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Long-term agronomic and environmental impact of aspen control strategies
in the Aspen Parkland

by

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I dedicate this work to Hannah Raven and Khloé Rose

“You can do anything, if you set your mind to it.”

I love you my little princesses!

Abstract

Since European settlement the Aspen Parkland has been subject to agricultural intensification. This research assessed the agronomic, ecologic and economic impact of native Parkland conversion into tame pasture, by building on a study initiated in 1980 investigating the short-term agronomic responses within 3 landscape-level treatments: an intensive Clear & Break (C&B), a Spray & Burn (S&B) and a burnt Native Check (NC). Historical information was supplemented with recently collected data (2005-2006). Production remained high within the NC relative to the others after 25 years, in part due to contributions from browse in areas with increasing woody species. Plant species composition also demonstrated considerable convergence (i.e. overlap) between native and tame grasslands, and although not different in soil organic matter, microfaunal activity differed marginally. Net present value (NPV) economic analysis indicated the NC and S&B provided greater aggregate returns over the study period, and has implications for aspen management in the future.

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"At times our own light goes out and is rekindled by a spark from another person. Each of us has cause to think with deep gratitude of those who have lighted the flame within us."
Albert Schweitzer 1875-1965

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1. INTRODUCTION

1.1 Woody Species Encroachment

Native woody species contribute to the landscape diversity of rangeland ecosystems. However, over the past century the expansion of woody species into grasslands has become a widespread phenomenon in both cool-season (Bailey and Wroe 1974, Scheffler 1976, Burkinshaw and Bork 2009) and warm-season grasslands (Brown and Archer 1989, Havstad and Schlesinger 1996, Van Auken 2000, Briggs et al. 2005). Significant increases in woody cover combined with decreases in grass production threaten the economic viability of the ranching industry in North America (Osborn and Witkowski 1974, Schumann et al. 2001, Teague et al. 2001).

While the mechanisms of woody species expansion are not fully understood, a number of key factors have been implicated as a catalyst of change including changes in regional and global climate (Archer et al. 1995, Polley 1997, Hibbard et al. 2003, Mangan et al. 2004, Knapp et al. 2008, Hogg et al. 2008), disturbance regimes such as fire and grazing (Campbell et al. 1994, Van Auken 2000, Mangan et al. 2004, Briggs et al. 2005), biochemical dynamics (Hibbard et al. 2003, Briggs et al. 2005) and changes in soil structure or moisture (Brown and Archer 1989, Bragg et al. 1993, Polley 1997).

Although trees and shrubs add to the structural complexity of the landscape and provide habitat for a diversity of wildlife species (Hansen et al. 1991), the expansion of woody species into open grasslands can reduce herbaceous productivity, and hence the economic potential of the ranching industry (Jurena and Archer 2003). Encroachment into open grasslands can reduce decomposition rates (Scholes and Archer 1997, Throop and Archer 2007) and associated soil organic matter (Johnston et al. 1999), herbaceous biomass (Jurena and Archer 2003) and plant species diversity (Henkin et al. 2007). Finally, the distribution pattern and abundance of brush across rangelands can have a significant influence on livestock distribution and may lead to uneven utilization of the forage resource (Owens et al. 1991, Asamoah et al. 2003), ultimately reducing carrying capacity (Jones 1983).

1.2 Managing Woody Species on Rangeland

Rangeland management is an art and science dedicated to the enhancement of ecological processes using the best information available, while ensuring the provision of multiple uses (SRM 2008). Ultimately, range management strives for the sustainable use of biotic and abiotic resources and to maximize the synergistic capacity of all components to contribute to a healthy, fully functional ecosystem. Research and monitoring are key tools required to increase the knowledge base of scientists and rangeland managers alike, and thereby develop more effective rangeland management strategies.

Today, when ranchers are faced with economic hardship (i.e. declining livestock prices and increasing costs) and a shrinking agricultural land base, the potential impacts of woody species invasion in grassland ecosystems is of global significance (Osborn and Witkowski 1974, Schumann et al. 2001, Teague et al.

2001). Decreased grazing capacity for livestock coupled with increased annual production costs (Stuth et al. 1991) has become an issue for producers using both private and public lands. Public land managers are further challenged with the integration of livestock and wildlife management and a shrinking grazing resource heightens the potential for land use conflict (Burkinshaw and Bork 2009). However a number of studies suggest that brush management strategies, including mechanical clearing (Lezberg et al. 2006), fire (Howe 2000), grazing or combinations thereof, will increase grass production and restore accessibility to grazing animals (Anderson and Bailey 1980, Fitzgerald and Bailey 1984, Ben-Shahar 1992, Bork et al. 1997, Lett and Knapp 2005).

A number of brush management strategies are available to mitigate the negative impacts of woody species encroachment on forage resources. Contemporary brush management may enhance the ability of ecosystems to meet social demands while maintaining resiliency to significant change in ecosystem structure and function (Herbel 1983, Folke et al. 2002). A reduction in woody cover and the promotion of open grasslands increases forage availability and accessibility, while simultaneously restoring habitat for grassland obligate wildlife (Anderson and Bailey 1980, Clary and Jameson 1981, Burgess 1988, Ben-Shahar 1992, Roques et al. 2001, Olenick et al. 2004, Ansley et al. 2006).

Of particular importance in the process of woody species management is the principal philosophy of those responsible for the management of range resources. There are two primary but contrasting approaches to conducting rangeland management (Herbel 1983), and they include intensive and extensive management. The approach chosen may be most paramount on deeded (i.e. private) land where a single stakeholder controls management activities and subsequent impacts to the landscape.

Intensive management strategies are designed to rapidly increase rangeland productivity through the investment of significant capital (i.e. money), time, labor, and natural resources (Herbel 1983), and are often associated with yield maximization within the natural limits of the ecosystem. In contrast, extensive management strategies focus on managing holistically at the landscape level, ultimately balancing optimal financial and labor inputs with potential returns (Savory and Butterfield 1988). Extensive management strives to generate favorable net returns through the attainment of moderate yields at lower cost. As there are a number of range management practices available to control woody plants on a continuum from extensive to intensive strategies, the selection of brush control strategies has the potential to greatly influence associated ecosystem characteristics (i.e. biodiversity), levels of production (i.e. forage quantity and quality), as well as net profitability.

All levels along the continuum of brush management practices have met with some success in the past. Extensive management practices found effective for reducing woody cover include managed grazing systems and prescribed fire, used independently or in combination (Bailey et al. 1990, Holechek and Galt 2004, Bernués et al. 2005). Intensive management strategies include mechanical dozing, mowing and chaining to clear unwanted vegetation, and comprehensive preparation of sites for revegetation to agronomic species, leading to the advantage of substantial forage increases in the short-term (Bailey et al. 1987). Intermediate forms of brush control may include herbicide application, which have also been proven effective at reducing woody species (Hilton and Bailey 1974, Bowes 1996, Heaton et al. 2003). Ultimately, an integrated brush management approach that combines both extensive and intensive practices may optimize the use of available resources, and be the most effective and sustainable management strategy (Evans and Workman 1994).

1.3 Aspen Parkland

The Aspen Parkland is a transitional ecoregion comprised of trembling aspen (*Populus tremuloides* Michx.) and plains rough fescue (*Festuca hallii* (Vasey) Piper) that links grasslands of the Great Plains to the Boreal Forest. This characteristic intermingling of grassland and forested environments creates a mosaic of plant communities and habitats that vary spatially across the landscape (Moss 1932). Moreover, species composition in the Parkland is influenced by changes in climatic conditions and disturbance regimes (Vance et al. 1992, Campbell and Campbell 2000, Vujnovic et al. 2002, Simonson and Johnson 2005). Although climatic changes may work in concert with fire, particularly during drought, the notion of increased fire impacts on vegetation during drier periods has been challenged by some (Campbell and Campbell 2000). Nevertheless, the Parkland represents an ecological transition zone with dynamic plant communities that compete for available resources (Breshears and Barnes 1999).

European settlement and agricultural land modifications, and to a lesser extent, fire suppression, have collectively reduced native rough fescue (*Festuca hallii* (Vasey) Piper) grasslands to less than 5% of their original area (Trottier 1986, Gerling et al. 1995). Moreover, where uncultivated landscapes remain, fire suppression has often led to the loss of native grasses and/or aspen encroachment (Moss 1932, Moss and Campbell 1947, Johnson and Smoliak 1968, Rowe and Scotter 1973, Bailey and Wroe 1974, Sheffler 1976, Bork et al. 1997). Research from the 1960's indicated brush coverage in the Parkland landscape of southwest Alberta increased 3% over a twenty year period (Bailey and Wroe 1974). Much greater rates of encroachment, approaching 1% annually, were detected by Scheffler (1976) in the Parkland of east central Alberta.

Aspen encroachment has reduced both the productivity of rangeland for grazing livestock (largely cattle) (Bailey and Wroe 1974, Wheeler 1976), as well as led to the loss of native grassland habitats, considered important for various obligate wildlife species (Prescott and Murphy 1996). A primary objective of contemporary rangeland managers in the Parkland of Alberta has been to increase the availability and accessibility of preferred forage through a reduction in woody species. This reduction has been achieved using herbicides (Hilton and Bailey 1974), prescribed fire (Anderson and Bailey 1980, Bork et al. 1997), intensive livestock grazing (Bailey et al. 1990, Alexander 1995), or combinations of two or more control methods (Bailey et al. 1990, Alexander 1995).

1.4 Purpose of the Research

Although many studies have documented the effectiveness of various aspen control practices on short-term aspen control, including survival and regrowth, as well as forage responses, no comparative information exists on the long-term implications of various aspen control strategies implemented at the landscape level. The purpose of this research is to examine the long-term agronomic (forage yield and quality), ecologic (community diversity), and economic value of three contrasting brush management strategies available to producers operating in the Parkland. More specifically this research attempts to answer three primary questions:

1. What are the long-term agronomic and ecological responses to the three aspen management strategies?
2. Do native and tame grassland soils differ in soil microfaunal activity after 25 years?
3. Does aspen control pay (economically) in the long-term?

The aspen management strategies being compared were first established in replicate form in the late 1970's, and were assessed again during 2005 and 2006 for differences in forage quantity and quality, as well as plant community composition. Recent agronomic data were then combined with that from the early 1980's to conduct a net present value (NPV) economic assessment.

The 3 aspen management strategies include:

- Single point in time burn (Native Check) and retention of a landscape dominated by aspen.
- Extensive integrated control (i.e. Spray & Burn) of aspen using a combination of aerially applied herbicide, prescribed burning, and broadcast seeding of burned forests.
- Intensive aspen control (i.e. Clear & Break) using wholesale landscape conversion from native vegetation to tame pasture using bulldozing and brush piling, followed by sod breaking, soil preparation, and the seeding of tame forages.

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2 LITERATURE REVIEW: THE ASPEN PARKLAND

2.1 Physical Characteristics

2.1.1 Geography

The Aspen Parkland is a broad transition zone between Tallgrass, Fescue and Mixedgrass Prairies, and the Boreal forest (Moss 1932, Bailey and Wroe 1974). Geographically, the region is identified by Smoliak et al. (1990) from west to east, as a fringe along the foothills of the Rocky Mountains of southern Alberta that widens and expands NE across south-central Alberta into Saskatchewan, and then dips south through western Manitoba into northern Minnesota (Figure 2-1). In Alberta, the Parkland is approximately 50,000 km^2 (Moss 1932).

2.1.2 Climate

The continental temperate climate of the Aspen Parkland is characterized by four distinct seasons with long, cold winters, low humidity, prevailing westerly winds and elevated precipitation during the growing season (Pettapiece 1969, Moss 1932) that peaks in July. Mean annual precipitation is approximately 400 mm per year, with over half precipitated during the growing season (Asamoah et al. 2004). However, the region can experience varying degrees of moisture deficits late in the growing season (Strong and Leggat 1981).

The Parkland was glaciated until 20,000 years BP and underwent a series of glacial advances and recessions (approximately 18,000 years ago) that led to the formation of the Northern Great Plains (Campbell and Campbell 1997). Today the Parkland landscape has a combination of relatively level plateaus and widespread regions of rolling hills, the latter comprised of a mosaic of unsorted glacial till and sorted glacio-lacustrine deposits (Moss 1932).

2.1.3 Soils

The two prevalent soil orders of the region are Chernozems and Luvisols. Chernozems are characterized by a brown to black humified Ah horizon. The development of this thick (>10 cm) Ah horizon is attributed to drier soil conditions and significant below ground addition of organic matter from shallow rooted grasses and forbs (Jenny 1941, in Bélanger and Pinno 2008). The most common soils in the Parkland are generally Black Chernozems under productive grasslands, with Dark Gray Chernozems to Dark Brown Chernozems in more arid uplands (Wheeler 1976). Today the majority of the soils in the Aspen Parkland have likely been altered by the cultivation of annual crops or tame pastures.

2.1.4 Ecology

The Aspen Parkland is an ecological tension zone between two contrasting plant communities: Mixed Prairie grasslands to the south and the wetter aspen (*Populus tremuloides* Michx.) forests of the Boreal to the north. This landscape (gamma) diversity is influenced by localized variations in edaphic and climatic conditions and strong interspecific competition. These conditions create a fluid mosaic of interspersed dry upland grasslands and moist aspen groves (Moss 1932, Moss 1959, Pettapiece 1969, Wheeler 1976).

Vegetation in the Parkland can be categorized into two primary plant formations: trembling aspen forest groves and grasslands dominated primarily by plains rough fescue (*Festuca hallii* (Vasey) Piper). Aspen groves tend to form in bottom and north-facing forest positions with adequate soil moisture. In contrast, fescue grasslands are associated with well-drained upland sites, most often on warm south-facing slopes (Moss 1932). The microclimate within aspen groves can generate efficient moisture and nutrient recycling (Ellison and Houston 1958), which in the absence of disturbance, will lead to the expansion of aspen forest and a reduction in native grassland (Moss 1932).

2.2 Aspen Biology

Trembling aspen (*Populus tremuloides* Michx.) a member of the Salicaceae family, is North America's most widely distributed tree species (Fowells 1965, Peterson and Peterson 1992, Campbell and Bartos 2000). Aspen stands can tolerate a wide range of soil moisture and nutrient regimes (Farrar 1995) but grow best in well-drained, moist loamy soils (MacKinnon et al. 1992). Although aspen is an abundant seed producer, the conditions for germination are very specific, and therefore aspen more commonly reproduces vegetatively from adventitious root buds, producing genetically identical ramets (Barnes 1966, Romme et al. 2005, Strand et al. 2008). Barnes (1966) reported a clonal aspen stand of approximately 47,000 stems with an interconnected root system over 43 ha in size. More importantly, aspen can be a disturbance dependant species that provides habitat (food and cover) for a diversity of wildlife species (Prescott et al. 1995, McDonald et al. 1998, Månsson et al. 2007).

2.3 Aspen Dynamics

2.3.1 Drought and Climate Change

Variability in seasonal and annual climate influences the productivity and composition of plant species in the Aspen Parkland. The Parkland and adjacent regions have historically experienced intermittent drought (Bork et al. 2001, Bradshaw et al. 2007). Growing season deficits in soil water can limit plant growth, and hence forage production (Johnston et al. 1969). In semi-arid to arid regions ecosystem response to slight decreases in precipitation can lead to major reductions in plant cover and forage yields (Herbel et al. 1972). Although aspen is a fast growing, short lived species (Maini 1960, Cooke and Roland 2007), whose growth is inhibited by drought (Maini 1960), its ability to create microclimates that retain moisture and recycle nutrients reduces its susceptibility to variable precipitation. In contrast, the success of low growing grasses and forbs of open grasslands are limited by water availability and soil nitrogen (Lamb et al. 2007).

Terrestrial eutrophication (nutrient loading) of nitrogen from anthropogenic sources may contribute to the success of aspen encroachment in the Parkland. Köchy and Wilson (2001) measured atmospheric nitrogen deposition, available soil N and encroachment levels in six western Canadian national parks, four of which were closely related to the boundary of the Aspen Parkland. They hypothesized that the greater leaf area of trees and shrubs enabled greater interception of incoming nitrogen. Results of their study generally indicated that areas closer to densely populated regions had the highest rates of N deposition and available soil N, which in turn, corresponded to significantly higher rates of forest encroachment (Köchy and Wilson 2001). However, tree encroachment within Grasslands National Park of southern Saskatchewan remained low despite high N deposition, an observation attributed to a more arid moisture regime (Köchy and Wilson 2001). Although many studies indicate aspen expansion (Johnson and Smoliak 1969, Bailey and Wroe 1974, Sheffler 1976, Köchy and Wilson 2001), dieback and reduced growth of aspen has also been recorded in drier areas of the Parkland (Hogg and Hurdle 1995, Hogg et al. 2005).

The future environmental and socio-economical implications of climate change are difficult to predict. Climate forecast models for Canada project potential increases in mean annual temperatures from 2 to 6 °C over the next 50 years (Environment Canada 2005). Variable climate in aspen dominated forests could have significant implications on the timing and intensity of natural disturbances in this ecosystem, including fire regimes (Thompson et al. 1998) and insect populations (Hogg and Hurdle 1995). Although studies in the Parkland have focused on the expansion of aspen and its associated control, in many areas of western North America aspen stands are on the decline (Strand et al. 2008), leaving ecologists and land managers scrambling to retain critical aspen habitats.

2.3.2 Insects

Aspen dieback in the Parkland has been attributed to the combined effects of severe drought and insect defoliation. Aspen growth ring records of 180 trees examined by Hogg et al. (2002) in the Parkland of NW Alberta indicated that periods of reduced aspen growth coincided with outbreaks of defoliating insects. Herbivory by insects not only leads to reduced growth (Cooke and Roland 2007), but may also increase the susceptibility of aspen to secondary insect and fungal infections (Hogg et al. 2002), and subsequently increase the damaging effects of fire (Hogg et al. 2002).

2.3.3 Fire

Fire generally reduces tree and shrub cover and associated litter to restore natural openings and grassland communities (Anderson and Bailey 1979, Anderson and Bailey 1980). Structural changes in plant communities can have both facilitative and competitive influences on species within the community through changes in litter decomposition rates (Hobbie 2000, Powell and Bork 2006, Balshi et al. 2007), and the indirect influences of changes in canopy structure including available moisture and light (Throop and Archer 2007). Although the accumulation of plant litter helps to conserve soil moisture and reduce soil temperature, it may also delay plant growth and reduce both herbage yield (Dyksterhuis and Schmutz 1947) and species diversity (Weaver and Rowland 1952, Hopkins 1954). Plant communities on recently burned sites are generally greater in nutrients (Cowan et al. 1950, Bork et al. 2002) and initial forage production tends to increase (Anderson and Bailey 1980). However, Bogen et al. (2003) found that fire reduced the initial production of foothills rough fescue (*Festuca campestris* Rydb) despite an increase in tillering, largely due to slow regrowth on those plants affected by fire. Moreover, corresponding decreases in biomass production may make burned plants less likely to experience added defoliation stress by grazing animals, leading to improved recovery from fire.

Natural and anthropogenic fire have historically shaped Parkland plant communities and helped maintain a balance of grassland and forested ecosystems (Nelson and England 1971). Conversely, fire suppression since European settlement in the region has limited the influence of fire and contributed to an increase in brush encroachment (Rowe and Scotter 1973, Bailey and Wroe 1974). The result of fire suppression was to increase woody cover in the Alberta Parkland from only 5 to 10% at the turn of the 19th century (Bailey and Wroe 1974, Scheffler 1976), to 10-100% of native ecosystems in the 1970's (Bailey and Anderson 1980).

The elevated growing points of trees and shrubs make them susceptible to damage by fire. In contrast, grasses have lower growing points, which may be an adaptation to endure the effects of frequent fire (Bailey et al. 1990). Therefore, regular low intensity fires open up the aspen canopy and increase available forage. In contrast, aspen populations can remain relatively tolerant of fire, with aspen stem densities as much as doubling when exposed to severe fire (Keyser et al. 2005). Interestingly, the fires of 1988 in Yellowstone National Park had a positive impact on aspen forests. Changes in environmental conditions, including soil moisture, temperature and nutrients, appeared to favor the germination and survival of aspen seedlings. Widespread aspen establishment occurred even at higher elevations within stands that previously did not have an aspen component (Romme et al. 2005).

Nonetheless, used repeatedly, prescribed fire is a recognized tool that can be effective in reducing aspen and managing for a diversity of plant communities in the Parkland (Anderson and Bailey 1979, Anderson and Bailey 1980). Fire shifts the structure and dynamics of aspen communities by decreasing the height and cover of the canopy (Bork et al. 1997, Alexander 1995) thereby increasing the amount of light reaching the forest floor. Partial canopy removal in the Aspen Parkland can increase net primary production of understory vegetation through an increase in light availability and retained moisture in the leaf litter (Powell and Bork 2006).

2.3.4 Native Ungulate Herbivory

In the past, the Parkland provided late summer to mid winter seasonal habitat for large herds of bison (*Bison bison* L.) (Seton 1929, Stephenson et al. 2001, White et al. 2001). Bison would graze open grasslands late into the dormant season and calve in the shelter of river valleys and aspen groves (Moodie and Ray 1976). Fall and winter grazing by bison may have influenced plant species composition of the Parkland (Bird 1930) as bison likely selected for species like rough fescue, which is adapted to low-intensity, dormant-season grazing (Dormaar and Willms 1998). Bison foraging may have directly suppressed aspen growth and expansion via browsing of aspen shoots, and indirectly through physical damage from rubbing, trampling and stem breakage (Campbell et al. 1994). In support of this hypothesis, historic pollen records imply that aspen expansion in the Parkland closely followed the near extinction of bison (Campbell et al. 1994), although this also coincides with the simultaneous suppression of wildfires in the region.

