown (10 YR 5/3, dry); clay-loam;medium sub-angular blocky;random, inped roots; 15 to 20 cm
	Cca	50 +	Gray; clay loam; amorphous; oblique, exped roots;
RP18	Ah	0-10/16	Dark brown (7.5 YR 3/2, moist) loam; fine granular;few, medium, distinct orange and light brown mottles;gradual wavy boundary; 10-16 cm thick. Slight platy structure at bottom of horizon
	Bm	10/16-29/32	Brown (10 YR 5/3, dry) loam; medium granular;gravelly; gradual wavy boundary;19 to 22 cm thick
	Bfj	29/32-50/52	Dark yellowish brown (10YR 4/4, moist) and light yellowish brown (10 YR 6/4 dry) sandy loam; moderate fine subangular blocky; plentiful, fine, fibrous inped roots to 56cm, most common to 35 cm; gradual wavy boundary; 18-23 cm thick.
	C	50/52-67+	Gray;amphorous
RP19G	Ah	0 - 4/7	Dark brown (7.5 YR 3/2, moist) loam; fine granular;gradual wavy boundary; 4-7 cm thick.
	Bmgj	4/7 - 80+	Brown (10 YR 5/3 dry) clay loam; medium to coarse subangular blocky;common,medium, distinct light brown to orange mottles; many fine fibrous inped roots to 55 cm most common to 30 cm and few coarser inped roots; weak effervescent
RP19S	LFH	7/4 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 4-7cm thick.
	Bml	0 - 50/60	Dark brown (10 YR 4/3, moist) loam; medium subangular blocky;many fine inped roots to 58 cm(most common at 38 cm and few oblique exped roots to 18 cm; 50 -60 cm thick

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	Bm2gj	50/60 - 74+	Brown (10 YR 5/3, dry)clay loam; medium to coarse subangular blocky;few medium distinct mottles
RP21	Ah	0 - 2	Brown (10 YR 4/4, dry) loam;weak to moderate fine granular; gradual wavy boundary;2 cm thick
	Bm1gj	2 - 58/64	Brown (10 YR 5/3, dry) loam; medium to coarse subangular blocky;few medium faint mottles; gradual wavy boundary;56 to 62 cm thick
	Bm2	58/64 - 77+	Dark brown (10 YR 4/3, moist) loam; medium to coarse subangular blocky;many fine inped roots to 53 cm and few oblique exped roots to 5 cm
RP22	LFH	3/2 - 0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots;2-3cm thick.
	Bmgj	0 - 43/46	Brown (10 YR 5/3, dry) loam; medium to coarse subangular blocky;few medium faint mottles; gradual wavy boundary;40-44 cm thick
	Bmkgj	43/46 - 80+	Brown (10 YR 5/3, dry) loam; very coarse subangular blocky;few medium distinct mottles;strong effervescence at 45 cm;many;fine fibrous inped roots to 60 cm and many, oblique exped roots to 8 cm; gradual wavy boundary
RP23	Ah	0-3/10	Dark brown (7.5 YR 3/2, moist) loam; fine granular;few, fine, distinct orange and light brown mottles;gradual wavy boundary; 3-10 cm thick.
	Bm1gj	3/10-40/57	Brown (10 YR 5/3 dry) loam; fine to medium subangular blocky;common, medium, distinct mottles; 37-47 cm thick
	Bm2	40/57-75+	Brown to dary gray clay loam; medium subangular blocky containing 30% rounded gravel sized fragments
RP24	LFH	7/12-0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots;7-12 cm thick.
	Ahe	0-7	Loamy;fine granular; many fine inped roots;7 cm thick.
	Bmfj	0-30	Orangey, Clayey; course to very course subangular blocky23 cm thick.
	C	30+	Clayey;amphorphous.
RP25G	Ah	0 - 6/10	Dark brown (7.5 YR 3/2, moist) loam; weak fine granular;friable, slightly sticky, slightly plastic; gradual, wavy boundary;many, fine inped roots
	Bm	6/10 - 37/43	Brown with ochre hues loamy;medium to course subangular blocky; fine fibrous roots common to 22 cm and course fibrous roots inpead and exped to 64 cm but most common to 36 cm.