Today, aspen regrowth may be reduced by an increased susceptibility of low growing trees and shrubs to the effects of browsing by cattle (Fitzgerald and Bailey 1984) and wildlife (Ripple and Larsen 2001). Aspen is an important source of winter browse for wildlife species (Romme et al. 1995). Historically, the Parkland provided forage and habitat for significant populations of elk (*Cervus elaphus* L.) and moose (*Alces alces* L.). Research in Elk Island National Park (EINP) and elsewhere indicates that large ungulate populations can play a role in aspen suppression (Bork et al. 1997, Kay and Bartos 2000). Records from EINP indicate that periods of expansion of grassland ecosystems coincided with high ungulate populations, and when ungulate populations were reduced, aspen groves expanded (Blyth and Hudson 1987, Campbell and Campbell 2000).

2.4 Contemporary Land Use

2.4.1 Livestock grazing

The Aspen Parkland of Western Canada constitutes approximately 86% of the Prairie Provinces forage production and contains 66% of the beef cattle herd

(McCartney 1993). In Alberta, the Parkland covers 12% of the land base. Today, less than 5% of Alberta's Parkland remains as native rangelands dominated by rough fescue grasslands, intermittent wetlands, and aspen groves. In contrast, more than half of the 5.5 million ha has been converted to annual cropping systems and another 36% managed as hay land and tame pasture (McCartney 1993). Furthermore, the biodiversity and sustainability of remaining native rangelands in Alberta's Parkland are threatened by fragmentation and overgrazing (Trottier 1986), or been encroached upon by aspen and shrub species (Sheffler 1976, Bailey and Wroe 1974). With a growing season only 4-5 months long (McCartney et al 1999), forage production in the Parkland can be variable from year to year and is further affected by localized site conditions. Considerable research in the 1970's and 80's focused on trembling aspen as a weed (Bailey 1986, Bailey et al. 1990), with a common perception that forest communities are undesirable habitats less conducive to supporting livestock (McCartney 1993). Mean annual herbage production under closed forest was less than 700 kg/ha despite browse production closer to 4000 kg/ha (Wheeler 1976). In contrast, mean annual dry matter yields of grasses on upland sites in the Parkland are approximately 2000 kg/ha (Asamoah et al. 2004).

Previous studies have documented the role of moderate livestock grazing in increasing species diversity at the community level (Milchunas et al. 1988); results which appear to hold for grasslands in western Canada as well (Bai et al. 2001). However, less is known about the role of landscape biodiversity in contributing to grazing opportunities, including the optimal mix of habitats for supporting cattle production.

High diversity of production from plant communities across typical Parkland landscapes may be important for supporting livestock throughout different times of the growing season (Asamoah et al. 2003). Moreover, cattle may actually prefer to utilize aspen stands early in the grazing season (Arthur and Bailey 1983, Asamoah et al. 2003). Bailey (1986) found that browse constituted 33-77% of cattle diets. As a result management for optimal livestock production may occur with diverse landscapes that offer a variety of foraging opportunities for livestock (including forest and wetlands), which in turn, should allow for greater selectivity during foraging, and thus improved opportunities to match forage intake with forage requirements.

2.4.2 Aspen Encroachment

Aspen encroachment into grasslands in the Parkland has been well documented (Moss and Campbell 1947, Johnston and Smoliak 1968, Bailey and Wroe 1974, Scheffler 1976). Simonson and Johnson (2005) observed a 6% change from grass to tree cover in remnant Parkland areas. Although this shift could be attributed largely to landscape management practices (i.e. grazing or land conversion) the influence of below-ground competition on above-ground dynamics is more difficult to evaluate including differences in soil structure, moisture and nutrients across the landscape, and the differential competitive ability of trees and grasses to utilize these soil resources (Meelis and Aveliina

2007). The success of aspen expansion into adjacent grasslands has been attributed to clustered clonal reproduction at high densities (Sankey 2008) which enables the species to benefit from heterogeneous distribution of soil resources (Meelis and Aveliina 2007). Aspen suckers (shoots) that have been continually mowed or grazed demonstrate an extraordinary ability to persist and can have root systems that extend 30 m in distance, 1 m in depth and generate annual vertical growth of 1.5 m (Buell and Buell 1959). In contrast the roots of grasses are smaller with a corresponding smaller access to available resources.

The potential loss of forage following aspen invasion is a key concern to ranchers and other rangeland managers. Consistent with the notion that aspen expansion reduces forage (herbage) production for both wildlife and cattle, Fitzgerald and Bailey (1984) demonstrated that forage production in aspen groves can be less than 10% that of adjacent grasslands in east central Alberta. This reduction is likely a worst case scenario that reflects dense tree cover. In contrast, other research indicates more conservative losses in forage production, with aspen forest falling to one-third that of adjacent grasslands in SW Alberta (Johnston and Smoliak 1968). Despite large effects of encroachment by aspen within remaining native grasslands of the Parkland, these impacts remain minimal compared to the broader effects of European settlement and associated conversion of native grasslands to tame pasture and cropping systems (Simonson and Johnson 2005).

The chemical properties of soils in the Aspen Parkland are strongly influenced by the plant communities they support. Generally native grassland communities on moist sites are represented by Black Chernozemic soils.

However, tree encroachment onto these grasslands leads to the loss of organic matter and associated nutrients, hence degrading Chernozems to Luvisols (Lutwick and Dormaar 1968). This process appears to occur quite rapidly, as aspen leaf leachate is known to cause rapid biochemical and physical changes to these soils in controlled studies (Lutwick and Dormaar 1968). While it is difficult to pinpoint the mechanisms as to how aspen are able to promote such significant changes in soil structure and plant community dynamics over a relatively short period of time (45 to 150 years) (Dormarr and Lutwick 1966), one could speculate that a number of adaptive characteristics associated with aspen ecology contribute to the success of this species. Strong vegetative reproduction, nutrient allocation, leaf resorption strategies and an extensive matrix of fine root systems that form symbiotic relationships with ectomycorrhizae fungi, are all life strategies that aid aspen success across the landscape (Peterson and Peterson 1992).

2.5 Aspen Management Strategies

There are two contrasting agricultural land management strategies for dealing with aspen encroachment, including intensive and extensive practices. The prevailing trend in the Aspen Parkland since the turn of the 20th century has been wholesale land conversion from native ecosystems to annual cropping systems and secondarily tame pastures. Traditionally, livestock producers have managed trees on the land base via the intensive clearing of trees and breaking of the soil followed by agronomic seeding (Bailey et al. 1984). However, livestock producers may benefit from a low cost alternative management strategy when compared to the high inputs of capital and labor costs associated with the standard clear break and seed (intensive land management) which may require refurbishment on a 5 to 10 year cycle. Conventional land clearing is a form of intensive management that significantly alters the resource base (Hedrickson et al. 2008). This type of intensification of agricultural systems may not be sustainable and can lead to a reduced carrying capacity through declines in soil organic matter and changes in nutrient regimes (Huang et al 2002).

In contrast, native pastures in the Aspen Parkland are examples of extensive pasture management strategies that include production systems with fewer inputs to the landscape. However, the winter feeding period of beef cattle in the Aspen Parkland is approximately 200 days (Entz et al. 2002), necessitating hay crops to support beef herds over the winter. Therefore, an integrated approach of extensive and intensive agriculture that includes both native pastures for summer range and tame pastures for winter feeding may be a valuable management strategy.

Where present, excessive increases in forest on native rangeland can be controlled by intensive or extensive management strategies, or by an integrated approach that includes both. The predominant method of removing aspen forest has been through widespread tree brushing and breaking of the soil (Bailey 1972, Bailey 1986). Known as land clearing, this process includes the mechanical removal of trees and shrubs with bulldozers or other means, and is followed by soil breaking with a disk or plow. This intensive method of forest removal requires significant financial investment and typically results in a minimum one year loss of grazing opportunity (Alexander 1995).

The negative effects of canopy closure from aspen encroachment can also be mitigated through the use of prescribed burning and appropriately timed high intensity, low frequency (i.e. mob) livestock grazing (Bailey et al. 1990) in either summer (Alexander 1995) or fall (Fitzgerald 1985). Early comparisons on the effectiveness of fire, grazing, and mechanical clearing suggested that an integrated approach using aerial herbicide application, fire, and broadcast seeding could be used to increase forage production at one-third the cost of traditional mechanical clearing (Bailey 1986).

Hilton and Bailey (1972, 1974) found that spraying of aspen in the Parkland with herbicides such as 2, 4-D increased forage production and those cattle preferred to use sprayed aspen stands, especially during dry conditions, suggesting aspen forest provided important emergency forage. Grazing management strategies that include seasonal heavy browsing of aspen suckers can play an important role in the integrated management of aspen encroachment (Fitzgerald and Bailey 1984).

2.6 Economic Modeling of Net Present Value

In today's global economy cattle producers are challenged to develop and maintain competitive management strategies that maximize profit and balance the diversity of societal and political views (Hendrickson et al. 2008). Net Present Value (NPV) modeling attempts to predict future outcomes of potential investment projects through the analysis of cash flows over time. Profits associated with cow-calf operations depend on the number of market weight calves the producer is able to sell (Miller 2002). Therefore, the land (i.e. aspen control) management strategy that offers the highest rate of return with a relatively low degree of risk would generally be the preferred investment (Novak et al. 1993). However, at the beginning of a project there is a high degree of uncertainty and as time passes this uncertainty (risk) decreases. In general, with higher levels of risk there is potential for greater returns but there is also a higher risk of losing money (Boardman et al. 1996).

Net Present Value (NPV) models are a tool used to analyze the economic potential of proposed capital investments. The NPV model uses a discount rate to estimate the present value of all future cash flows (Unterschultz and Quagrainie 1996). A discount rate is essentially an interest rate that includes a project specific risk premium designed to remove the time value of money from future cash flows (Unterschultz and Quagrainie 1996). Project specific risk premiums are defined as the amount someone would be willing to pay to move from an expected to a certain profit (Novak et al. 1993). Risk is measured by the potential for deviation from the expected returns (variance) (Novak et al. 1993). Therefore, greater variance or uncertainty in expected returns is associated with a higher discount rate for a particular investment.

Beef producers manage risk through retained ownership, on-farm diversification, and government programs (Unterschultz 2004). Diversification of the investment portfolio aids in reducing the potential impact of negative returns in any one investment (Unterschultz 2004). Net Present Value models predict annual quantitative costs and benefits over the life span of a project and attach a dollar value to those impacts. The quantitative data required to conduct an NPV assessment includes: the initial cost of the project (C0), the life (time) span of the project (T), and the expected cash flow in each period (C_1, C_2, ….. C_T) (Unterschultz and Quagrainie 1996). NPV is then calculated from equation (2):

$$NPV = C_0 + \sum_{t=1}^{T} \frac{C_t}{(1+r)^t} \tag{1}$$

The basic decision rule on whether to invest is based on the total present value of benefits and the total present value of costs. Where benefits exceed costs the project would be expected to proceed. However when comparing the NPV of multiple management strategies the strategy with the greatest return is generally the best option.

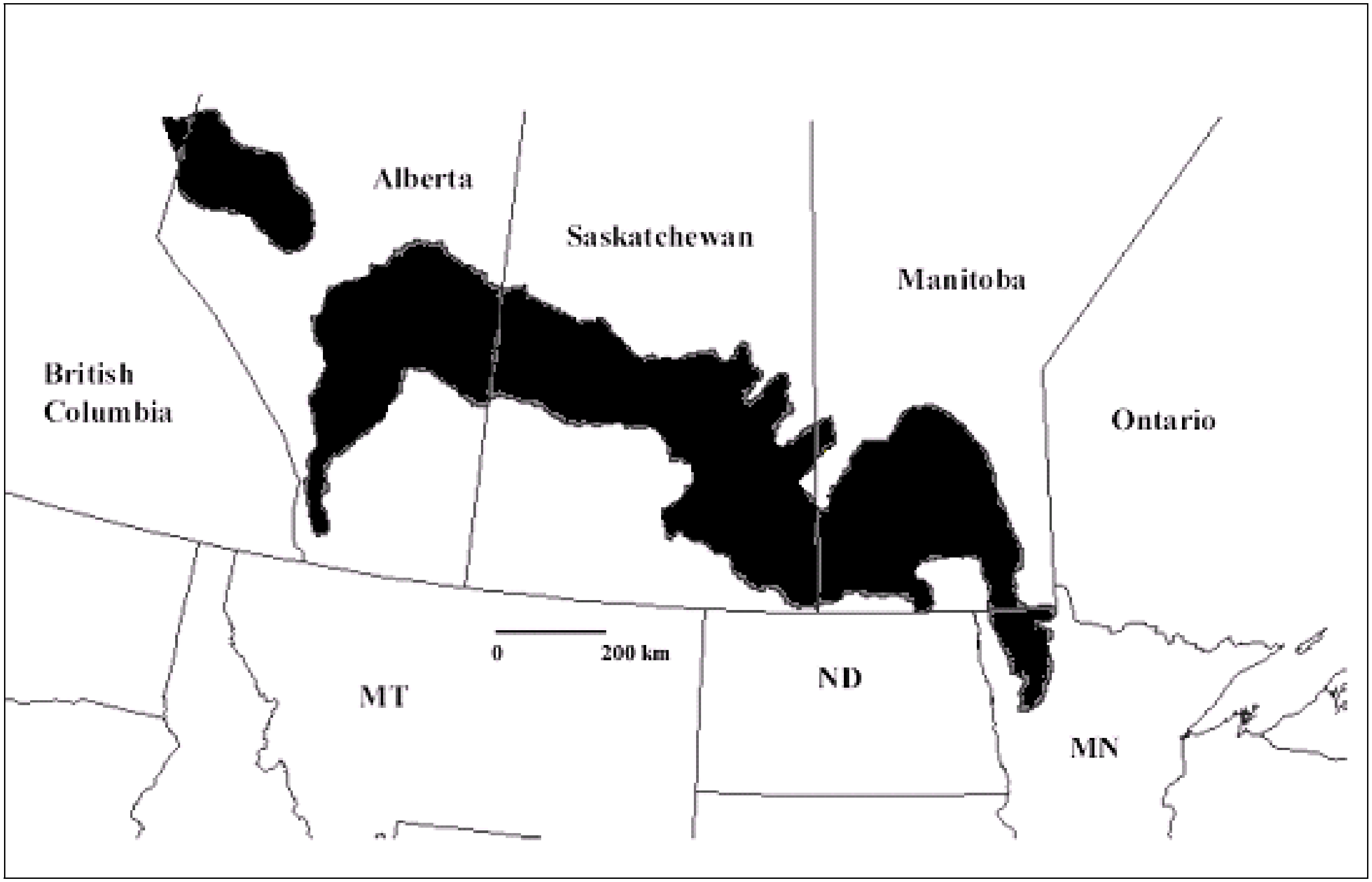

Figure 2-1: The Aspen Parkland Ecoregion of Canada and the northern United States (http://www.blm.gov/wildlife/pl_30sum.htm).

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3 LONG-TERM AGRONOMIC AND PLANT COMPOSITIONAL RESPONSES TO THREE ASPEN MANAGEMENT STRATEGIES

3.1 Introduction

Forage production in the Parkland can be highly variable due to plant community, soil and topographic characteristics (Wheeler 1976, Bailey 1986). Expansion of aspen into open grasslands has been widespread (Bailey and Wroe 1974, Sheffler 1976) and has led to considerable declines in herbaceous productivity (Bailey and Wroe 1974), thereby reducing the potential wealth of livestock producers. While the complexities of aspen expansion are not fully understood, changes in disturbance regimes (Anderson and Bailey 1979), climate (Köchy and Wilson 2001, Hogg et al 2002) and resource availability (Lamb et al. 2007, Powell and Bork 2007) may give aspen a competitive advantage over other species. Today, as a result of European settlement and wholesale land conversion to tame pasture or annual cropping systems, less than 5% of native Aspen Parkland grasslands remain intact (Trottier 1986).

3.1.1 Initial Project Background and Methods

In the past, rangeland conversion was based on the perception that native rangeland landscapes are less productive and therefore less profitable than lands converted into agronomic-based production systems. However, the risks associated with converting native Parkland rangeland into tame forage remain unclear. From 1979 to 1984, the Alberta Agricultural Research Institute Farming for the Future program funded research to compare two aspen management strategies at the University of Alberta, Kinsella Research Station. The first management strategy was a conventional Clear and Break (C&B) treatment, and the alternative a low-cost experimental Spray and Burn (S&B) treatment. Previous research has shown that both herbicides (Hilton and Bailey 1972, Sinton 1994) and repeated fires (Anderson and Bailey 1980, Alexander 1995) can contribute to aspen control.

3.1.1.1 Clear and Break Treatment

Clearing of two of the three C&B replicates was done in March of 1979. Later that same year the paddocks were broken twice using a Kelo belt offset disc and once again in the spring of 1980. Tame forages were then seeded (1980) with a press drill to a mixture of Magna smooth brome (5.2 lb/ac), Boreal creeping red fescue (2.6 lb/ac) and dryland alfalfa (1.5 lb/ac). No grazing was permitted in the establishment year of 1980. Due to equipment problems and delays the third replicate was cleared in the winter of 1979/80, broken and disced in the summer of 1980 but was not seeded until the autumn of 1980 (Bailey et al. 1984).

3.1.1.2 Spray and Burn Treatment

Firebreaks and fence lines were cleared between March 1979 and March 1980. The area was sprayed by helicopter mid June 1981 with 46 ounces per acre of 2,4-D butyl ester mixed with 7.3 gallons/acre of water. The three replicates of the S&B were treated with a prescribed burn on April 22nd 1981. On April 30th

1981 a helicopter was used to broadcast seed Kay orchard grass (4.9 lb/ac), magna smooth brome (4.9 lb/ac), creeping red fescue (2.6 lb/ac) and drylander alfalfa (2.2 lb/ac) (Bailey et al. 1984).

The objectives of this initial study were not only to compare the effectiveness of aspen control using C&B versus S&B, but also to compare long-term differences in forage production, utilization and live weight gains of beef cattle. These 3 years of data would then be used to compare costs and returns of the two aspen management strategies in a cost-benefit analysis.

Production data for the C&B and S&B treatments were initially collected for three growing seasons immediately after treatment (1981-1983). Unfortunately production data were not collected prior to treatment and data collection for the Native Check (NC) was limited to two years (1982 and 1983). Nonetheless, significant trends in both herbage and shrub biomass were observed (Table 3-1). During 1981 the C&B treatment exhibited a strong response with total production more than doubling compared to the S&B. In contrast the S&B demonstrated a one year delay in response, with total production surpassing the C&B by the second year of the study (1982). After 1981, the C&B exhibited a steady decline of approximately 1000 kg/ha/year in total production, which can largely be attributed to annual reductions in herbage production. In contrast, no significant differences in total production were observed in the NC treatment (1982-1983).

The initial report by Bailey and Irving (1984) further stratified production by treatment, growth form (herbaceous and shrub) and landscape position (Table

3-2). Those results indicated that shrub contributions to total biomass were consistently greater in both the S&B and NC compared to the C&B treatment. Notably, herbaceous production in the C&B remained consistently greater than both the S&B and Native treatment across all three landscape positions for all three years, with the exception of the bottom forest position in 1983. Hence, the initial comparison on the effectiveness of fire and mechanical clearing suggested that an integrated approach using aerial herbicide application and fire was comparable to those forage increases observed in the C&B (Bailey and Irving 1984).

From 1980 to 2004 the paddocks of the study area were managed as part of an operational ranch with rotational grazing occurring at approximately 2 Animal Unit Months per hectare. During this time period, (approximately 1990) portions of the original Native Check were treated by a low intensity prescribed burn (with no other treatment). For the purpose of clarity and consistency this study will continue to refer to this treatment as the Native Check.

The current study examines the long-term agronomic and vegetation compositional responses to the original three aspen management strategies conducted in 1980. More specifically, this paper reports on differences in forage quantity and quality, as well as plant species composition, between the C&B, S&B and NC treatments in 2005 and 2006, approximately 25 yrs after the initial treatments were conducted. Biophysical characteristics considered in the current study include comparisons of biomass production, plant species cover, species richness and the nutritional value (i.e. crude protein and digestibility) of available forage.

3.2 Materials and Methods

3.2.1 Study Area

The study area was located in east central Alberta at the 2,700 ha University of Alberta Kinsella Research Station, situated 150 km SE of Edmonton (53°0'N; 111° 31.2' W). The station is an active ranch that also serves as the center for range ecology and management research. Located in the Aspen Parkland natural subregion at approximately 700 m elevation, the station has large areas of predominantly intact native vegetation surrounded by a diversity of agricultural land uses, including abundant cultivation.

The continental climate of the region is characterized by long cold winters and short warm summers (Figure 3-1) with elevated precipitation in summer. Average annual precipitation is approximately 430 mm, with more than half occurring during the growing season (May to August), and peaking in July (Environment Canada 1971-2000) (Figure 3-2). A comparison of mean monthly precipitation and temperatures for the study years (2005-2006 and 1980-83) are summarized in Figures 3-1, 3-2, and Appendices 11 and 12).

The topography is described as "knob and kettle" due to its undulating landscape of glacial moraine knolls and ridges intermingled with depressions. Area soils are generally classified into three primary orders: Chernozems, Luvisols and Gleysols. Gleysols are typically associated with low-lying topographic positions that experience periodic or sustained saturation. Dark Grey Luvisols and Eluviated Black Chernozems are often associated with forest or shrubland, respectively. In contrast, soils under grasslands on upper slopes are generally Orthic Dark Brown or Black Chernozems (Wheeler 1976). The undulating landscape supports a diversity of range (i.e. plant community) types that provide a variety of grazing opportunities to cattle. Differences in late season grass production of upland grasslands (2000 kg/ha) and riparian meadows (5520 kg/ha) can be as high as 3000 kg/ha (Asamoah et al. 2004).