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	Bmgj	37/43 - 80+	Dark gray clay loam; medium to coarse angular blocky; common, medium prominent mottles;
RP25S	LFH	4-0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 3-4cm thick.
	Ah	0-3	Brown (10 YR 4/4, dry) loam;weak to moerate fine granular;friable, slightly sticky, slightly plastic; gradual wavy boundary;3 cm thick
	Bmlgj	7 – 43/52	Dark yellowish brown (10YR 4/4, moist) and light yellowish brown (10 YR 6/4 dry) loam; moderately to strong course platy and moderate fine subangular blocky;distinct strong brown mottles;firm, sticky, plastic; plentiful, fine, oblique inped roots; gradual wavy boundary;34-45 cm thick
	Btj	43/52-75+	Gray to dark gray clay loam; course to very course subangular blocky with definite clay skins. Fine fibrous roots inped to 47 cm most common to 17 cm. Few shrub roots exped to 23 cm.
RP26	LFH	3/4-0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 3-4cm thick.
	Bm	3/4-19/26	Dark yellowish brown (10YR 4/4, moist) and light yellowish brown (10 YR 6/4 dry) loam; moderately to strong course platy and moderate fine subangular blocky;firm, sticky, plastic; plentiful, fine, oblique inped roots; gradual wavy boundary; medium acid; 16-22 cm thick
	Ck	19/26 - 65+	Brown (10YR 5/3, moist) clay; containing 10% rounded gravel sized fragments; massive; firm, very sticky,plastic; very few fine vertical roots; strongly effervescent
RP27	LFH	6-0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; gradual irregular boundary
	Ah	0-2/4	Dark brown (7.5 YR 3/2, moist) loam; weak to moerate fine granular;friable, slightly sticky, slightly plastic; discontinuous boundary.
	Bmgj	2/4 –45	Gray; clay loam; coarse angular blocky;many,few, medium, distinct mottles.
	Bgj	45 – 67+	Clay;course angular blocky
RP28	Ah	0-6/10	Dark brown (7.5 YR 3/2, moist); silty-loam; fine subangular blocky; random, inped roots; 6-10 cm
	Bmlgj	6/10-36/50	Brown (10 YR 5/3 dry); clay-loam, fine to medium subangular blocky; many, medium, prominent mottles; 30-40 cm thick.

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	Bm2	36/50 –80	Brown (10 YR 5/3 dry); clay-loam; medium subangular blocky; 30 –44 cm thick.
	C	>80 cm	Gray; clayey, amphorphous
RP29	Ah	0-3/4	Dark brown (7.5 YR 3/2, moist) loam; weak to moderate fine granular;friable, wavy gradual boundary;3-4 cm thick.
	Bm	3/4 - 47/55	Dark brown (10 YR 4/3, moist) loam; medium to coarse subangular blocky;distinct orange/red mottles firm, many fine inped and few oblique exped roots; gradual wavy boundary; 44-52 cm thick
	Ck	47/55 - 75+	Brown (10YR 5/3, moist) clay loam; strongly effervescent at 55 cm.
RP30	Ah	0 - 2/4	Dark brown (7.5 YR 3/2, moist); fine granular;fine, many, inped roots.
	Bm	2/4 - 38/44	Dark brown (10 YR 4/3, moist); fine to medium blocky subangular; few, medium, distinct mottles
	Bfj	38/44 - 57/68	Mostly reddish soil; fine to medium subangular blocky
	C	57/68+	Gray; clay loam; amorphous; coarse to very coarse blocky subangular blocky.
RP31	LFH	3/5-0	Very dark brown (10YR 2/2 moist) organic mat consisting of slightly to moderately decomposed grass, leaves and roots; 3-5cm thick.
	Ah	0-2	Dark brown (7.5 YR 3/2, moist) and brown (10 YR 4/4, dry) loam; weak to moerate fine granular;friable, slightly sticky, slightly plastic; discontinuous boundary.
	Bm1	2 - 18	Dark yellowish brown (10YR 4/4, moist) and light yellowish brown (10 YR 6/4 dry) clay loam; moderately to strong course platy and moderate fine subangular blocky;firm, sticky, plastic; plentiful, fine, oblique inped roots; gradual wavy boundary; medium acid; 10-21 cm thick
	Bm2	18-36	Brown to dark brown (10 YR 4/3, moist) and brown (10 YR 5/3 dry) clay loam with pockets of course sand, moderate medium granular; firm, sticky, plastic; few fine oblique inped roots; gradual smooth boundary; strong effervescent; 37 cm thick
	Ck	36-75+	Brown (10YR 5/3, moist) clay; containing 10% rounded gravel sized fragments; massive; firm, very sticky,plastic; very few fine vertical roots; strongly effervescent

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Appendix G. Analysis Summary

Table G-1. Summary of empirical relationships between understory variables and various overstory (shrub) characteristics.