Dominant plant communities at Kinsella are representative of the Parkland and form a complex mosaic of aspen (*Populus tremuloides* L.) forest in mesic areas, open grasslands on well drained uplands, ecotonal western snowberry (*Symphoricarpos occidentalis* Hook) and silverberry (*Elaeagnus commutata* Bernh. ex Rydb) shrublands, and either freshwater or saline riparian meadows (Wheeler 1976). Dominant native grass species on upland grasslands include plains rough fescue (*Festuca hallii* (Vasey) Piper), western porcupine grass (*Hesperostipa curtiseta* Hitchc.) and western wheatgrass (*Agropyron smithii* Rydb). Introduced grasses common to the area include smooth brome (*Bromus inermis* Leyss), Kentucky bluegrass (*Poa pratensis* L.), and quack grass (*Agropyron repens* (L.) Gould).

3.2.2 Experimental Design

The experimental design for this investigation was first established in 1979 to compare the impacts of various aspen forest management treatments

(Bailey and Irving 1984). A total of 8 adjacent pastures were established in a 'wagon wheel' landscape study design, where each pasture was approximately 40 acres in size, and radiating outward from a central watering area. The initial experiment was designed to compare forage production among three management treatments across each of three topographic positions (grasslands, north-facing forests and bottom forests). Treatments were designed to contrast strategies for increasing livestock carrying capacity in the Aspen Parkland through both intensive and extensive aspen forest control. Each pasture (i.e. experimental unit) was selected to receive one of three aspen forest management treatments.

Clear and Break treatments (n=3 replicates) represented intensive conversion from native rangeland with aspen forest into tame pasture involving the mechanical clearing (i.e. dozing) of woody vegetation (aspen forest and shrublands) followed by sod breaking and seeding to introduced tame forages. Spray and Burn (n= 3 replicates) treatments were comprised of a more ecologically-based (less intensive) treatment to reverse aspen encroachment involving an initial aerial herbicide application of 2,4-D (intended to open up the canopy to facilitate burning), followed by prescribed burning and broadcast seeding of forests. Finally, an untreated native check (NC) treatment was included for comparison (n=2 replicates), though this area was exposed to a single, low intensity prescribed burn during the early 1990s.

Within each pasture, vegetation sampling was further stratified into 3 topographic positions, with 10 plots sampled at each position per pasture. Stratification by landscape position was undertaken to explore the influence of landscape position on plant species composition, forage quality and biomass. The three positions were assumed to be equally distributed (area-wise) and included upland grasslands, bottom aspen forest and north-facing forest.

Following the implementation of landscape treatments in 1979-80, initial sampling of vegetation between 1981 and 1983 was confined to the assessment of current annual biomass for each of the 236 plots. Samples were sorted into vegetation components, including herbage (i.e. grasses, sedges and forbs) and shrubs (i.e. current annual growth of shrubs and aspen saplings less than 2 m tall). Initial biomass responses of each treatment are summarized in Table 3-1, with further detail provided on the role of topographic position in Table 3-2.

3.2.3 Field Sampling

During the summer months of 2005 and 2006 (July 10^{th} until August 2^{nd}) plots were resampled for biomass availability to document treatment responses 25 years later. A similar sampling methodology was used to that employed in the 1980’s to ensure data compatibility. All plots were ungrazed by livestock prior to sampling in each year to ensure undisturbed plant growth.

During the summer of 2005 original field sampling plots were relocated as close as possible using aerial photos and documentation obtained from the original study in 1980. In most cases, 10 plots for each topographic position (upland grassland, N-facing forest and bottom forest) were sampled in each pasture. In four situations, the original sample plot could not be adequately relocated and was therefore excluded from the data set (i.e. 4 out of 240 plots).

Additionally, where recent disturbance had clearly altered the vegetation at the immediate plot these plots were moved a short distance to a representative habitat within adjacent areas of the landscape. In all, less than 10 plots had to be moved. Each re-located plot location was permanently marked and an NTS coordinate recorded using a hand held Geographic Positioning System (GPS) to facilitate relocation in 2006.

From July 15th until August 2nd of 2005 and 2006 relocated plots were sampled for peak biomass and vegetation composition. Each plot contained five, 0.25m^2 (50 x 50 cm) quadrats. A central quadrat was sampled for biomass, with four peripheral quadrats assessed for vegetation composition by foliar cover. Biomass quadrats were clipped to ground level and current annual growth sorted to growth form, including grass (and grasslike), forb and shrub components.

Peripheral quadrats were assessed for plant species composition of understory and midstory cover using visual estimates of foliar cover of each plant species, together with litter cover and exposed bare soil. Values were averaged across the four quadrats to obtain a single value for each cover component in each plot.

3.3 Biophysical Analysis

Harvested plant materials were oven dried at 50˚C to constant mass, weighed, and converted to kg/ha to determine annual production potential by growth form. Although there was no nutrient analysis done on the biomass samples collected during the early 1980’s, samples from 2005 and 2006 were assessed for nutritive value to identify any key differences in potential animal production arising from vegetation changes induced by the original treatments.

Biomass samples were ground through a 0.5 mm Wiley Mill in preparation for nutritional quality analysis. Individual samples were analyzed for nitrogen (N) and acid detergent fiber (ADF) concentrations using standard analytical techniques (Sweeny and Rexroad 1987). Concentrations of ADF were determined using the Ankom filter bag technique (Komarek 1993). The amount of ADF in forage samples is the fraction of non-digestible plant material such as cellulose and lignin, with lower ADF values associated with greater digestibility. Estimates of N were obtained using a LECO auto-analyzer (Sweeney and Rexroad 1987, Simonne et al. 1995, Lee et al. 1996), with N values multiplied by 6.25 to obtain crude protein (CP) concentrations [see equation (3)]. Crude protein yield (CPY) was derived by multiplying the proportion of CP by the biomass production (dry matter yield) in kg/ha [see equation (4)].

$$\%CP = \%N \times 6.25 \tag{3}$$

$$(\%CP/100) * DM \tag{4}$$

Finally, five samples of the most abundant plant species (17) were randomly collected and pooled by species to get a representative assessment of the quality (i.e. CP content) of these species across the landscape.

3.4 Statistical Analysis

Plant species cover values were subsequently used to determine species richness and diversity (Shannon-Weiner Diversity Index) for both native (i.e. endemic) and introduced vegetation, as well as for total species presence (n=236 plots).

Forage quality (CP and ADF concentrations) and quantity (biomass and CPY) data, as well as species richness and diversity, were each analyzed using a repeated measures Mixed Model ANOVA (fixed and random effects) over the two years of sampling, with aspen management treatment, topographic position and year of sampling as fixed factors. All data were tested for normality using the Shapiro Wilk test ($P>0.05$) (SAS Institute Inc. 2003). Biomass, crude protein concentration, crude protein yield, and acid detergent fiber concentration were all found to be non-normal and therefore normalized using a square-root transformation (Zar 1999).

For all analyses, experimental paddocks were considered the experimental unit, with paddocks random. All analyses utilized Least Squares Means (LSmeans) to predict population means of unbalanced designs. Main effects and interactions were considered significant at $p<0.10$ due to the relatively small number of paddocks (n=3, 3, 2) for each treatment. Additionally, in all analyses conducted, emphasis was placed on the aspen management treatments, including any 2 or 3-way interactions that included the main treatment with either topographic position, year of sampling, or both. For all significant effects or interactions, data were evaluated further using a Tukey mean comparison ($p<0.05$) to assess differences among primary treatments.

Finally, regressions were performed to identify relationships between variation in understory production (primarily herbage) and the amount of woody midstory (either aspen, or total woody species including trees and shrubs). Moreover, these regressions were done separately for different growth forms, topographic positions, and years. Empirical relationships derived from the regressions were essentially used to further describe the impact of the midstory where it remained on the associated understory, and assist with data interpretation.

3.5 Results

3.5.1 Long-Term Production Responses

Summary results of all ANOVA procedures evaluating the effects of aspen management treatment, and any interaction of this treatment with position or year of sampling, are provided in Table 3-3. Average annual net primary production (ANPP) during 2005 and 2006 ranged from a high of 4560±648 kg/ha on the NC to a low of 3284±529 kg/ha on the S&B treatment. Although total production of the NC was approximately 1000 kg/ha greater than both the C&B (3466±529 kg/ha) and the S&B treatments, average biomass of the three treatments did not differ ($p=0.35$).

There were also few differences in biomass involving the aspen management treatments within each growth form (grass, forb and shrub) investigated (Table 3-3). Only grass demonstrated a treatment by position effect

($p=0.05$). Grass production in the NC was greater ($p<0.05$) than the S&B treatment at both the bottom and north-facing forest positions (Figure 3-3). Additionally, the C&B treatment exceeded that of the S&B, but only in the bottom forest position (Figure 3-3).

As expected, the availability of biomass also varied among landscape positions for most vegetative components (all but forb) (Table 3-3). Grass biomass was lower ($p<0.05$) in north-facing forests than either grassland or bottom forest locations, while the latter did not differ from one another (Table 3-4). Moreover, this pattern also existed for total herbage biomass. In contrast, shrub biomass was greatest in north-facing forests, followed by bottom forest, and then grasslands (Table 3-4). When individual components were combined, total biomass remained greater ($p<0.05$) in each of the forested locations relative to open grasslands, by 608 to 967 kg/ha (Table 3-4).

Shrub biomass also exhibited a 3-way interaction between aspen management treatment, topographic position and year (Table 3-3). Closer examination of this effect indicated that the lone difference among treatments occurred in 2006 within the bottom forest position, when shrub production in the C&B (1193 ±296 kg/ha) was less ($p<0.05$) than that in the NC (2158 ±360 kg/ha) and the S&B (2056 ±296 kg/ha) treatments.

Finally, strong year effects were evident on the production of all growth forms, as well as total production (Table 3-3). Production was generally lower in 2005 compared to 2006. For example, grass (1825±176 vs 2578±176 kg/ha), forb (386±72 vs 733±72 kg/ha), shrub (515±130 vs 1577±130 kg/ha) and total biomass

(2690±334 vs 4850±334 kg/ha) increased by 41, 90, 306 and 80%, respectively, from 2005 to 2006.

3.5.2 Relationship Between Understory Biomass and Woody Cover

Within the Native Check (NC) plots, relationships between understory biomass and the midstory were limited largely to 2006, and were particularly strong within north-facing forests (see Table 3-5). At this location, grass and herbage (i.e. grass+forb) biomass consistently demonstrated strong negative relationships with woody cover, with up to 53% of the variance in understory explained by the midstory. Moreover, these relationships were consistent regardless of whether they examined the tree (i.e. aspen regeneration) component only, or included the intermediate shrub layer (Figure 3-4A). Reductions in herbage biomass reached 70 kg/ha for each additional 1% increase in woody cover. Also of note was that the inclusion of shrub biomass in current annual growth was unable to compensate for herbage decreases within north-facing forests, as evidenced by the continued negative relationship between total ANPP and aspen regen cover (ANPP declined by 115 kg/ha for each 1% increase in aspen) (Figure 3-4A).

Similar results were found in bottom forest plots of the NC, with strong declines in herb biomass in relation to increasing woody cover (Figure 3-4B). Unlike north-facing forests however, the inclusion of shrub biomass within estimates of ANPP were able to compensate for herbage declines under increasing aspen in bottom forests, thereby leading to no association between total ANPP and aspen cover (Figure 3-4B).

Few relationships were found between the understory and midstory aspen during 2005 in the S&B treatment, particularly bottom forests (Table 3-5). Within north-facing forests of this treatment, total herbage exhibited a moderate decline of 9.6 kg/ha for each 1% increase in the cover of shrubs and aspen during 2005 (Figure 3-5A). Despite this, total woody cover accounted for only 18% of the variance in herbage biomass, suggesting that other unknown factors were responsible for much of the variation in herbage biomass at this location. Also notable was the observation that within north-facing forests of the S&B, the inclusion of shrub biomass within ANPP appeared to offset any potential losses associated with the decline in herbage (Figure 3-5A).

Finally, similar results were found one year later during 2006 within both north-facing forests (Figure 3-5B) and bottom forests (Figure 3-5C) of the S&B treatment. At both locations, no relationship existed between herb biomass and total woody cover. However, when shrub contributions to total ANPP were examined in relation to increasing aspen regeneration (i.e. cover), ANPP increased with forest cover by 72 and 58 kg/ha for each 1% increase in aspen at the north-facing and bottom forest locations, respectively (Figure 3-5B and 3-5C).

3.5.3 Crude Protein (CP) Concentration

Treatment level differences in CP were observed for both grass (p=0.07) and forb (p=0.01) growth forms, but not shrub (Table 3-3). Grass CP concentrations were greater ($p<0.05$) in both the C&B and S&B treatments compared to the NC treatment (Table 3-6). However, this trend reversed for forbs, with CP concentrations greater ($p<0.05$) in the Native treatment relative to both the C&B and S&B treatments (Table 3-6). No higher level interactions were evident of the aspen management treatments on observed CP concentrations (Table 3-3).

Topographic position had relatively few impacts on CP values, with only shrub biomass displaying a response in CP (Table 3-3). Shrub CP values in grasslands (10.1±0.3%) were lower ($p<0.05$) than the CP values of shrubs harvested from the bottom forest position (11.4±0.3%). Grass CP concentrations also varied among years (Table 3-3), with grass CP values greater ($p<0.05$) in 2005 (8.6±0.2%) than 2006 (7.4±0.2%), a finding particularly apparent at the bottom forest position where CP was significantly greater in 2005 (8.51%) compared to in 2006 (7.19%). Finally, results of the individual plant species sampled for crude protein concentration are provided in Appendix 1.

3.5.4 Crude Protein Yield (CPY)

Only grass CPY exhibited treatment and treatment by position effects (Table 3-3). Grass CPY was significantly greater ($p<0.05$) in the NC (180±17 kg/ha) and C&B (180±14 kg/ha) treatments relative to the S&B (127±14 kg/ha) treatment. Closer examination of this effect, however, indicated that this difference occurred primarily within the bottom forest position, where the CPY of grass in the S&B (116±26 kg/ha) was much lower ($p<0.05$) than that obtained within either the C&B (229±26 kg/ha) or NC (253±32 kg/ha) treatment.

All growth forms examined (i.e. grass, forb and shrub), as well as total biomass, demonstrated responses in CPY to year of sampling (Table 3-3). Grasses increased ($p=0.08$) from 151.5±10.3 kg/ha in 2005 to 173.0±10.4 kg/ha in 2006, while forbs more than doubled ($p<0.001$) in CPY from 42.6±10.6 kg/ha in 2005 to 90.4±10.6 kg/ha during 2006. In contrast, shrubs declined sharply in CPY from 61.6±3.7 kg/ha in 2005 to only 21.2±3.7 kg/ha in 2006. Despite this decrease, total CPY from all growth forms increased ($p=0.02$) from 255.1±15.5 kg/ha in 2005 to 290.9±15.7 kg/ha in 2006.

Finally, both forb and total CPY exhibited a treatment by year effect (Table 3-3). Forb CPY was markedly greater across all three management treatments in the second year of this study (i.e. 2006). Increases in forb CPY from 2005 to 2006 were greatest in the NC treatment (43.5±20.7 kg/ha to 122.2±20.8 kg/ha), followed by the S&B (35.8±16.9 kg/ha to 74.4± 17.0 kg/ha), and finally the C&B (48.3± 16.9 kg/ha to 74.73± 16.9 kg/ha). Significant increases in total CPY were limited to the NC treatment only, with an increase of 96 kg/ha in 2006 over the year before [i.e. 269.2 kg/ha in 2005 vs 365.4 kg/ha) in 2006].

3.5.5 Acid Detergent Fiber (ADF) Concentration

Responses in ADF concentration were evident primarily in the grass component, and included treatment, position, treatment by position and treatment by year effects (Table 3-3). Among treatments overall, grass ADF concentrations were greater ($p<0.05$) in the S&B treatment (44.1±0.6 %) compared to both the C&B (39.3±0.6 %) and the NC (40.8±0.8 %). Among landscape positions, grass ADF was lower ($p<0.05$) in open grasslands (40.5±0.4 %) compared to both the north-facing forest (41.8±0.5 %) and bottom forest (41.9±0.4 %) positions. The treatment by position interaction within grass ADF revealed differences between treatments within each of the three topographic positions (Figure 3-6), the S&B was consistently greater ($p<0.05$) in ADF than the C&B treatment. In contrast, the NC and C&B treatments remained similar ($p>0.05$) in grass ADF concentration regardless of topographic position (Figure 3-6).

Grass ADF values also declined ($p<0.001$) from 43.3±0.4 % in 2005 to 39.5±0.4 % in 2006. The treatment by year interaction arose due to the presence of more pronounced differences in ADF concentration during the second year of sampling (Table3-7). While the S&B treatment was greater in ADF concentration than both the C&B and Native treatments in either year, during 2006 the C&B was associated with an additional reduction ($p<0.05$) in grass ADF relative to the Native treatment (Table 3-7).

Finally, the aspen management treatments exhibited limited effects on forb ADF concentration, with only a treatment by year interaction (Table 3-3). Similar to grasses, forb ADF concentration decreased from 36.8±1.1 % in 2005 to 30.2±1.1 % in 2006 ($p<0.05$), with no differences evident in forb ADF among treatments within a year (Table 3-7). No responses in ADF were evident in the shrub component (Table 3-3).

3.5.6 Plant Species Diversity and Richness

Significant effects on total plant species diversity and richness were limited to the main aspen management treatments (Table 3-3), and are provided in summary form in Table 3-8 and Table 3-9, respectively. Total species richness was lower ($p<0.05$) in the C&B treatment (11.1±0.5) compared to the S&B (13.7±0.6) and NC (13.7±0.7) areas. Similarly, total species diversity was lower ($p<0.05$) in the C&B treatment (1.76±0.05) relative to both the S&B (1.96±0.05) and Native (1.97±0.06) treatments. Total species richness and diversity also varied by year (Table 3-3), decreasing ($p<0.001$) from 14.2±0.4 and 2.03±0.03, respectively, in 2005, to 11.4±0.4 and 1.77±0.03 in 2006.

Both the richness and diversity of native plant species responded to aspen treatment (Table 3-3). Trends in native species richness and diversity paralleled those of total species richness and diversity, suggesting it was native species that were responding to the aspen management treatments. Native species richness was lower ($p<0.05$) in the C&B (8.6±0.5) than both the S&B (11.1±0.6) and NC (11.4±0.7) treatments. Similarly, native species diversity in the C&B was 1.45±0.07, a value lower ($p<0.05$) than that in the S&B (1.72±0.07) and Native (1.86±0.08) areas.

Native richness and diversity were also affected by treatment by position interactions (Table 3-3), indicating the effects of treatment varied depending on topographic position. Differences in native diversity were most evident at both the grassland and bottom forest positions. Within grasslands, diversity in the

Native treatment (H'= 1.90 ± 0.14) was significantly greater ($p<0.05$) than in the S&B (H'=1.52 ± 0.12) treatment. Notably, native species diversity of the C&B at the grassland position (H'=1.65 ± 0.11) did not differ ($p>0.05$) from either of the other two treatments. In contrast, native species diversity at the bottom forest position tended to be lower ($p<0.05$) in the C&B (H'=1.03 ± 0.12) compared to the same position in either the Native (H'=1.73 ± 0.15) or S&B (H'=1.65 ± 0.13) treatments. Native species richness responses were also limited to the bottom forest position. Native richness at this location was lower ($p<0.05$) in the C&B treatment (5.71 ± 0.89) compared to both the S&B (10.79 ± 0.98) and Native (10.5 ± 1.13) treatments.

Introduced species diversity was affected by treatment and treatment by position effects (Table 3-3). Introduced species diversity was generally greater ($p<0.05$) in the C&B (0.55±0.03) and S&B (0.56±0.03) treatments than the Native (0.43±0.04) treatment. However, closer examination indicated that the lower introduced diversity in the NC treatment compared to the C&B and S&B was limited to the grassland and bottom forest positions (Table 3-10). In contrast, introduced diversity remained similar across all treatments at the north-facing forest position.

Finally, both introduced species diversity and richness were affected by a treatment by year effect (Table 3-3). Differences in introduced diversity among treatments were expressed to a greater extent during 2005 than 2006. In contrast, introduced plant species richness remained remarkably similar among treatments in 2005, but was greater in the C&B compared to the other treatments in 2006 (Table 3-11).

3.6 Discussion

This research examines differences in forage quantity (biomass and CPY) and quality (CP and ADF concentrations) between two contrasting agricultural management strategies for dealing with aspen encroachment in the Aspen Parkland, including intensive land conversion to tame pasture and a more extensive program of aspen control within existing native pastures. Although most agricultural research is short-term, with little information on the long-term impacts of certain management practices, this study is unique as it provides insight into the long-term responses of the system to integrated management for aspen control (i.e. C&B vs S&B). In doing so, this research provides valuable data needed to assess the collective impacts of these management systems on both ecological and economic sustainability. An understanding of how native and converted pastures at the Kinsella Research Station differ in forage quantity and quality, as well as species richness and diversity, will aid in the implementation of grazing management systems that optimize their use.