	Total Mean Shrub			Bog Birch			Willow			Shrubby Cinquefoil		
Response Variable	**Density**	**Cover**	**Height**	**Density**	**Cover**	**Height**	**Density**	**Cover**	**Height**	**Density**	**Cover**	**Height**
Species Richness	r^2=0.02,NS	r^2=0.07,NS	r^2=0.15*	r^2=0.01,NS	r^2=0.03,NS	r^2=0.18*	r^2=0.NS	r^2=0.06,NS	r^2=0.11,NS	r^2=0.00,NS	r^2=0.07,NS	r^2=0.00,NS
Species Diversity	r^2=0.22*	r^2=0.28*	r^2=0.23*	r^2=0.16,NS	r^2=0.12*	r^2=0.21*	r^2=0.19*	r^2=0.17,NS	r^2=0.21*	r^2=0.00,NS	r^2=0.03,NS	r^2=0.00,NS
Bunchgrass Density	r^2=0.37**	r^2=0.47***	r^2=0.40***	r^2=0.50***	r^2=0.43***	r^2=0.40***	r^2=0.21*	r^2=0.16,NS	r^2=0.26*	r^2=0.03,NS	r^2=0.01,NS	r^2=0.03,NS
Bunchgrass Cover	r^2=0.50***	r^2=0.71***	r^2=0.52***	r^2=0.64***	r^2=0.58***	r^2=0.53***	r^2=0.38**	r^2=0.32**	r^2=0.39**	r^2=0.05,NS	r^2=0.03,NS	r^2=0.00,NS
Forb Cover	r^2=0.,NS	r^2=0.00,NS	r^2=0.00,NS	r^2=0.03,NS	r^2=0.05,NS	r^2=0.00,NS	r^2=0.01,NS	r^2=0.13,NS	r^2=0.08,NS	r^2=0.01,NS	r^2=0.00,NS	r^2=0.02,NS
Total Cover	r^2=0.10,NS	r^2=0.11,NS	r^2=0.04,NS	r^2=0.19*	r^2=0.23*	r^2=0.06,NS	r^2=0.00,NS	r^2=0.00,NS	r^2=0.00,NS	r^2=0.01,NS	r^2=0.00,NS	r^2=0.01,NS
Grass Production	r^2=0.62*	r^2=0.82***	r^2=0.46**	r^2=0.61***	r^2=0.59***	r^2=0.36**	r^2=0.42***	r^2=0.30**	r^2=0.49***	r^2=0.02,NS	r^2=0.01,NS	r^2=0.04,NS
Forb Production	r^2=0.26*	r^2=0.38**	r^2=0.29*	r^2=0.43***	r^2=0.45***	r^2=0.31***	r^2=0.12,NS	r^2=0.62,NS	r^2=0.16*	r^2=0.02,NS	r^2=0.02NS	r^2=0.07,NS
Herbage Production	r^2=0.52***	r^2=0.79***	r^2=0.42***	r^2=0.65***	r^2=0.64***	r^2=0.40***	r^2=0.36**	r^2=0.24*	r^2=0.44***	r^2=0.02,NS	r^2=0.02,NS	r^2=0.06,NS
Moss Cover	r^2=0.21*	r^2=0.47***	r^2=0.21*	r^2=0.62***	r^2=0.69***	r^2=0.23*	r^2=0.12,NS	r^2=0.03,NS	r^2=0.08,NS	r^2=0.05,NS	r^2=0.04,NS	r^2=0.11,NS
LFH Thickness	r^2=0.40***	r^2=0.45***	r^2=0.21*	r^2=0.21*	r^2= 0.44***	r^2=0.25*	r^2=022*	r^2=0.16,NS	r^2=0.20*	r^2=0.01,NS	r^2=0.00,NS	r^2=0.01,NS
Ah Thickness	r^2=0.29**	r^2=0.31**	r^2=0.35**	r^2=0.33**	r^2=0.34**	r^2=0.39**	r^2=0.02,NS	r^2=0.02,NS	r^2=0.06,NS	r^2=0.01,NS	r^2=0.00,NS	r^2=0.02,NS

* Denotes significance at P<0.01

** Denotes significance at P<0.001

*** Denotes significance at P<0.0001