3.6.1 Long-term Production Responses

Total forage production during 2005-06 contrasted sharply with 1980-83 (Figure 3-7). While production was greatest in the C&B and S&B in the 1980's, presumably due to the introduction of high yielding agronomic species

(McCartney 1993) or the potential increase in fertility associated with nutrient release following cultivation and/or burning (Roberts et al. 1989, Cowan et al. 1950, Bork et al. 2002), production tended to be greater within the NC relative to the S&B by 2005-06. These findings are consistent with Willms et al. (1999) who observed a decrease in yield of introduced crops/forages as nutrient resources became limited and the agro-ecosystem lost 'vigour'. The lack of further differences among treatments during the latest sampling period likely reflects, at least in part, the low number of paddocks (2-3 per treatment) involved in the study and associated limited experimental power to detect treatment effects. Nevertheless, grass production was the key component that remained greater in the NC over the S&B, specifically within the two forested locations (i.e. bottom forests and north-facing forests). This was unexpected, and although unlikely may reflect some negative residual impact of more severe fire occurring 25 years earlier within the S&B. For example, while fire is known to temporarily increase soil temperatures, microbial activity, mineralization, and subsequent soil nitrogen (<5 years) (Driscoll et al. 1998), this can be followed by sharp declines in soil N availability (Marion and Black 1988, Carreira et al. 1994), particularly following advanced decomposition of roots from woody species killed by fire. Given that aspen communities are often associated with relatively low fertility Gray or Dark Gray Luvisolic soils (or derivatives thereof), the low grass production found within forested locations suggests that the introduced forages in the C&B were well adapted to respond to the initial nutrient pulse from 1980-83, but thereafter may have been susceptible to stagnation under low nutrient conditions (Kentucky bluegrass and Smooth brome). Similarly, loss of topsoil after fire in areas that once supported shrubs could lower the potential of these areas to produce forage (White el al. 2006). Animal behavior and grazing preferences can also affect long-term forage production and species composition as cattle tend to concentrate grazing within recently burned areas due to an increase in the accessibility of more palatable and nutritious forage (Wright 1980, in White et al. 2006).

Stagnation of introduced forages, particularly those consisting of rhizomatous species, has commonly been observed within seeded forage swards of western Canada (Lardner et al. 2002), and may account in part for the reduction in grass growth within swards containing seeded species. Stagnation is characterized by compacted soils and limited space for new root growth under rhizomatous vegetation, in turn limiting the ability of vegetation to access nutrients, with large inputs of fertilizer required to achieve production increases (Lardner et al. 2002). Stagnation alone is unlikely to account for all of the production declines observed within the understory of forests within the S&B however, particularly as cover data from the S&B indicated that a limited amount of introduced grasses remained within 'forested' locations of this treatment (x=29.7% cover). Instead, mechanisms other than stagnation may account for the poor grass production at this location.

For instance, residual forest communities within the S&B treatments following aspen regrowth are prone to reduced light levels, and introduced species may be less tolerant of low light than native species (Cabin et al. 2002), thereby accounting for the depressed herbage yields. Second, aspen and shrub regrowth is likely to have occurred following initial treatment, results corroborated by cover estimates of shrubs (see Appendix 2), which in turn, may have suppressed grasses in the understory. Third, greater grass production within bottom forests of the NC may be partly attributed to more recent invasion of introduced ‘tame’ grasses from adjacent grasslands. Belowground connectivity of smooth brome and Kentucky bluegrass provides not only a means for plants of these species to share resources (Otfinowski and Kenkel 2008), but may also redistribute soil nutrients to support adjacent populations, and could therefore increase biomass under recent encroachment (and before stagnation). Finally, differences in grass production could be caused by differences in water availability (Abrams et al. 1986) and/or available nutrients including nitrogen and phosphorus. Notably, competition processes are mostly belowground in nature within soils containing low nutrients and/or water (Casper and Jackson 1997). In contrast, sites with adequate nutrients and moisture (i.e., bottom forests) may experience greater competition for light as the limiting factor for growth (Newman 1973). The presence of tame grasses would not only influence total biomass, but could also lead to an increase in species richness, particularly in the mesic bottom and north-facing landscape positions.

An intermediate level of grass production associated with the C&B treatment in 2005-06 is unexpected, as stagnation could be expected to be particularly apparent 25 years after conversion to introduced agronomic forages. However, this reduction did not occur to the same degree as in the S&B, which may be due to the simultaneous absence of forest cover coupled with the presence of native grasses returning to the community 25 years after land conversion (see Appendix 2-4). Native grasses may be stabilizing biomass production in grasslands of the C&B, as native species would contribute to increased stability through increased diversity (Tilman et al. 2006).

Although herbage biomass within north-facing forests was markedly lower than that observed in the bottom forest and grassland positions, the inclusion of shrub biomass into estimates of total ANPP resulted in both forested positions producing greater potential forage than adjacent grasslands. This observation reinforces the importance of woody species in contributing to total forage availability within the Aspen Parkland region (by 600-1000 kg/ha). Shrubs not only make significant contributions to available biomass, they also meet the nutritional requirements for all classes of grazing cattle (NRC 1989). Holechek et al. (1982 ab) found that the contributions of forbs and shrubs played an important role in the vegetation composition of cattle diet in forested pastures and that during summer grazing forbs and shrubs could make up to 71 percent of cattle diets. Holechek and Varva (1982) speculate that cattle intake would be greater on lands with high component of palatable forbs and shrubs over open grasslands, mainly during periods of forage dormancy or drought.

While shrub production was greatest on north-facing slopes, shrubs continued to be present within all treatments, including the C&B (both grassland and forest habitats). North-facing slopes likely received less direct sunlight due to the severity of slope gradients (~4-8^{o}), thereby reducing light availability and increasing the potential moisture availability for the understory. These conditions, in turn, would favor plant species moderately tolerant of shading (Marquis 1975), and in the process convey a competitive advantage to taller shrubs (i.e. over neighboring herbs).

Although changes in forage quality were minimal in response to the landscape treatments after 25 years, grass CP and ADF demonstrated contrasting responses to the treatments. Greater grass CP in the C&B and S&B relative to the NC may be associated with the prior establishment of introduced grass cultivars, many of which were selected based on favorable protein content (Anderson et al. 1988), including species like smooth brome, bluegrass, and other early growing soft grasses (Willms et al. 1996), all of which remained present in 2005 (Appendix 3). In contrast, the NC retained greater rough fescue, junegrass and wheatgrass, all of which were representative of a hard grass growth form (Adams et al. 2003). Hard grasses have morphological properties such as a small leaf area that tends to reduce forage quality during the early and mid growing season (Smoliak and Bezeau 1966), results corroborated by species level sampling in the current study (Appendix 1). Interestingly, low grass CP values within communities of the NC were at least partially offset by favorable CP concentrations within the forb component, and to a lesser extent the shrub component.

Not surprisingly, ADF responses of grasses were opposite CP, with the greatest ADF (i.e. lowest digestibility) found in grasses of the S&B treatment. Though likely to be high in N (i.e. CP), introduced forage grasses are also typically early growing, and thus early senescing, which in turn, is likely to increase ADF and reduce digestibility in these species by the time of sampling at peak growth in mid to late summer (July/August). This reduction in digestibility was most apparent in the S&B where more than 50% of the cover consisted of Kentucky bluegrass (Appendix 3). Despite the observed reduction in grass CP within the NC relative to the other treatments, total grass CPY remained greater or similar in the NC. These results indicate the yield of native grasses within the latter treatment remained sufficiently greater to offset any reductions in CP concentration due to the retention of native plant species.

Grass CP concentrations were also greater in 2005 (8.6%) than 2006 (7.4%), a response attributable to the cooler growing conditions in the first year of sampling. By 18 July 2005, the study area had accumulated only 1224 growing degree days (GDD) compared to 1364 by that time in 2006. Cooler temperatures in 2005 appeared to slow growth, thereby maintaining vegetation at an earlier phenological state relative to those plants clipped at the same time in 2006, perhaps accounting for the greater quality during the first year. Differences in CP were most pronounced within the bottom forest position, where 2006 values (7.2% CP) were markedly lower than those in 2005 (8.5% CP). Bottom forests would be particularly susceptible to delayed phenology due to delayed snowmelt and soil warming. These results highlight the impact of inter-annual variability in altering not only forage quantity but also forage quality, and thereby highlight the tradeoff associated with these two responses between years: greater production in 2006 coincided with lower forage quality, while the reverse was true in 2005. Similar trends have been observed elsewhere (Clark et al. 2000, Ganskopp and

Bohnert 2003), with lower forage quality in years with abundant soil moisture and greater quality in drier years. Nonetheless, these results might differ if sampled according to phenology.

Shrub quality generally remained greater in bottom forests compared to upland grasslands, and may reflect landscape-based differences in water and nutrient availability, as well as changes in shrub composition between landscape positions. Mesic and nutrient-rich conditions in bottom forests, coupled with delayed growth due to an elevated water table in spring, may result in vegetation that is relatively younger phenologically at the time of harvest in midsummer. In addition, greater shrub quality may arise due to the unique composition of shrubs at that location, which included abundant rose, a species known to be high in quality (Renecker and Hudson 1988). In contrast, the predominant shrub within upland grasslands was western snowberry (*Symphoricarpos occidentalis*) (Appendix 5), which is not known for being a high quality, palatable shrub species (Fitzgerald et al. 1986).

The lack of difference in total CPY values between the C&B and NC treatments suggests that both these management systems have similar potential for overall livestock production in the Aspen Parkland after 25 years. However, this also assumes that cattle will feed with similar preference on all grasses, forbs and shrubs. Previous studies have shown that cattle grazed under rotational systems in the Parkland will preferentially occupy forested habitats early in the year, even to the point of achieving similar relative use among forest, meadow and grassland habitats (Asamaoh et al. 2003). Preferential use of woody species in aspen habitats under intensive grazing systems has been used as an effective form of biological control of aspen following burning (Fitzgerald et al. 1986, Alexander 1995). Cattle demonstrate a particularly strong affinity for forests in drought periods (Hilton and Bailey 1974), as well as later in the growing season (Fitzgerald et al. 1986) during which time aspen stands and their associated understory's provide important emergency forage. It should also be noted that similarities in CPY among treatments may be attributed to the entry of tame species into native pastures (see Section 3.6.3) and vice-versa, essentially resulting in significant 'homogenization' of habitats across treatments in the study area. Conversely, both spraying (Hilton and Bailey 1972, 1974) and burning (Bailey et al. 1990) of aspen forests have been shown to directly attract cattle. Concentrated use by livestock over an extended period within this treatment may have led to the observed negative impacts on long-term forage production and quality within the S&B treatment, and may also account for the unexpected positive relationship between understory grass biomass and aspen cover within this treatment (see Section 3.6.2 below).

Finally, interpretation of the production data from the NC treatment must be done cautiously. In addition to having limited replication (n=2), this treatment experienced a low intensity prescribed burn during the early 1990's, thereby complicating interpretation of data from this treatment. The more recent fire event within this treatment may partly explain the greater forage yields in 2005-06, as the effects of aspen removal and associated changes to nutrient cycling may have favored greater productivity at the time of recent sampling. Similarly, comparison of herbage and browse production within this treatment was observed to be markedly above that of an adjacent area (Field 9A) that had not received fire over the previous 25 years (see Appendix 8). Both mean herbage (3391 kg/ha) and total biomass (4959 kg/ha) in forested areas of the NC treatment were above that observed in Field 9A, which had herbage and total biomass of only 1084 and 2173 kg/ha, respectively. However, production responses in the latter may also be lower due to differences in growing conditions rather than the lack of fire, including baseline soil characteristics. As a result, the interpretation of production responses among treatments is most appropriate between the C&B and S&B treatments in this study.

3.6.2 Relationship Between Woody Cover and Understory Biomass

Herbage availability was consistently greatest in open grasslands rather than forests, a finding consistent with previous studies documenting spatial forage availability in the Aspen Parkland (Asamoah et al. 2003, Bailey and Wroe 1974). Numerous studies have documented a reduction in productivity in areas with high tree densities compared to adjacent openings (Burrows et al. 1990, Powell and Bork 2007). Within forested communities, competition between trees and grass can be regulated by differences in available moisture and light. Reduced herbage under aspen highlights the competitive nature of woody vegetation and its key influence in reducing understory productivity and associated opportunities for livestock grazing. Moreover, greater reductions in herbage were found within plots of the NC treatment where aspen stands remained at or near their potential canopy closure in the absence of recent disturbance.

Despite the apparent negative influence of the aspen midstory on understory growth, the inclusion of shrub production in estimates of forage availability had a notable impact in altering midstory-understory relations in some landscape positions (i.e. all bottom forests, and north-facing forests of the S&B), even to the point of resulting in similar or compensatory total ANPP responses across forests of varying canopy closure. Provided that the shrub species found produced biomass suitable for consumption by cattle, the contribution of these species to production appeared capable of offsetting the suppressive influence of the midstory on understory herbage availability.

Differences in midstory-understory relationships were also evident between sampling years, with more prominent suppression of the understory during 2006. The limited relationship between the midstory and understory in 2005 may reflect the growing conditions of that year. Although total annual precipitation was similar between years, spring rains in 2006 were more timely and associated with warmer temperatures (Figure 3-1 and 3-2), and may have led to accelerated tree leaf-out early in the season, thereby contributing to increased light reductions and stronger negative relationships between the midstory and understory in that year. Conversely, the combined effects of moisture timing and/or cool temperatures in reducing herbage growth in 2005 may have limited competitive suppression by the midstory during that growing season, as the frequency and intensity of precipitation events combined with soil structure influence individual plant growth (Robertson et al. 2009).

Marked differences were expressed between the impact of aspen trees on north-facing and bottom forests in influencing understory productivity, both of the herbaceous and shrub components. The greatest understory suppression was consistently found on north-facing slopes, and was particularly strong in the NC under dense aspen cover. These findings support the notion that habitats in this treatment continued to have the most well developed forested communities despite the single prescribed burn conducted during the 1990s. The extensive woody midstory at this location, particularly during the superior growing conditions of 2006, likely reduced both light and water availability simultaneously, thereby leading to the greatest negative impact on the understory. Moreover, the inability of shrubs to compensate for herbage declines within north-facing slopes of the NC suggests that abundant trees reduced productivity at this location by impacting both the herb and shrub layers. Thus, removal or reduction of aspen appears to be particularly important on north-facing slopes where competition on the understory is expressed more strongly.

Similar to north-facing forests, bottom forests of the NC also exhibited negative relationships between herbage and woody cover, with one key difference. Inclusion of shrubs into ANPP estimates compensated for herbage declines, suggesting that overall competition by the tree midstory may be lower at this location of the landscape compared to north-facing slopes. There are several explanations for this finding. In addition to the previous incidence of fire, bottom forests were generally situated in a landscape position where vegetation can access near surface moisture, reducing the likelihood of competition for water. Similarly, bottom forests were typically on landscape profiles with little to no slope, and can therefore be expected to have greater incident light than north-facing slopes (i.e. with ~5-10% slopes). Thus, light availability may have been less limiting for understory growth in bottom forests compared to north-facing forests, despite mesic conditions favoring forest development. Although herbs and shrubs may compete with one another, changes in herb abundance in the bottom forest position appeared to be offset by increases in shrub growth, thereby stabilizing ANPP. Notably, this same pattern of shrub compensation for herbage declines was found in north-facing forests of the S&B, and could therefore have been made possible by the previous reduction in tree canopy cover arising from the aspen control treatment.

Differences in the type and intensity of competition between the midstory and understory within bottom forests may also explain the divergent responses observed at this location between different landscape treatments: herbage was negatively influenced by woody species in the NC, and positively in the S&B. Within NC plots, grass and total herbage were generally high under low woody cover (7559-8950 kg/ha), only to sharply decline under an increasing woody canopy. Moreover, the strongest negative relationships were observed when both shrub and tree cover were jointly included, suggesting that both trees and shrubs detrimentally impacted grass and/or forb (i.e. total herb) biomass. Given the abundant moisture present during the 2006 growing season, and the lack of significant relationships in 2005, perhaps the apparent competition in 2006 was related not to moisture, but rather to limitations in light reaching the understory within these heavily forested plots in the absence of any previous major disturbance. This is further supported by the speculation that bottom forest plots should have had ample moisture in both years of the study due to overland flow and an elevated underlying water table.

In contrast, herbage biomass in bottom forest plots of the S&B treatment exhibited a very different relationship to the midstory. Biomass in these plots was generally low, even in plots with little to no woody cover (i.e. 1460 kg/ha grass) following prior herbicide application and burning, and subsequently increased in relation to greater aspen (but not shrub) cover. There are several potential explanations for this observation, including that grass and aspen growth were simply auto correlated, with each dependent on another as of yet unaccounted for environmental factor not measured in the current investigation. However, another equally plausible explanation is that low density aspen stands may facilitate greater grass production through facilitative interactions among plant species. A partial aspen canopy has been shown to support greater grass growth by reducing frost (i.e. lengthening the growing season), conserving water (by lowering temperatures and associated evaporation in the understory), and promoting water use efficiency by lowering water vapor differentials for underlying herbs (Powell and Bork 2007). Therefore, understory vegetation can exhibit a positive response to increased shade under low moisture conditions (Belsky 1994). A number of studies have documented improved productivity under isolated (mature) trees

(Holland 1980, Stuart-Hill et al. 1987, Belsky et al. 1989, 1993a, 1993b, Frost and McDougald 1989, Weltzen and Coughenour 1990, Belsky 1994).

3.6.3 Long-Term Species Dynamics

Lower total species richness and diversity within the C&B treatment compared to the others was primarily due to a lower number of native species. This response is a direct artifact of the widespread conversion of a more diverse native grassland and forest into seeded agronomic communities. As expected, C&B treatments had greater introduced species richness and diversity than the other treatments, suggesting distinct residual effects associated with aspen control 25 years prior.

Nevertheless, the finding that native species diversity of grasslands in the C&B did not differ from that of the other treatments indicates that despite the intensive nature of this management treatment, many native species continued to survive at this topographic position, either through volunteer establishment from the seedbank (Johnston et al. 1969), or from vegetative propagules surviving the process of forest clearing, tillage and subsequent seeding (Hughes and Fahey 1991). Moreover, as grasslands are often on drier, exposed south-facing slopes in these landscapes, native species may have superior adaptations compared to introduced forage (Baruch et al. 1985), thereby enabling them to survive better under these conditions and leading to their favorable return. For example, native bunchgrasses and co-habitants are known for having very deep rooting depths (Coupland and Johnson 1965), which will increase their competitiveness under low nutrient and moisture conditions.

In contrast to grasslands, the lower native species richness and diversity within bottom forests of the C&B compared to the S&B and NC suggests that introduced species may be more competitive where moisture and nutrients are in greater supply (Baruch et al. 1985). Agronomic species in particular are known to be highly responsive to moisture and nutrient availability. In the current study, plant species that appeared to benefit the most from these conditions were smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*) (Appendix 6). Positioned in or near depressional areas, bottom forests had abundant moisture and nutrients due to overland flow, a near-surface water table, and likely deposition of nutrients from grazing animals that may prefer to occupy depressional areas (Asamoah et al. 2004). These factors combined may have favored seeded agronomic species within the C&B that are adapted to producing abundant aboveground shoot biomass (Nernberg and Dale 1997). Greater shoot biomass in turn, would lead to greater suppression of remaining native plant species (Nernberg and Dale 1997), particularly the many short-statured native forbs found in the understory of aspen forests.

3.7 Conclusion and Management Implications

The purpose of this research was to examine the long-term agronomic (forage yield and quality) and ecologic (community diversity) responses among three contrasting brush management strategies available to producers operating in the Parkland. These strategies represent contrasting philosophies on the degree of inputs necessary to control aspen and theoretically restore rangeland productivity, and thereby optimize wealth. Results demonstrated that a number of key treatment-based effects in terms of species richness and diversity, the relative dominance of introduced and native species, and forage production (i.e. yield and quality) remain.

Although the initial increase in forage production observed in the C&B and the S&B treatments (1980-83) appeared to be positive, this benefit was not evident 25 years later. Instead the S&B had lower grass production compared to both the C&B and the NC. Production within the NC and forested portions of the S&B was largely maintained by the presence of production from shrubs, which remained an important contributor to forage quality and production. Moreover, contributions of browse tended to stabilize production, both spatially across the landscape and between years. Therefore, it is important to consider the contributions of aspen and shrubs to biomass production and their ability to create microclimates that retain moisture and recycle nutrients (Maini 1960), thereby providing an optimum forage supply on a season-long basis (Asamoah et al. 2003).

Although the results of this study indicate that the Native Check can provide a consistent level of forage, it is important to recognize the potential influence of more recent disturbances, including the prescribed burn in the early 1990’s and the introduced forages that have invaded into these pastures. Today, the majority of the Aspen Parkland has been modified beyond natural (i.e. historical) disturbance regimes (e.g. introduction of agronomic pastures and cropping systems), which in turn, may have a strong influences on the ecology of remaining fragments of native Parkland vegetation. Declining native Parkland vegetation may result in habitat loss for wildlife that are obligate dependent on one or more vegetation types within the Parkland landscape. Given the ability of adaptive management and innovative grazing strategies to effectively utilize, at least in part, available shrub and tree biomass, these results suggest that the long-term agronomic benefits provided by native Parkland environments that include shrublands and forests may be similar to those derived from other areas treated for aspen control, including areas that have undergone wholesale land conversion into introduced forages.

3.8. Literature Cited

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Table 3-1: Mean dry matter (DM) yield (kg/ha) of forage available by treatment in the Aspen Parkland of central Alberta (1981-1983). Data are adapted from Bailey et al. (1986).

		Forage Production		
Year	Treatment	Total	Herbage	Shrub
1981	Clear & Break	5115 a[1]	4342 a	773 de
	Spray & Burn	2121 d	656 d	1464 bc
	Native Check	n/a[2]	n/a	n/a
1982	Clear & Break	4111 b	3490 b	621 de
	Spray & Burn	4304 b	2186 c	2119 a
	Native Check	1807 d	752 d	1055 cd
1983	Clear & Break	3171 c	2706 c	473 e
	Spray & Burn	4383 b	2562 c	1821 ab
	Native Check	1778 d	947 d	831 de

[1] Within a column, means with the same letter do not differ, $P < 0.05$, according to a Tukey's test.
[2] n/a Indicates data were not collected for that period.

Table 3-2: Mean dry matter biomass (kg/ha) of forage available by treatment and topographic position in the Aspen Parkland of central Alberta (1981-1983). Data are adapted from Bailey et al. (1986).

		Herb (Grass + Forb)			Shrub		
Topographic Position	Aspen Treatment	1981	1982	1983	1981	1982	1983
Grassland	Native Check	n/a[2]	1404 b	2141 a	n/a	73 a	122 a
	Clear and Break	2686 a[1]	3226 a	2586 a	605 a	614 a	537 a
	Spray and Burn	1200 b	2026 ab	2014 a	413 a	454 a	521 a
North Forest	Native Check	n/a	364 b	272 b	n/a	1817 b	1454 b
	Clear and Break	3622 a	3075 a	2900 a	1242 a	826 b	381 b
	Spray and Burn	328 b	1834 a	2639 a	2165 a	3508 a	2997 a
Bottom Forest	Native Check	n/a	488 c	429 b	n/a	1275 a	916 ab
	Clear and Break	6719 a	4168 a	2640 a	473b	424 b	470 b
	Spray and Burn	442 b	2696 b	3034 a	1820 a	2393 a	1994 a

[1] Within a topographic position and year, means with different letters differ, P<0.05, according to a Tukey's test.
[2] n/a Indicates data that were not collected in 1981.

Table 3-3: Summary of ANOVA results (p-values) assessing differences in biomass, crude protein (CP) concentration, crude protein yield (CPY), and acid detergent fiber (ADF) of various vegetation components, including grasses, forbs, shrubs, and total ANPP, together with responses in the diversity and richness of native, introduced and all plant species.

Response	Component	Treatment (T)	Position (P)	T*P	Year (Y)	T*Y	P*Y	T*P*Y
Biomass	Grass	0.17	0.01	0.05	<0.01	0.36	0.78	0.21
	Forb	0.59	0.10	0.35	<0.01	0.24	0.80	0.40
	Shrub	0.64	<0.01	0.25	<0.01	0.68	<0.01	0.08
	Total ANPP	0.35	0.03	0.78	<0.01	0.37	<0.01	0.12
CP	Grass	0.07	0.55	0.32	<0.01	0.22	0.04	0.38
	Forb	0.01	0.63	0.14	0.16	0.81	0.35	0.18
	Shrub	0.41	0.01	0.28	0.59	0.94	0.73	0.42
CPY	Grass	0.05	0.01	0.07	0.08	0.64	0.01	0.18
	Forb	0.57	0.57	0.81	<0.01	0.02	0.26	0.63
	Shrub	0.67	<0.01	0.33	<0.01	0.62	<0.01	0.18
	Total ANPP	0.11	0.12	0.34	0.02	0.02	0.34	0.20
ADF	Grass	<0.01	<0.01	0.05	<0.01	<0.01	0.07	0.44
	Forb	0.86	0.69	0.61	<0.01	<0.01	0.50	0.43
	Shrub	0.36	0.68	0.56	0.27	0.31	0.24	0.52
Diversity	Native	0.04	0.03	0.09	<0.01	0.43	0.22	0.69
	Introduced	0.01	<0.01	0.10	<0.01	<0.01	0.12	0.13
	Total	0.06	0.01	0.19	<0.01	0.29	0.39	0.96
Richness	Native	<0.01	<0.01	0.08	<0.01	0.51	0.72	0.21
	Introduced	0.21	<0.01	0.19	<0.01	<0.01	0.08	0.36
	Total	0.01	<0.01	0.19	<0.01	0.43	0.97	0.18

Table 3-4: Mean (±SE) dry matter biomass of various forage components by landscape position in the Aspen Parkland of central Alberta during 2005-06.

Response Variable	Landscape Position	Biomass (kg/ha)
Grass	Grassland	2234 ± 188.5 a
	North Forest	1802 ± 190.46 b
	Bottom Forest	2569 ± 216.22 a
Forb	Grassland	508 ± 74.42 a
	North Forest	635 ± 76.44 a
	Bottom Forest	536 ± 74.9 a
Shrub	Grassland	476 ± 148.7 c
	North Forest	1493 ± 165 a
	Bottom Forest	1170 ± 163.72 b
Herbage	Grassland	2739 ± 243.21 a
	North Forest	2408 ± 247.49 b
	Bottom Forest	3077 ± 266.33 a
Mean Total DM Production	Grassland	3245 ± 363.04 b
	North Forest	3853 ± 376 a
	Bottom Forest	4212 ± 386.26 a

1 Within a response variable, means with different letters differ, $P<0.05$.

Table 3-5: Summary relationships between understory biomass [either grass (G), herbage (H), or total ANPP] and the amount of woody midstory (either aspen alone or aspen and shrub combined), within plots exposed to each of two landscape treatments, Native Check (NC) and Spray and Burn (S&B) in north forest (NF), and each of two sampling years.

Treatment	Position	Year	Midstory (X)	Response (Y)	R^2	P-value	Relationship
Native Check	NF	2005	Aspen	Grass	0.12	0.14	1908-60.3x
Native Check	NF	2005	Aspen	Herbage	0.18	***0.07***	2449-66.6x
Native Check	NF	2005	Aspen	ANPP	0.07	0.28	3432-38.6x
Native Check	NF	2005	Aspen + Shrub	Grass	0.01	0.66	1901-5.4x
Native Check	NF	2005	Aspen + Shrub	Herbage	0.02	0.55	2478-6.7x
Native Check	BF	2005	Aspen	Grass	<0.001	0.97	2310+2.1x
Native Check	BF	2005	Aspen	Herbage	<0.001	0.92	2609-5.6x
Native Check	BF	2005	Aspen	ANPP	0.02	0.60	3081+26.4x
Native Check	BF	2005	Aspen + Shrub	Grass	<0.001	0.65	1535-3.3x
Native Check	BF	2005	Aspen + Shrub	Herbage	0.01	0.64	2829-9.0x
Native Check	NF	2006	Aspen	Grass	0.37	***0.006***	3338-70.3x
Native Check	NF	2006	Aspen	Herbage	0.36	***0.007***	4841-98.0x
Native Check	NF	2006	Aspen	ANPP	0.21	***0.05***	7775-115.4x
Native Check	NF	2006	Aspen + Shrub	Grass	0.48	***0.001***	4916-47.2x
Native Check	NF	2006	Aspen + Shrub	Herbage	0.53	***0.0004***	7293 - 70.2x
Native Check	BF	2006	Aspen	Grass	0.004	0.78	4352-20.2x
Native Check	BF	2006	Aspen	Herbage	0.01	0.66	5337-35.2x
Native Check	BF	2006	Aspen	ANPP	0.0005	0.92	7066+9.2x
Native Check	BF	2006	Aspen + Shrub	Grass	0.24	***0.03***	7559-83.9x
Native Check	BF	2006	Aspen + Shrub	Herbage	0.26	***0.02***	8950-97.6x
Spray & Bn	NF	2005	Aspen	Grass	0.06	0.21	1126-7.2x
Spray & Bn	NF	2005	Aspen	Herbage	0.07	0.19	1471-8.0x
Spray & Bn	NF	2005	Aspen	ANPP	0.03	0.36	2025-6.7x
Spray & Bn	NF	2005	Aspen + Shrub	Grass	0.13	***0.06***	1537-7.7x
Spray & Bn	NF	2005	Aspen + Shrub	Herbage	0.18	***0.03***	2003-9.6x
Spray & Bn	BF	2005	Aspen	Grass	0.001	0.86	1427-2.4x
Spray & Bn	BF	2005	Aspen	Herbage	0.001	0.87	1849-2.3x
Spray & Bn	BF	2005	Aspen	ANPP	0.003	0.80	2362+3.5x
Spray & Bn	BF	2005	Aspen + Shrub	Grass	0.009	0.65	1535-3.3x
Spray & Bn	BF	2005	Aspen + Shrub	Herbage	0.04	0.34	2152-7.3x
Spray & Bn	NF	2006	Aspen	Grass	0.08	0.17	1333+2.1x
Spray & Bn	NF	2006	Aspen	Herbage	0.02	0.53	2165+11.5x
Spray & Bn	NF	2006	Aspen	ANPP	0.14	***0.06***	3473+71.3x
Spray & Bn	NF	2006	Aspen + Shrub	Grass	0.05	0.27	1138+10.4x
Spray & Bn	NF	2006	Aspen + Shrub	Herbage	0.0008	0.89	2251+1.5x
Spray & Bn	BF	2006	Aspen	Grass	0.13	***0.06***	1460+26.9x
Spray & Bn	BF	2006	Aspen	Herbage	0.1	0.11	2269 + 26.4x
Spray & Bn	BF	2006	Aspen	ANPP	0.15	***0.04***	3922+58.3x
Spray & Bn	BF	2006	Aspen + Shrub	Grass	0.01	0.62	1614+5.1x
Spray & Bn	BF	2006	Aspen + Shrub	Herbage	0.004	0.75	2474+3.7x

Table 3-6: Mean (±SE) crude protein (CP) concentration of forage within each of three aspen management strategies in the Aspen Parkland of central Alberta from 2005 through 2006.

Response Variable	Management Strategy	CP (%)
Grass	Native Check	7.41 ± 0.33 b[1]
	Clear and Break	8.37 ± 0.27 a
	Spray and Burn	8.25 ± 0.27 a
Forb	Native Check	12.99 ± 0.36 a[2]
	Clear and Break	11.56 ± 0.3 b
	Spray and Burn	11.55± 0.3 b
Shrub	Native Check	11.18 ± 0.47 a
	Clear and Break	10.34 ± 0.39 a
	Spray and Burn	10.91 ± 0.38 a

[1] Within the grass response variable, means with different letters differ, P<0.10.
[2] Within the forb response variable, means with different letters differ, P<0.05.

Table 3-7: Mean (±SE) concentration of acid detergent fiber (ADF) in grasses and forbs in each of three aspen management treatments of the Aspen Parkland of central Alberta in 2005 and 2006.

Growth Form & Year		Aspen Management Treatment		
		Native Check	Spray and Burn	Clear and Break
Grass	2005	42.42 ± 0.76 a[1]	44.94 ± 0.61 b	42.5 ± 0.62 a
	2006	39.17 ± 0.87 b	43.18 ± 0.7 a	36.15 ± 0.7 c
Forb	2005	36.19 ± 2.16 a	34.77 ± 1.77 a	39.51 ± 1.76 a
	2006	30.14 ± 2.25 a	31.3 ± 1.83 a	29.14 ± 1.84 a

[1] Within a row, means with different letters differ, P<0.05.

Table 3-8: Mean (±SE) species diversity (Shannon's Index) of plant species found in each of three aspen management treatments in the Aspen Parkland of central Alberta during 2005-06.

Response Variable (F-stat signif.)	Management Treatment	Diversity Index
Total Species Diversity	Native Check	1.97 ± 0.056 a[1]
(P<0.06)	Clear and Break	1.76± 0.046 b
	Spray and Burn	1.96 ± 0.046 a
Native Species Diversity	Native Check	1.86 ± 0.081 a
(P<0.04)	Clear and Break	1.45± 0.066 b
	Spray and Burn	1.72 ± 0.069 a
Introduced Species Diversity	Native Check	0.427 ± 0.04 b
(P<0.10)	Clear and Break	0.552 ± 0.032 a
	Spray and Burn	0.555 ± 0.034 a

[1] Within a response variable, means with different letters differ, P<0.05.

Table 3-9: Mean (±SE) plant species richness found in each of three pasture management treatments in the Aspen Parkland of central Alberta during 2005-06.

Response Variable	Management Treatment	Species Richness
Total Species	Native Check	13.69 ± 0.65 a[1]
	Clear and Break	11.1 ±0.53 b
	Spray and Burn	13.65 ±0.55 a
Native Species	Native Check	11.39 ±0.65 a
	Clear and Break	8.6 ±0.53 b
	Spray and Burn	11.07 ±0.55 a
Introduced Species	Native Check	2.16 ± 0.12 a
	Clear and Break	2.41 ± 0.1 a
	Spray and Burn	2.44 ± 0.1 a

[1] Within a response variable, column means with different letters differ, P<0.05.

Table 3-10: Mean (±SE) introduced plant species diversity (Shannon's index) by aspen management treatment and landscape position in the Aspen Parkland of central Alberta during 2005-06.

Landscape Position	Native Check	Clear and Break	Spray and Burn
Grassland	0.11 ± 0.07 b	0.35 ± 0.06 a	0.33 ± 0.06 a
North Forest	0.59 ± 0.07 a	0.55 ± 0.06 a	0.61 ± 0.06 a
Bottom Forest	0.58 ± 0.07 b	0.76 ± 0.06 a	0.73 ± 0.06 a

[1] Within a row, means with different letters differ (P<0.1).

Table 3-11: Mean (±SE) introduced species diversity and richness by study year and aspen management treatment in the Aspen Parkland of central Alberta.

		Aspen Management Treatment		
Response	Year	Native Check	Spray and Burn	Clear and Break
Diversity	2005*	0.50 ± 0.05 c	0.65 ± 0.04 a	0.56 ± 0.04 b
	2006	0.36 ± 0.05 b	0.46 ± 0.04 ab	0.55 ± 0.04 a
Richness	2005	2.48 ± 0.15 a[1]	2.82 ± 0.12 a	2.42 ± 0.12 a
	2006	1.85 ± 0.15 b	2.07 ± 0.13 b	2.41 ± 0.12 a

[1] Within a year and response, means with different letters differ, P< 0.05
* Within this year, means with different letters differ, P< 0.1

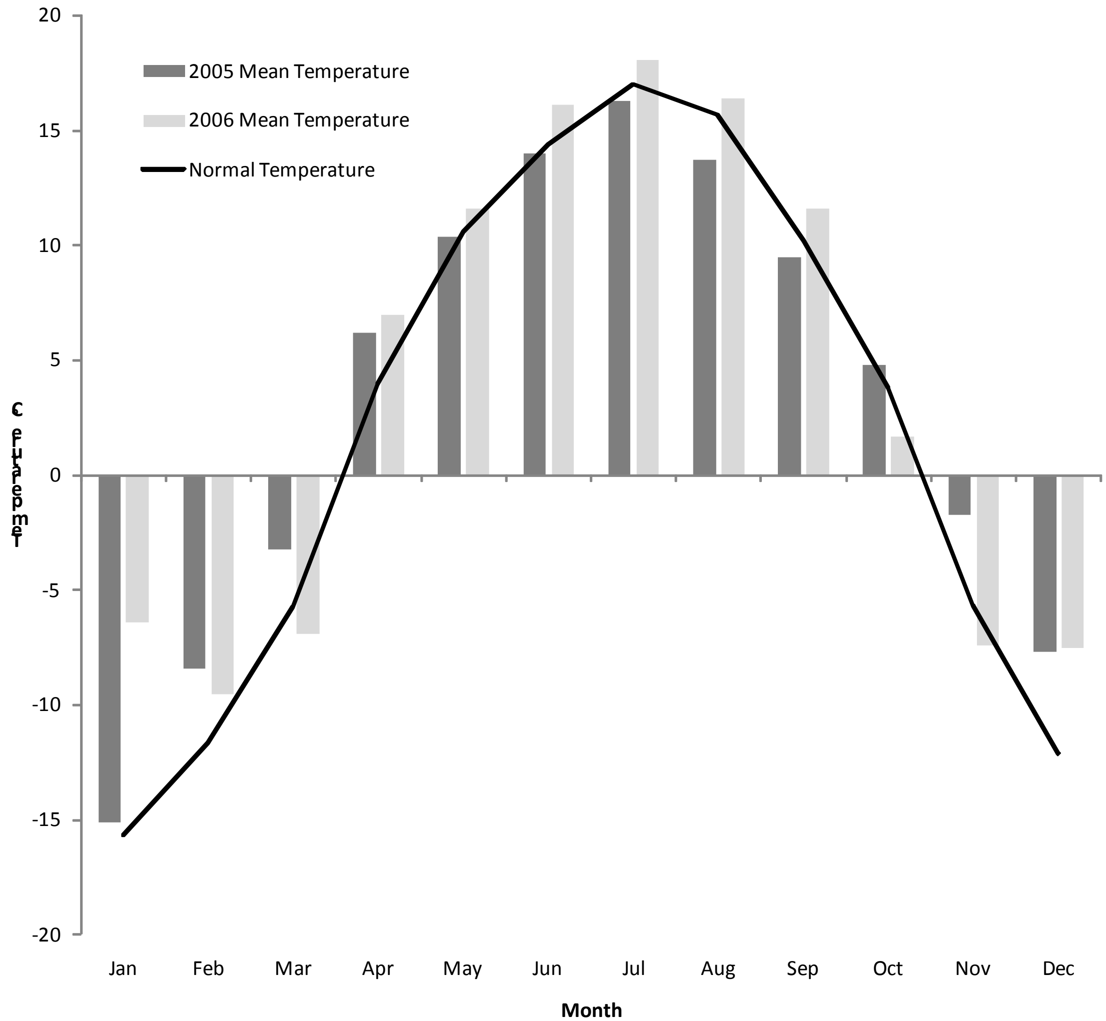


Figure 3-1: Comparison of mean monthly temperatures for 2005-2006 to regional norms (Environment Canada 1949-2004)

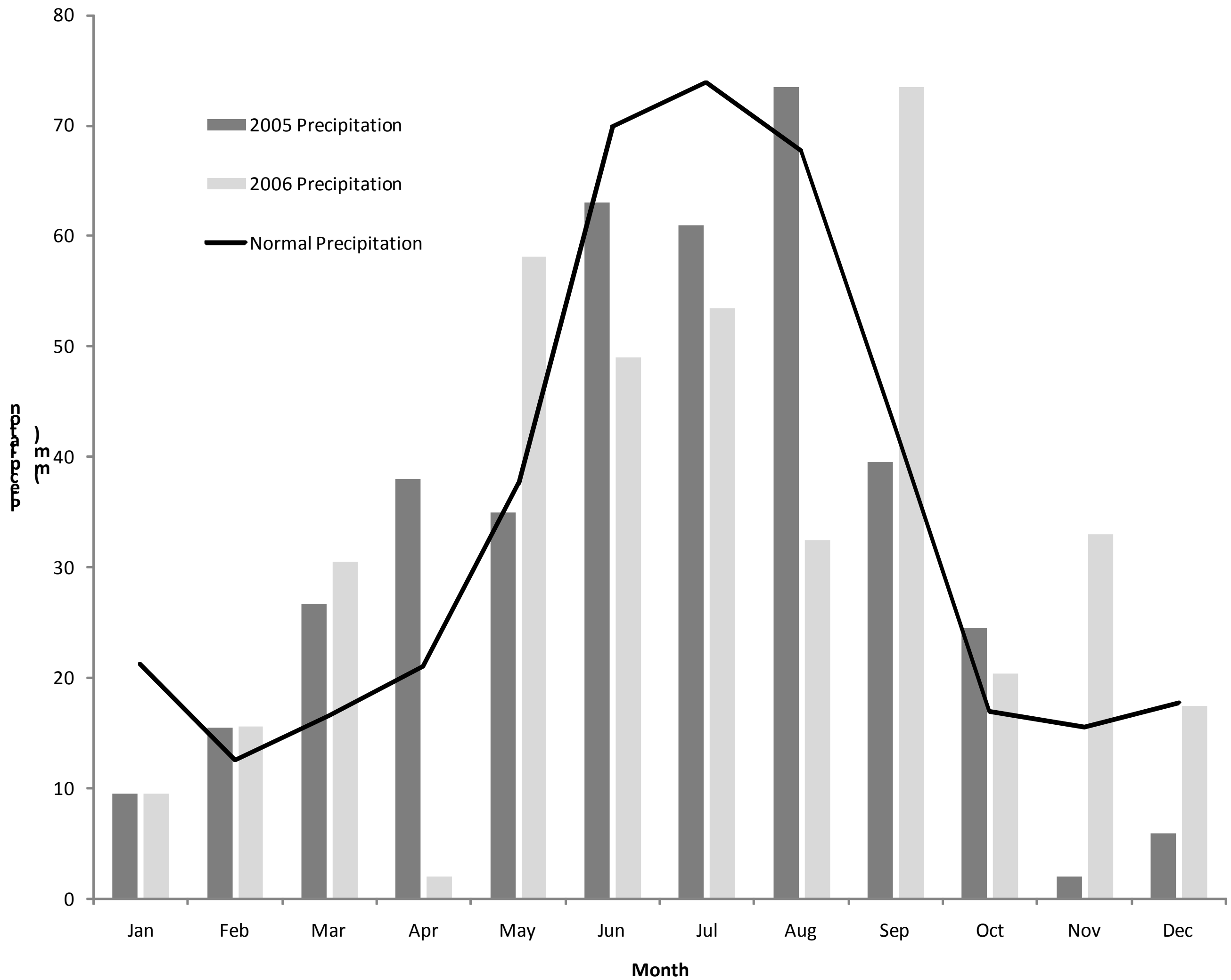


Figure 3-2: Comparison of mean monthly precipitation for 2005-2006 to regional norms (Environment Canada1949-2004)

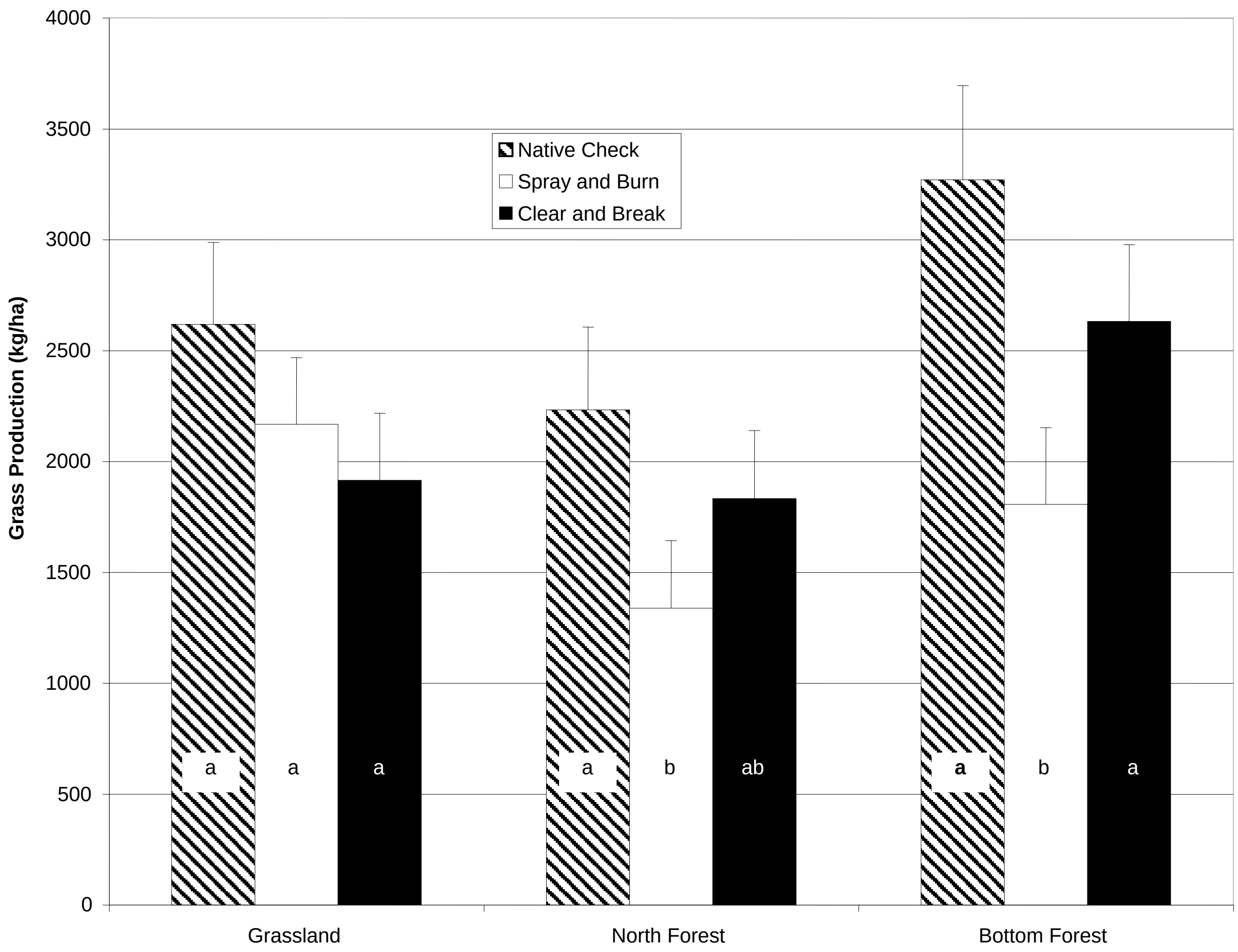
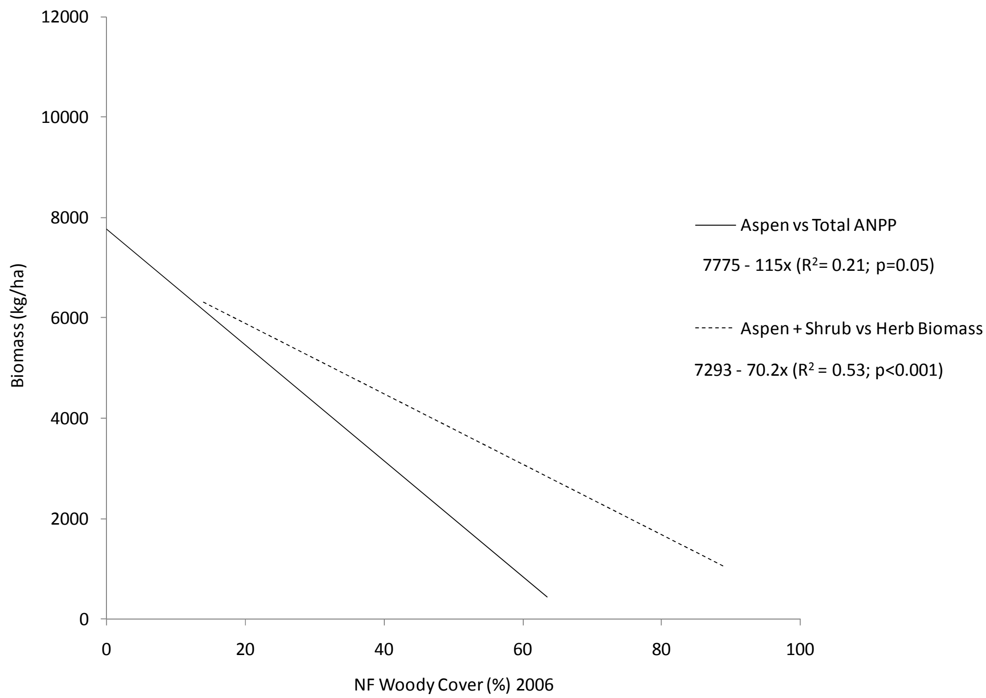


Figure 3-3: Comparison of mean (±SE) grass production in each of three aspen management treatments and three landscape positions from 2005-06. Within a position, treatment means with different letters, differ, $P<0.05$.

(A)



(B)

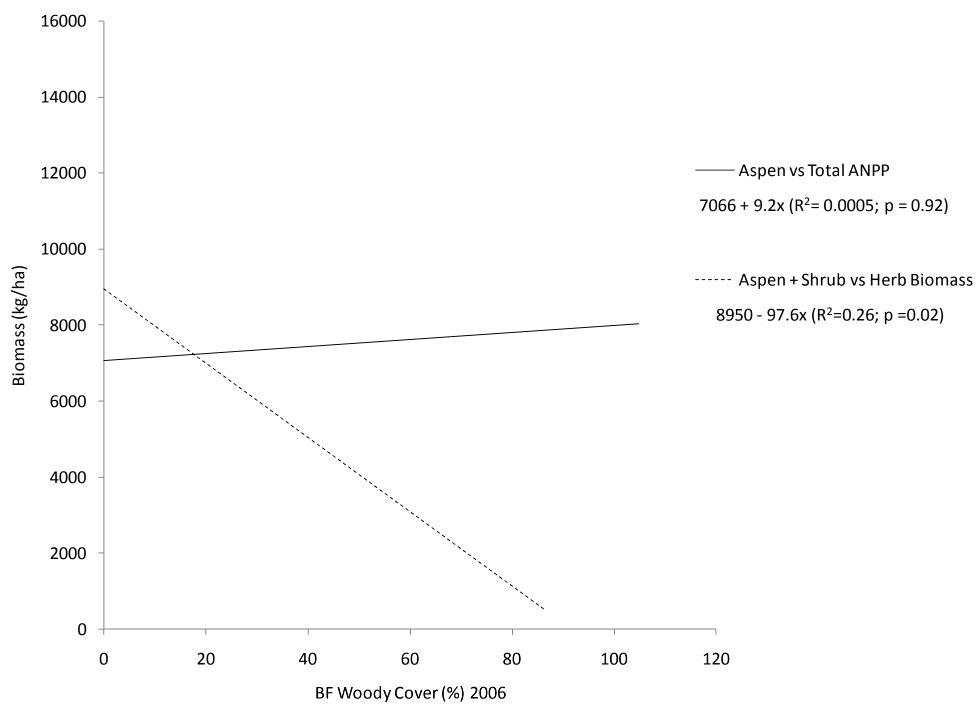


Figure 3-4: Understory biomass changes in response to overstory woody canopy cover in Native Check for (A) north facing (NF) and (B) the bottom forest (BF) position during 2006

A)

B)

C)

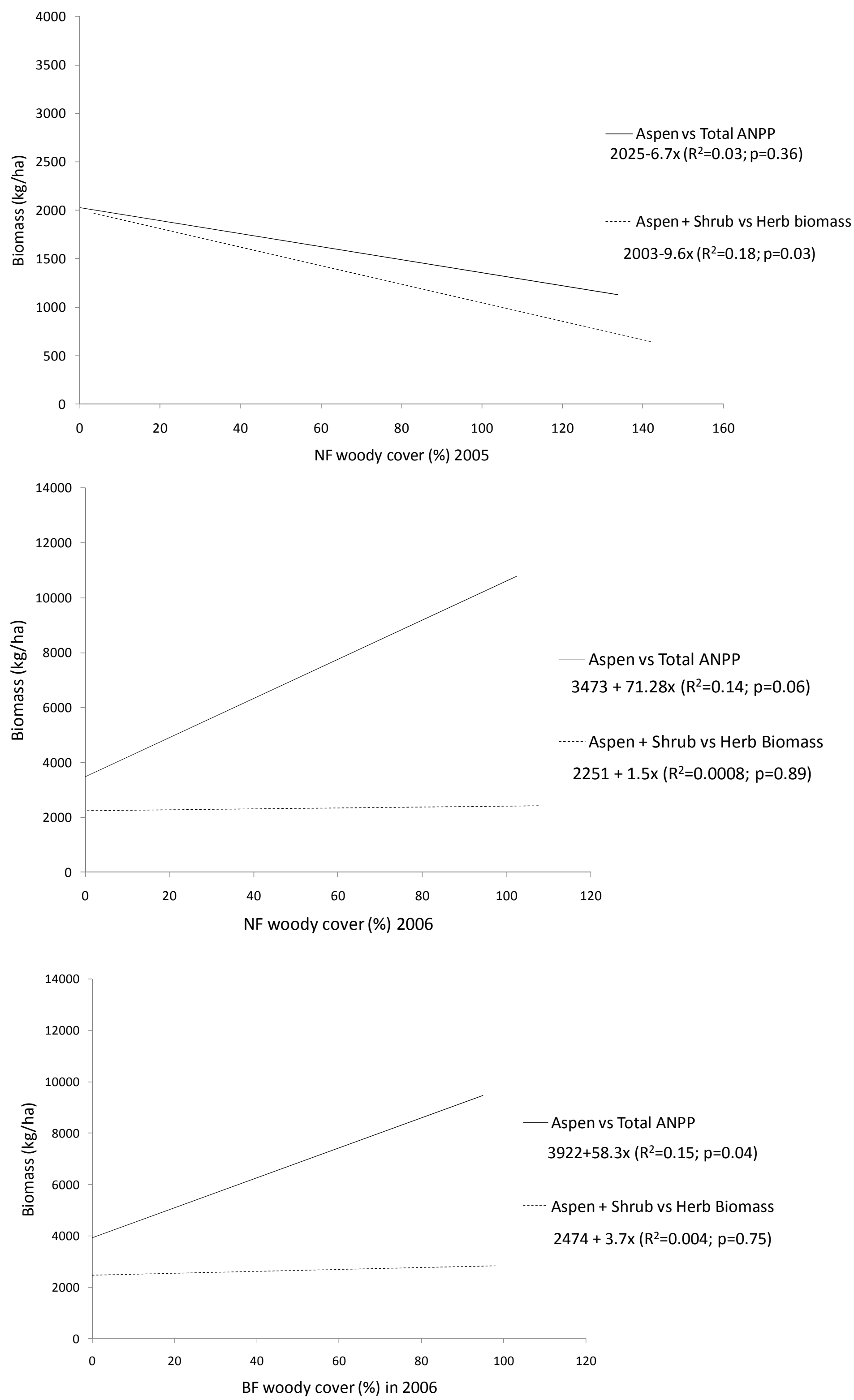
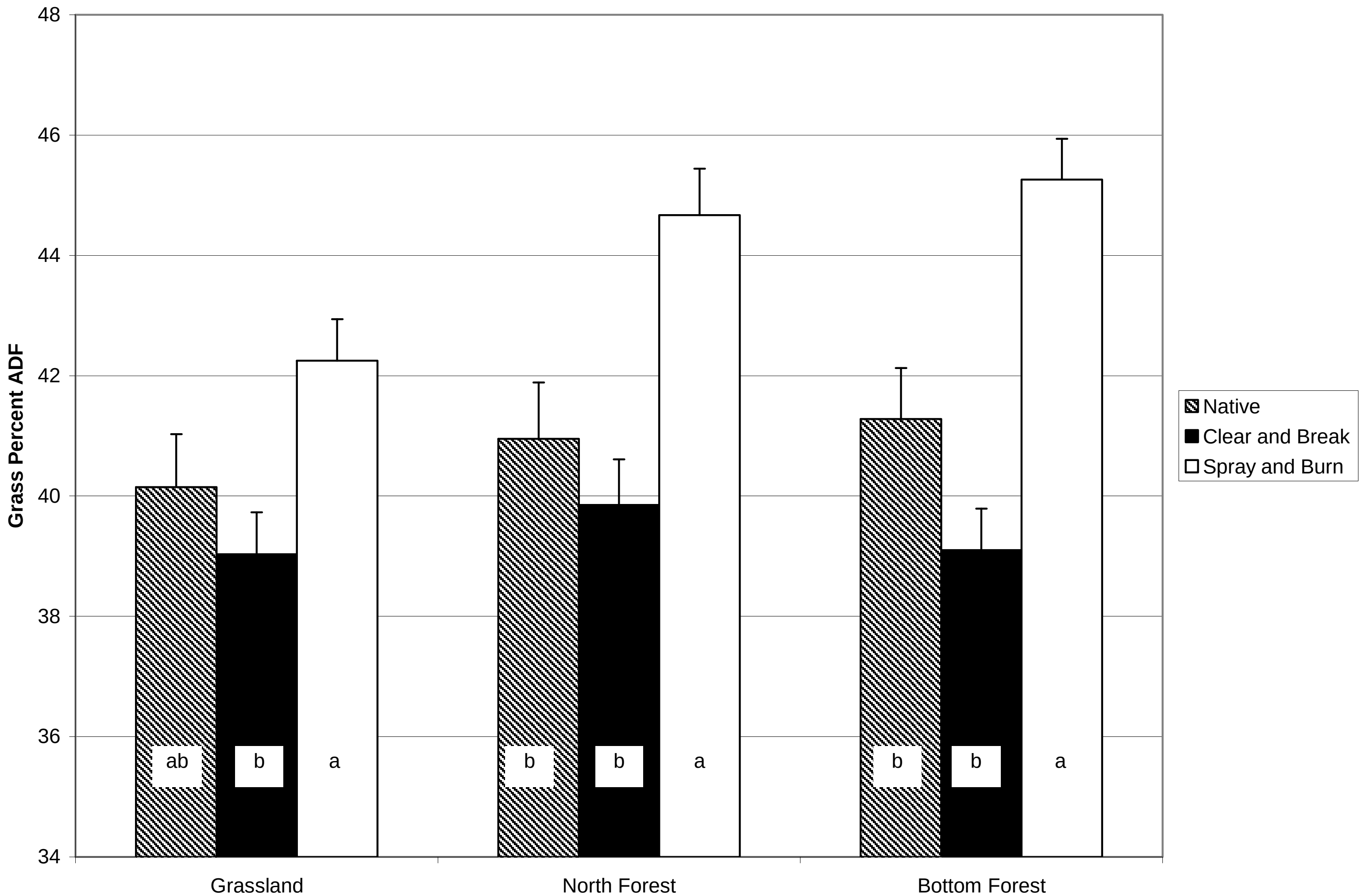


Figure 3-5: Understory biomass in relation to midstory woody canopy cover in the S&B treatments for (A) the north-facing (NF) position during 2005, (B) the NF position during 2006 and (C) the bottom forest (BF) position during 2006.



Grass Percent ADF
48
46
44
42
40
38
36
34
ab
b
a
b
b
a
b
b
a
Grassland
North Forest
Bottom Forest
Native
Clear and Break
Spray and Burn

Figure 3-6: Mean (±SE) concentration of acid detergent fiber (ADF) of grass biomass in 2005-06 within each of 3 aspen management treatments and 3 landscape positions. Within a position, treatments with different letters differ ($P<0.05$).

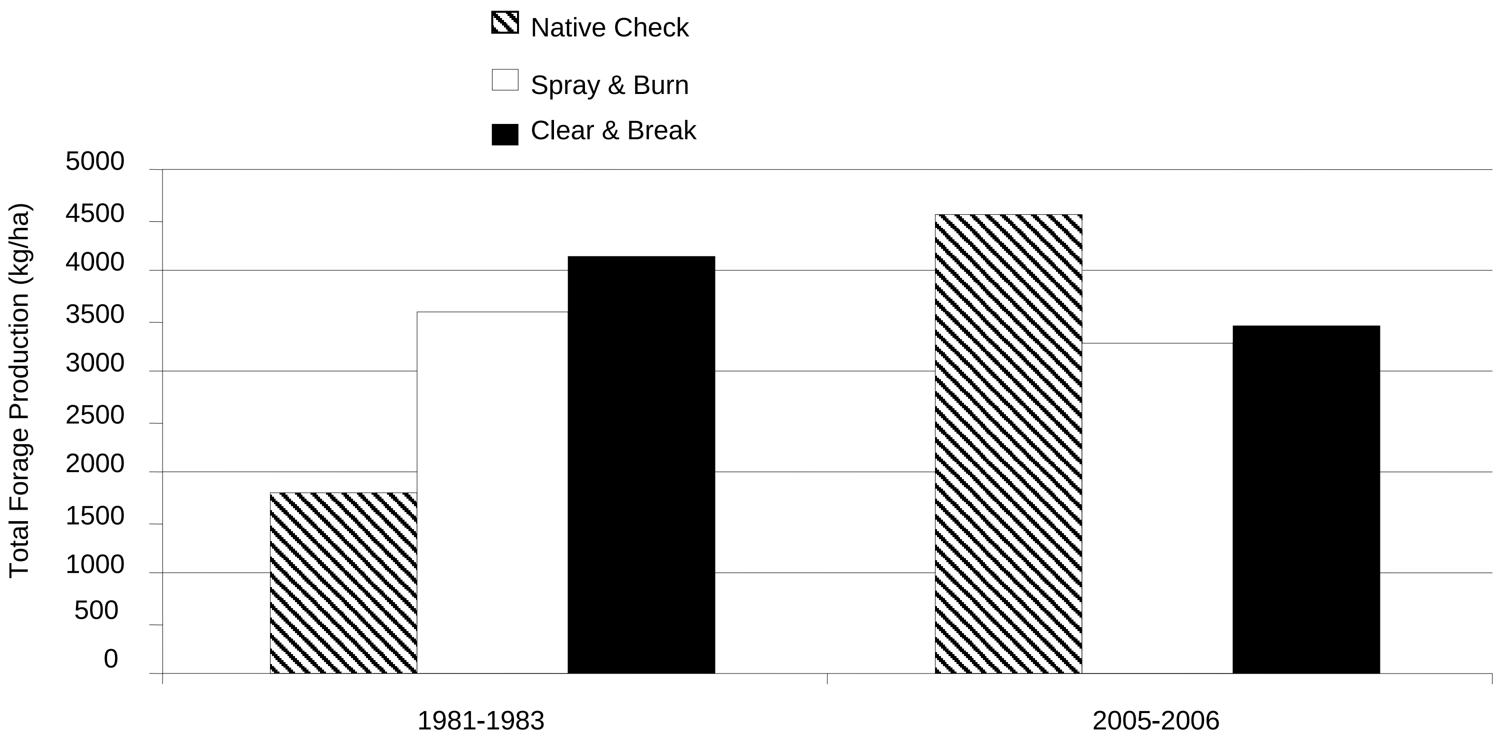


Figure 3-7: Comparison of mean forage production across all landscape positions among the 3 primary aspen management treatments during each of 1981 to 1983, and again in 2005 to 2006. Treatment means in 2005-06 do not differ, $P=0.35$.

4 Do Native and Tame Grassland Soils Differ in Soil Microfaunal Activity?

4.1 Introduction

Within the same climate, soil is the main factor that influences the potential for forage production (Holechek et al. 2003). Soil is a living matrix that forms over time as the result of interactions between parent material, climate, topography, and biota. Soil organisms, including decomposers (grazers, microorganisms, bacteria and fungi) can form complex communities that interact with each other and their environment and contribute to soil development and structure via decomposition, nutrient release and cycling of organic matter (Hamel et al. 2007). These complex communities influence the rate of litter and fine root decomposition and the breakdown of organic matter into inorganic compounds needed for plant uptake. As a result, the activity of soil organisms can be an important indicator of changes in soil quality (Dormaar and Willms 2000a).

Consequently, it is important to understand the impact of management practices on the capacity of soil to sustain plant and animal productivity and its ability to store and utilize water and nutrients (Karlen 1999). In agricultural cropping systems, differences in soil quality have been observed when comparing no-till fields to conventionally tilled fields. For example, conventional cropping systems have been associated with reduced water holding capacity (Moldenhauer et al. 1960), lower organic carbon (Rochette and Angers 1999), higher respiration rates (Parkin et al. 1996) and lower earth worm populations (Curry 1998).

In the majority of Alberta's Parkland, native plant communities have been replaced by agrarian/cultivated communities. Agricultural land modifications, including the use of introduced species in tame pastures and annual cropping systems, have displaced native species, and reduced diversity (Looman and Heinrichs 1973, Wilson 1988, Wilson and Gerry 1995, Christian and Wilson 1999). This in turn, has changed long-term soil structure and dynamics (Dormaar et al. 1990, 1995, Christian and Wilson 1999). Both the cultivation of native vegetation and associated changes in the identity of plant communities can alter ecosystem function (Vitousek 1990), soil carbon (Dormaar et al. 1990, 1995), soil chemistry (Gigon and Rorison 1972, Dormaar and Willms 2000a, Wang et al. 2006, Wu et al. 2006), and organic matter (Dormaar and Willms 2000b).

When native plant communities are converted to cultivated lands there are immediate changes in soil quality, with losses to total carbon and nitrogen. Changes in soil quality are highly influenced by soil mixing and to a lesser extent, by the new plant community (Dormaar and Willms 2000a). Within the Aspen Parkland, the impacts of land conversions on long-term soil health are not well known. Nonetheless, research in the Mixedgrass Prairie to the south has shown that crested wheatgrass (*Agropyon cristatum* (L.) Gaertn.) (Christian and Wilson 1999) and smooth brome (*Bromus inermis* Leyss), both introduced cool-season perennials, have had long-term deleterious effects on soil health including reduced soil nitrogen and carbon (Christian and Wilson 1999).

There is a need to examine soil quality in more northern temperate environments where many native prairies have been replaced with introduced forages to gain a better understanding of potential impacts to soil health. To further assess the environmental impact of converting native grasslands to introduced forages, we compared key soil characteristics associated with native and introduced grasslands at each of 11 sites in the Aspen Parkland of central Alberta, including soil organic matter, organic carbon and biological activity. Do native and tame grassland soils differ in soil microfaunal activity 25 years after wholesale landscape conversion?

4.2 Materials and Methods

4.2.1 Study Area

The study area was located in east central Alberta at the 2,700 ha University of Alberta Kinsella Research Station, situated 150 km SE of Edmonton (53°0'N; 111° 31.2' W). The station is an active ranch that also serves as a center for range ecology and management research. Located in the Aspen Parkland ecoregion at approximately 700 m elevation, the station is comprised of largely intact native vegetation surrounded by a diversity of agricultural land uses, including cultivation of annual crops and perennial forages.

The continental climate of the region is characterized by long cold winters and short warm summers with elevated precipitation. Average annual precipitation is approximately 430 mm, with more than half falling during the growing season (May to August), and peaking in July (Environment Canada 1971-2000). A comparison of mean monthly precipitation and temperatures for the study years (2005-2006 and 1980-83) are summarized in Figures 3-1, 3-2, and Appendices 11-12).

The topography is described as "knob and kettle" due to its undulating landscape of glacial moraine knolls and ridges intermingled with kettle depressions. Area soils are generally classified into three primary orders: Chernozems, Luvisols and Gleysols. Gleysols are typically associated with low-lying topographic positions that experience periodic or sustained saturation. Dark Grey Luvisols and Eluviated Black Chernozems are often associated with forest or shrubland, respectively. In contrast, soils under grasslands on upper slopes are generally Orthic Dark Brown or Black Chernozems (Wheeler 1976), depending on moisture regime. The undulating landscape supports a diversity of vegetation types that provide a variety of grazing opportunities to cattle. Differences in late season grass production of upland grassland sites (2000 kg/ha) and riparian meadows (5520 kg/ha) can be as high as 3000 kg/ha (Asamoah et al. 2004).

Dominant plant communities at Kinsella are representative of the Parkland and form a complex mosaic of aspen (*Populus tremuloides* L.) forest in mesic areas, open grasslands on well drained uplands, ecotonal western snowberry (*Symphoricarpos occidentalis* Hook) and silverberry (*Elaeagnus commutata* Bernh. ex Rydb) shrublands, and either freshwater or saline riparian meadows (Wheeler 1976). Dominant native grass species on upland grasslands include plains rough fescue (*Festuca hallii* (Vasey) Piper), western porcupine grass (*Hesperostipa curtiseta* Hitchc.) and western wheatgrass (*Agropyron smithii* Rydb). Introduced grasses common to the area include smooth brome (*Bromus* *inermis* Leyss), Kentucky bluegrass (*Poa pratensis* L.) and quackgrass (*Agropyron repens* (L.) Gould).

4.2.2 Experimental Design and Sampling

Field sampling was conducted on 11 plots, each of which included native and introduced grasslands as paired subplots. In order to reduce confounding effects of ecosite variability, paired subplots were established on uniform upland range sites: subplots were separated by a fenceline boundary between adjacent introduced grasslands and native grasslands. All plots were within pastures that received regular grazing at moderate stocking rates (~2 AUM/ha annually). Grassland types were considered subplots within each plot. Within each subplot, four soil cores 5 cm wide by 15 cm deep were collected and bulked, and then used for the analysis of soil organic matter (OM).

Additional data were collected within paired subplots using a bait laminae test (Von Törne 1990). The main objective of the bait laminae test was to examine feeding activity from soil organisms and determine how this activity varied with plant community (native vs introduced) and soil depth. Bait laminae have been used successfully in previous investigations in cultivated soils (Hamel et al. 2007) and natural ecosystems (Paulus et al 1999, Geissen and Brümmer 1999) as an indicator of microfaunal abundance. Hamel et al (2007) evaluated the effectiveness of bait laminae and determined that is was a useful tool to assess the feeding activity of macro soil organisms including Collembola (spring tails) and Enchytraeidae (earth worms) responsible for breaking up surface litter.

On 9 May 2006, bait laminae were installed on the 11 paired subplots of adjacent native and introduced grasslands. Within each subplot (n=22), 16 bait laminae strips, each approximately 18 cm long and 6 mm wide, and 1.2 mm in thickness, were inserted into the soil to 10-cm depth in an equidistant 4 x 4 matrix pattern with 10-cm spacing. Each bait laminae strip consisted of 16, 2-mm diameter holes with 5 mm spacing. Holes were filled in advance with a mixture that contained 6.5 grams of cellulose paper, 1.5 grams of agar, 1.0 gram of bentonite clay, 1.0 gram of wheat bran and approximately 25 ml of distilled water. This substrate served as a food source for soil microfauna.

At installation, a knife was used to pre-cut a slot in the soil for the length of the laminae and strips installed to 10-cm depth with the top hole just under the soil surface (approximately 3 mm down). After strips (laminae) were in place the soil at the surface was pinched closed. Field soil moisture content and soil temperature were assessed at the time of installation.

Four additional test sticks at each subplot were installed for preliminary monitoring. Each subplot was monitored bi-weekly using the 4 extra strips to determine the optimal timing of removal of all strips to capture variation in bait laminae readings in relation to vegetation type. All remaining bait laminae strips were subsequently removed on 7 July 2006. Strips were labeled, placed into plastic bags and frozen for later examination. Data were recorded for each bait laminae with readings for each ‘hole’ placed into one of three categories: full removal (greater than 50% of bait missing), partial removal (less than 50% missing) and no removal.

4.2.3 Soil Analysis

Soil OM and carbon were determined using the loss on ignition procedure (Ball 1964). This method provides quantitative oxidation of organic matter, but other soil constituents may be altered or destroyed in the process (Ball 1964). Soil organic carbon was determined using standard analytical techniques (AOAC 1995).

4.2.4 Statistical Analysis

Soil OM and carbon data from upland native and introduced (i.e. converted) grasslands were analyzed using a pair-wise t-test of plots to determine potentially significant differences between the soils of adjacent landscape treatments.

Bait laminae strips were first examined to determine the proportion of holes with bait fully intact, partially removed or fully removed. In addition, these data were obtained separately for the top and bottom half of each strip, representing the shallow (0-5 cm) and deep (5-10 cm) soil depths. Summary frequency data were then analyzed for the effects of grassland type (native vs introduced), soil depth, and their interaction, using a non-parametric contingency (i.e. chi-square) test with the CATMOD procedure in SAS (SAS Institute Inc. 2003).

4.3 Results

Results of the bait laminae fence-line comparison between native and introduced (C&B) pastures indicated differences in each of the main effects evaluated (Table 4-1). Total feeding activity varied among vegetation types, with approximately 5% greater removal in the native grassland, largely due to an increase in the number of laminae experiencing partial removal (Figure 4-1). Bait laminae removal also differed between the two soil depths shallow (0-5cm) and deep (5-10cm) ($p<0.01$). Full and partial removal of laminae in the shallow soil layer were 20.0% and 11.0% of all observations, while full and partial removal levels in the deep soil layer had an opposite trend of 5.5% and 17.7%, respectively. Thus, a total of 31.0% of bait laminae in the shallow soil layer experienced some degree of removal, which decreased to 23.3% deeper down (Table 4-2).

A treatment by soil depth interaction was also observed in the pattern of bait laminae removed ($p=0.03$). While the proportion of bait laminae experiencing some degree of removal in the shallow soil layer remained similar between native (32.0%) and introduced (30.5%) vegetation types, the native (26.5%) experienced greater ($p<0.03$) removal than the introduced vegetation (20.0%) at the deeper soil depth (Figure 4-2).

Although we hypothesized that soil organic matter and carbon differences may be evident between vegetation types, this did not occur. Soils in the top 15cm of native and introduced grasslands had 12.9% and 13.9% organic matter, respectively, but remained comparable ($p=0.85$). Similarly, soil carbon concentrations on native and introduced grasslands were 6.0% and 5.7%, respectively ($p=0.76$). Thus, no obvious differences in key soil properties relating to nutrient cycling and energy flow were apparent.

4.4 Discussion

Although significant differences in soil organic matter and carbon were not observed between native and introduced vegetation types, these results could be attributed to a small sample size (i.e. number of sampling ‘pairs’ along fencelines), coupled with high soil and vegetational heterogeneity, which is known to make the detection of treatment effects of soil characteristics difficult (Jobbágy and Jackson 2000). Previous studies have focused on more arid regions (Voroney et al. 1981, Dormaar and Willms 2000a), where tame species may have lower belowground carbon inputs from roots when compared to native grasses, whereas this study examines a more mesic northern temperate ecology where the introduced forages may maintain OM and C through similar contributions of litter and root biomass to the soil as the native grasslands in this area. Moreover, loss of soil organic matter and N as a result of cultivation has been associated with changes to the biological and physical processes of the soil (Voroney et al. 1981, Dormaar and Willms 2000a), leading to poorer soil ‘health’ and associated ecosystem function, including forage productivity. If future management practices for the C&B treatment were to include frequent pasture renovation (i.e. cultivation), soil OM and C may continue to be depleted (Campbell et al. 1976, Voroney et al. 1981) with the greatest losses associated with the first 5-10 yrs of treatment (Caldwell et al. 1939, Martel and Paul 1974).

Although Hamel et al. (2007) evaluated the effectiveness of using the bait-lamina test to assess microfaunal feeding activity in the Mixedgrass Prairie; the current study provides a unique assessment of soil faunal activity in northern temperate grasslands of the Aspen Parkland. Assuming the observed trends in bait laminae removal reflects microfaunal activity, this research suggests greater microfaunal populations and/or activity occurred within native grasslands. Several explanations may account for the observed differences in bait laminae removal between vegetation types, including differences in soil bulk density, arising from greater compaction on introduced grasslands, in part due to previous cultivation (Pennock et al. 1994), or the shallow-rooted morphology of introduced grasses (Peterson et al. 1979), which may limit microfaunal activity in this vegetation to the shallower rhizosphere (Christian and Wilson 1999). Given that the physical disturbance associated with introduced grasslands in the C&B treatment were over 25 years old, the lower microfaunal activity within these communities may also be due to the associated lower diversity of vegetation found there, particularly of native species, which in turn, may lead to a simpler trophic community (St. John et al. 2006), or those bacteria and fungi that favour the shallow soil profile (van Eekeren et al. 2008).

Changes in soil microfaunal activity, including across soil depths, may have implications for rangeland conservation and productivity in the future, including for example, differential carbon sequestration within soils under native and introduced vegetation. Reductions in soil organic matter and associated carbon have been linked to wholesale conversion of native Dry Mixedgrass Prairie (MGP) to introduced forages such as crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) in the past (Smoliak and Dormaar 1985, Dormaar et al. 1995, Christian and Wilson 1999, Willms et al. 1999), and have raised the possibility that while introduced grasslands are useful for providing forage (Kilcher and Looman 1983, Lawrence and Ratzlaff 1989, Asay et al. 2001), they may not be conducive to the maintenance of key ecosystem processes necessary to maintain long-term pasture condition and productivity. Sites dominated by introduced species in the MGP have not only differed in soil chemistry, but also demonstrated reduced water holding capacity through decreased infiltration and higher levels of runoff (Murphy et al. 2008).

Although the negative ecological implications of the introduction of crested wheatgrass to North American grasslands have been well documented (Love 1932, Dormaar et al. 1995, Dormaar et al. 1979, Eissenstat and Caldwell 1988, Christian and Wilson 1999, Whalen et al. 2003), the ecological influence of invasion by smooth brome, Kentucky bluegrass, and associated species, is less understood. Nonetheless, other research suggests that smooth brome may be dependent on soil biota to aid in self-facilitation, and that smooth brome may condition the soil by creating a potential hostile environment for native species (Jordan et al. 2007). Although the current study found that smooth brome successfully established within the Native Check (NC) treatment, suggesting that these native communities were unable to prevent smooth brome invasion; native species also continued to be found growing within the introduced grasslands, suggesting that introduced species were not able to fully exclude natives. Moreover, this ‘homogenization’ of plant species between native and introduced treatments may have limited the ability to find treatment-based differences in soil characteristics and/or microfaunal activity.

Further interpretation of the results of the bait-laminae test in the current study is limited given that specific information on the stocking rates and/or other grazing management practices (i.e. timing) associated with each pasture were not available for the years leading up to the study. Thus, the full implications of any differences in soil conditions among vegetation types, including microfaunal activity, remain unknown at this time, but merit further investigation.

4.5 Conclusion

Soil organisms play an integral role in the success of agricultural production systems. This research suggests that while the replacement of native grassland with introduced forages has not changed soil OM or carbon, changes in vegetation may be associated with subtle changes in soil microfaunal activity. Thus, historical land management practices have the potential to alter long-term ecosystem function, even in northern temperate grasslands. As microfaunal abundance and activity is known to help facilitate plant growth, this ecosystem attribute is considered an important indicator of ecosystem health. Today, with increased demands on our agricultural production systems, it is important to employ agricultural practices that sustain soil quality and promote soil health through the enhancement of soil biological activity, which in turn, may be optimized by the maintenance of native grassland communities.

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Table 4-1: Summary of bait laminae Chi-square analysis results assessing differences in soil microfaunal feeding activity in each vegetation type and at 2 soil depths.

Response	Deg Freedom	Chi-Square (X^2)	P-Value
Vegetation Type	2	15.12	0.0005
Plot (Treatment)	40	480.25	<0.01
Soil Depth	2	67.09	<0.01
Vegetation x Soil Depth	2	6.81	0.03

Table 4-2: Comparison of bait laminae exhibiting no, partial, full, and partial or full removal. Bait laminae removal levels differ between native and introduced vegetation based on a Chi-square test, p<0.001.

Response	Location	P value	No Removal	Partial Removal	Full Removal
Landscape Treatment	Native	<0.0005	35.2	10.4	4
	Introduced		37.7	8.5	4.3
Soil Depth	Top	<<0.01	34.5 b	10 a	5.51 a
	Bottom		38.4 a	8.9 b	2.74 b
Treatment x Soil Depth	Native Top	0.03	23.3 a	28.2 a	30.1 b
	Introduced Top		24 a	24.8 a	36.7 a
	Native Bottom		25 a	26.8 a	18.3 a
	Introduced Bottom		27.7 a	20.3 a	14.9 b

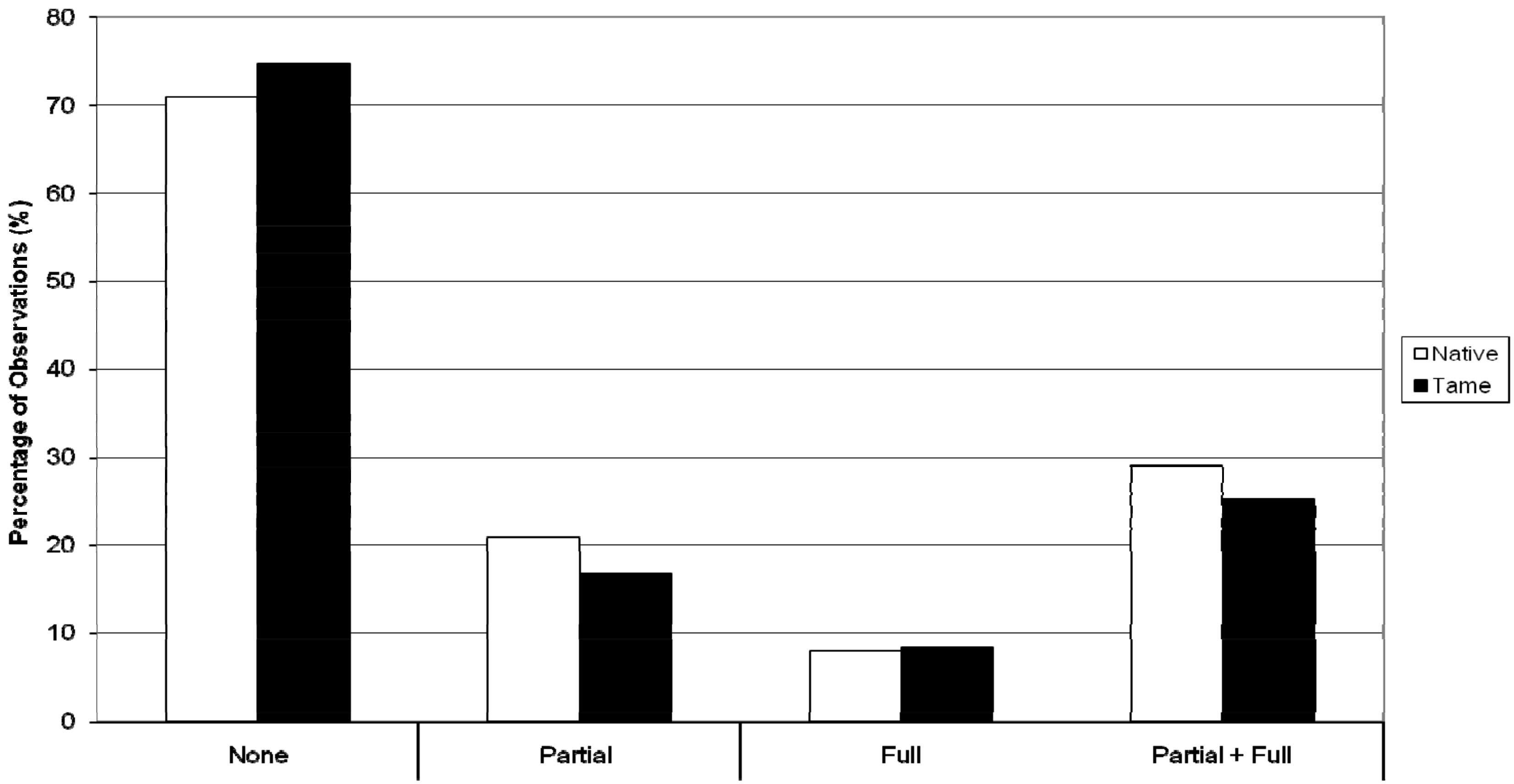

Figure 4-1: Differences in total feeding activity of soil fauna between native and introduced pastures using bait laminae, including partial and full removal of bait ($p<0.05$).

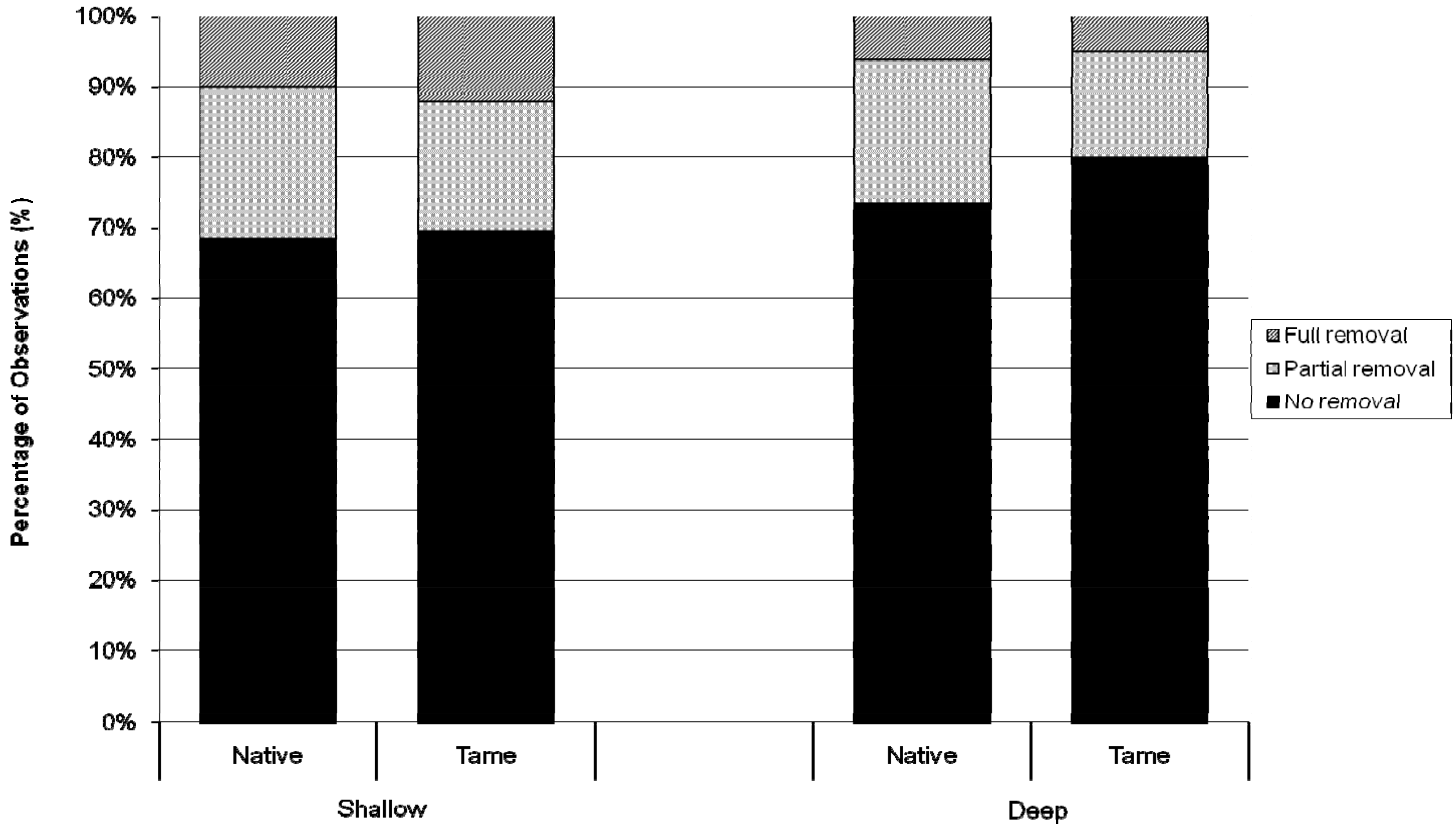

Figure 4-2: Comparison of bait laminae exhibiting either partial or full removal within soil under adjacent native and introduced vegetation. Data are further stratified by soil depth (left: 0-5 cm; right: 5-10 cm).

5 Does Aspen Control Pay in the Long-Term?: An NPV Analysis of Intensive and Extensive Aspen Management Treatments in the Parkland

5.1 Introduction

Profits associated with cow calf operations are dependent on the sale of market weight calves (Miller 2002). It is a common belief that tree and shrub encroachment in the Aspen Parkland leads to a decreased capacity to produce herbage (Bailey and Wroe 1974, Wheeler 1976), which can result in a decrease in the economic potential for cattle production systems (Osborn and Witkowski 1974, Schumann et al. 2001, Teague et al. 2001). Net Present Value models attempt to predict future outcomes of potential investment projects through the analysis of cash flows over time. A project is considered to be economically feasible if the benefits outweigh the costs. However, the project that promises the highest rate of return with a relatively low degree of risk would generally provide the investor with the most amount of wealth, hence the preferred investment (Workman and Tanaka 1991).

The purpose of this research is to compare and contrast the NPV and economic feasibility of three brush management strategies available to producers operating in the Aspen Parkland. These strategies represent contrasting philosophies on the degree of capital inputs necessary to control aspen and restore rangeland productivity, and thereby optimize wealth.

5.2 Materials and Methods

5.2.1 Study Area

The study area was located in east central Alberta at the 2,700 ha University of Alberta Kinsella Research Station, situated 150 km SE of Edmonton (53°0'N; 111° 31.2' W). The station is an active ranch that also serves as the center for range ecology and management research. Located in the Aspen Parkland natural subregion at approximately 700 m elevation, the station is comprised of largely intact native vegetation surrounded by a diversity of agricultural land uses, including abundant cultivation.

The continental climate of the region is characterized by long cold winters and short warm summers with elevated precipitation in summer. Average annual precipitation is approximately 430 mm, with more than half occurring during the growing season (May to August), and peaking in July (Environment Canada 1971-2000). A comparison of mean monthly precipitation and temperatures for the study years (2005-2006 and 1980-83) are summarized in Figures 3-1, 3-2, and Appendices 11-12).

The topography is described as "knob and kettle" due to its undulating landscape of glacial moraine knolls and ridges intermingled with depressions. Area soils are generally classified into three primary orders: Chernozems, Luvisols and Gleysols. Gleysols are typically associated with low-lying topographic positions that experience periodic or sustained saturation. Dark Grey Luvisols and Eluviated Black Chernozems are often associated with forest or shrubland, respectively. In contrast, soils under the grasslands on upper slopes are generally Orthic Dark Brown or Black Chernozems (Wheeler 1976). The undulating landscape supports a diversity of range types that provide a variety of grazing opportunities to cattle. Differences in late season grass production of upland grassland sites (2000 kg/ha) and riparian meadows (5520 kg/ha) can be as high as 3000 kg/ha (Asamoah et al. 2004).

Dominant plant communities at Kinsella are representative of the Parkland and form a complex mosaic of aspen (*Populus tremuloides* L.) forest in mesic areas, open grasslands on well drained uplands, ecotonal western snowberry (*Symphoricarpos occidentalis* Hook) and silverberry (*Elaeagnus commutata* Bernh. ex Rydb) shrublands, and either freshwater or saline riparian meadows (Wheeler 1976). Common native grass species on upland grasslands include plains rough fescue (*Festuca hallii* (Vasey) Piper), western porcupine grass (*Hesperostipa curtiseta* Hitchc.) and western wheatgrass (*Agropyron smithii* Rydb). Introduced grasses common to the area include smooth brome (*Bromus inermis* Leyss), Kentucky bluegrass (*Poa pratensis* L.), and quackgrass (*Agropyron repens* (L.) Gould).

5.2.2 Project Background and Experimental Design

In the past, rangeland conversion was based on the perception that native rangeland landscapes are less productive and therefore less profitable than lands converted into agronomic-based production systems. However, the environmental and economic risks associated with converting native Parkland rangeland into tame forage remain unclear. From 1979 to 1984, the Alberta

Agricultural Research Institute Farming for the Future program funded research to compare three aspen management strategies at the University of Alberta, Kinsella Research Station. The original landscape study consisted of 8 adjacent pastures, each approximately 40 acres in size, radiating outward from a central feeding and watering area. Forage production was compared among three management treatments across three topographic positions (grasslands, north-facing forests and bottom forests). Treatments were designed to contrast strategies for increasing livestock carrying capacity in the Aspen Parkland through both intensive and extensive aspen forest control. The first management strategy was an intensive conventional Clear and Break (C&B) treatment, and the alternative, a less intensive low-cost experimental Spray and Burn (S&B) treatment. The third treatment was essentially a Native Check (NC) with no aspen control measures undertaken. The forage production data (1980-83 and 2005 and 2006) collected would then be used to compare costs and returns of the two aspen management strategies in a cost-benefit analysis. However, due to a lack of funding this portion of the project was never seen to completion.

5.2.2.1 Clear and Break Treatment

The Clear and Break (n=3 replicates) treatment is a commonly practiced intensive land conversion from native rangeland into tame pasture involving the mechanical clearing (i.e. dozing) of woody vegetation (aspen forest and shrublands) followed by sod breaking and seeding to introduced tame forages. Clearing of two of the three C&B replicates was done in March of 1979. Later that same year the paddocks were broken twice using a Kelo belt offset disc and once again in the spring of 1980. Tame forages were then seeded (1980) with a press drill to a mixture of Magna smooth brome (5.2 lb/ac), Boreal creeping red fescue (2.6 lb/ac) and dryland alfalfa (1.5 lb/ac). No grazing was permitted in the establishment year of 1980. Due to equipment problems and delays the third replicate was cleared in the winter of 1979/80, broken and disced in the summer of 1980 but was not seeded until the autumn of 1980 (Bailey et al. 1984). Breaking and re-seeding of pastures on a 5 to 10 year cycle, is one of the most commonly used rejuvenation methods in the Aspen Parkland of western Canada as over time the carrying capacity of theses pastures declines (Lardner et al. 2001). However, at least one year of forage production can be lost to seedling establishment and if moisture conditions are not favorable it can take up to two years for re-establishment of the forage base.

5.2.2.2 Spray and Burn Treatment

The Spray and Burn (n= 3 replicates) treatment is a less intensive treatment used to reverse aspen encroachment which involved an initial aerial herbicide application of 2,4-D (intended to open up the canopy to facilitate burning), followed by prescribed burning and broadcast seeding of forests. Firebreaks and fencelines were cleared between March 1979 and March 1980. The project area was sprayed with 46 ounces per acre of 2,4-D butyl ester mixed with 7.3 gallons/acre of water using a helicopter in mid June 1980. The three replicates of the S&B were treated with a prescribed burn on April 22nd 1981.

On April 30th 1981 a helicopter was used to broadcast seed Kay orchard grass (4.9 lb/ac), magna smooth brome (4.9 lb/ac), creeping red fescue (2.6 lb/ac) and drylander alfalfa (2.2 lb/ac) (Bailey et al. 1984).

5.2.2.3 Native Check

It is important to note that in the early 1990's the Native Check pastures from the original study were treated with a low intensity prescribed burn (Irving 2006). Although noteworthy this incident of fire is not unlike pre settlement fire return intervals in the Aspen Parkland of 3-15 years (Kasischke and Stocks 2000).

5.2.3 Experimental Design

Following the implementation of landscape treatments in 1979-80, initial sampling of vegetation between 1981 and 1983 was confined to the assessment of current annual peak biomass (late July/early August) for each of 240 plots distributed throughout the study area, with 10 plots in each of three topographic positions (grassland, bottom forest and north-facing forest) in each of the 8 paddocks. The three topographic positions were assumed to be equally represented across the landscape. Samples were sorted into vegetation components, including herbage (i.e. grasses, sedges and forbs) and browse (i.e. current annual growth of shrubs, and aspen saplings less than 2 m tall), dried and weighed. Archived biomass data from immediately after the landscape treatments were established (1981-1983 inclusive) and again 20+ years after treatment (2005-2006, inclusive) were used as the basis for the comparative economic assessment between treatments. Two separate models were developed for each of the three treatments, the first to examine the implications of costs associated with the original treatments (1979-80), and the second to examine differences in outcome if the decision were made to proceed with this project in today's market.

5.2.4 Empirical Framework

Economic assessment was undertaken using net present value (NPV) models that incorporated the financial benefits/costs associated with changes in forage availability under different land management regimes, and the cost associated with undertaking each one. A similar procedure has been used in modeling the economics associated with improvements in lotic riparian area management in southern Alberta (Miller 2002, Unterschultz et al. 2004). The NPV models were developed to examine the economic feasibility and potential wealth of each management strategy. Static NPV modeling estimates a series of expected cash flows overtime but does not allow for managerial or design flexibility (Sullivan et al. 1999). The NPV analysis uses project specific risk premia (discount rate) to estimate the present value of all future cash flows (Boardman et al. 1996) and the risk premium is measured by the potential for deviation from the expected returns (variance) (Bauer 1997). The NPV model predicts annual quantitative costs and benefits over the life span of a project and attaches a dollar value to those impacts. The quantitative data required to conduct an NPV assessment include the initial cost of the project (C_0), the life (time span) of the project (T), and the expected cash flow in each period (C_1…….. C_T)

(Unterschultz and Quagrainie 1996). The final equation for NPV assessment is as follows:

$$NPV = C_0 + \sum_{t=1}^{T} \frac{C_t}{(1+r)^t}$$

(Equation 5.1)

The basic decision rule is based on the total present value of benefits and the total present value of costs. If benefits exceed costs then the decision would be to proceed with the project. However in this case the NPV of three management strategies were compared to determine which treatment has the highest expected NPV based on established predictions and valuations.

Results of the static NPV models compare potential scenarios where the highest NPV reflects superior economic benefits to the rancher. This assessment compares the potential for each treatment to support livestock grazing based on the short and long-term forage yields observed. Additionally, the analysis compares potential livestock use of herbage (grasses and forbs) only with the additional inclusion of browse (i.e. shrub current annual growth). Model values are discounted and do not represent profitability of cattle ranching, but rather are a means to compare treatments.

5.2.4.1 Project Risk

The discount rate of 8% is an approximate average based on two previous economic studies of cow/calf operations in Alberta (Bauer 1997, Miller 2002). The first study centered around 20 years of data collection in Stavely, Alberta, and calculated a real rate of return for a Southern Alberta cow/calf operation of 5.6% with a standard deviation of 19.4% (Bauer 1997). The second study, utilized Bauer's calculated standard deviation as a measure of risk in the Capital Market Line (CML) to estimate a discount rate of 12.25% for cow/calf operations (Miller 2002).

5.2.5 Statistical Analysis

For each of the three treatments (NC, C&B and S&B), linear regression analysis for the existing herbage (Figure 5-1) and total forage production (Figure 5-2) datasets was undertaken using the Regression Data Analysis option in Microsoft Excel. Statistically derived missing values (1984-2004) were calculated using the least squares method to calculate a straight line that best fits the known data points (Appendices 9 and 10). Therefore changes in production were assumed to represent a straight line relationship from 1981 through 2006.

5.2.6 NPV Model Assumptions

The primary assumption of the NPV models states that outside of the differences in initial input costs (Table 5-1) and the annual differences in output based on different production (kg/ha) levels (Appendices 9 and 10), all expenses remain equivalent. Therefore the final NPV does not account for annual operational costs as they are assumed to be equal in each pasture management strategy. The analysis was also done on a before-tax basis.

Estimates for land clearing, breaking, seeding, and prescribed burning and dollars per Animal Unit Month are from Alberta Agriculture, "Custom Rates

Survey” (AAFRD 1980 and 2005). Where unavailable in the literature (e.g. local costs associated with prescribed burning), expert opinion was employed to obtain cost information (Irving 2006**).**

Base case assumptions including costs of C&B versus S&B, annual forage utilization, and discount rate are outlined in Table 5-1. For example, 50% sustainable use of forage is assumed within all landscape treatments with the exception of the S&B, which received 70% use during the first 4 years (Table 5-1) to achieve biological control of aspen sucker regrowth (Fitzgerald and Bailey 1984). Finally the number of delay years in the C&B changes in some of the sensitivity analysis to allow for seedling establishment of the seeded agronomic grasses. In each scenario, all of the costs are at the beginning of the time horizon and then the subsequent cash flows are strictly benefits generated by potential forage production.

5.2.6.1 Sensitivity Analysis

Sensitivity analysis allows for a closer examination of the uncertainty within future predictions. As the decision to invest is based on our predictions of what may happen, there is inevitably a degree of uncertainty associated with those predictions (Boardman et al. 1996). A number of sensitivity analyses were performed where one single variable was changed while holding all others constant. This process determines the level of uncertainty associated with the model assumptions and helps depict how sensitive the analysis is to potential change. In this study we manipulated a number of variables, including fluctuations in the potential custom grazing rate of forage per Animal Unit Month (AUM), the allowable proportion of sustainable forage utilization, differences between annual net primary production (total ANPP, including browse) and herbage (i.e. grass and forb only) production, grazing delays in the C&B, and differences between historical and current conversion costs.

5.3 NPV Analysis Results and Discussion

Figure 5-1 graphs the pattern in forage yields for each scenario. Two scenarios are provided, including the comparative return on investment given the linear change in forage from 1981 through 2006 (Table 5-2), as well as a comparison of the projected returns going forward associated with newly initiated landscape treatments in 2007 (Table 5-3), assuming those trends in forage return are similar to those obtained from 1981 to 2006. Retrospective analysis (1979 costs) of the period 1981 to 2006 at $13.00/AUM (Table 5-2) indicated that when browse was included in the amount of usable forage, the greatest NPV was associated with the S&B landscape treatment, followed by the Native Check, and then the C&B (1 year delay): the former treatments are $14,778 and $4,999 greater than the C&B, respectively. Notably, this ranking of treatments occurred despite the S&B having lower biomass production in 2005 and 2006. However, it is also clear that much of the NPV generated from these treatments is obtained from the abundant browse situated in the forest communities of both the S&B and native treatments. When browse is excluded as usable forage in the analysis and all other variables held constant, the C&B treatment ($25,062) was marginally better than the native ($23,366) and S&B ($21,918) treatments. In this scenario, all three treatments remained economically feasible, yet the option with the highest potential for wealth fluctuated with the value (i.e. potential return) of an AUM (Table 5-2). Thus, while those landscapes containing a diverse mix of habitats, including forest, have the potential for high livestock use and production, assuming that all shrub species are palatable, this depends heavily on effectively utilizing the wooded habitats, primarily forest, and their associated browse, much of which is greater in quality than the others, as noted earlier in Chapter 3.

Early comparisons on the effectiveness of fire, grazing, and mechanical clearing suggested that an integrated approach using aerial herbicide application, fire, and broadcast seeding could be used to increase forage production at one-third the cost of traditional mechanical clearing (Bailey 1986). Similarly, when 2006-07 costs were used and various scenarios were examined in the base case analysis to compare the NPV of each treatment using annual net primary production, the S&B treatment was consistently the most economically profitable, followed by the NC, and finally the C&B (Table 5-3). In contrast, when using only herbage the most feasible option was more often the NC. The C&B became the most feasible only when the value of an AUM reached $25.00 (Table 5-3).

So why do landowners continue to use conventional land clearing and pasture refurbishment? Any number of social factors may play a role in the management of private lands from a sense of place (Wester-Herber 2004) to religion (Miller and Luloff 1981) to cultural background (Salamon 1985). Research suggests that when a farmer is connected with the previous generation their farming practices are anchored in the traditional methods, which limits the potential for innovation (Bennett and Kohl 1963). Traditionally, 20th century farmers considered natural landscapes to be a hindrance to crop production. Nonetheless, knowledge of fire behavior and specific training can be required to manage a successful prescribed burn program.

In addition, our analysis is constrained by the assumptions outlined in Table 5-1, and changes in these assumptions will change the analysis outcome. For example, we assumed a 50% sustainable use of forage within all landscape treatments with the exception of the S&B, which received 70% use during the first 4 years to achieve biological control of aspen sucker regrowth (Fitzgerald and Bailey 1984). While the 50% level of forage removal appears reasonable for native pastures, this may be too conservative for tame pastures, which can handle greater levels of use due to their grazing tolerance. Note that at $20.00 per AUM and when acceptable target use levels increased to 60% in the C&B, NPV values for the period 2007 to 2031 rose markedly from $36,280 to $52,197 for herbage use alone, with a break-even point relative to the Native Check area of 54% utilization ($39,986) (Table 5-4). Similar results were observed at $10.00 per AUM for the period of 1980 to 2004 (Table 5-5). Thus, only small increases in expected forage use can lead to the C&B remaining economically competitive in NPV returns relative to the other landscape treatments. Conversely, recent and continued increases in the costs associated with land conversion, including the cost of labour, fuel and equipment will also necessitate re-evaluation of NPV to ensure the most accurate assessment of the real costs and benefits associated with this practice of aspen management.

While this analysis provides a reasonable comparison of the landscape treatments implemented in 1981, there are also a number of important limitations associated with this analysis. For example, forage yields were projected using a linear trend from 1981 through 2006, and were based on five data points, including three years in the early period and two later on. Thus, the observed yield data from 2005 and 2006 have the potential to markedly change the outcome of this economic analysis, and are further subject to variation in growing conditions, including the relatively dry conditions observed in 2005. Although forage quality was not factored into this analysis, as forbs and shrubs were generally of greater quality (Chapter 3), and were common in the Native and S&B treatments, the inclusion of quality may have further separated the NPV differences among treatments. Additionally, interpretation of NPV values from the NC must be tempered by the fact that this area had experienced a prescribed fire during the interim, which may have increased forage production through more recent aspen removal, and thus led to greater than otherwise expected NPV. This is supported to some extent by observations that herbage and browse production were 68% and 56% lower, respectively, within an independent unburned area (Field 9A: herbage = 1084 kg/ha; total = 2173 kg/ha) outside of the study area, although it is remains unclear whether these differences were due to changes in previous disturbance regime or simply changes in growing (i.e. ecosite) conditions between locations.

Regardless of the disturbance regime, in retrospect there are a number of ecological benefits of maintaining native Aspen Parkland rangelands that this economic analysis also fails to capture. Most importantly perhaps, the long-term productive potential of any agricultural system is fundamentally dependent on the quality of its soil and water (Jones 1996). Biodiversity, nutrient cycling, soil retention and the aesthetic value of the natural landscape all have inherent social and economic values that should not be overlooked.

5.4 Conclusion

It has been a common perception that pasture renovation, including the seeding of introduced forages, leads to an increase in the productive capacity of the land base. Yet what is often overlooked is the less intensive landscape methods used to promote forage production. Although the NPV for the C&B treatment in most scenarios remained economically feasible, this method of production generally provided the least amount of wealth and in todays global market it is important to maximize efficiencies to maximize your rate of return on investment.

As a rangeland manager and as a business entity it is important to evaluate potential projects and compare them to proposed alternatives. Identify what the advantages and disadvantages of each project are and what potential outcomes to expect. This study compared the NPV of a Native Check, a Spray and Burn and a Clear and Break treatment in specific scenarios, and any flexibility in the key assumptions or land management has not been accounted for.

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Table 5-1: Base-case assumptions associated with the NPV analysis of the Clear & Break and Spray & Burn treatments, including costs.

Treatment	Treatment Costs 1980 ($/ac)	Treatment Costs 2006/07 ($/ac)	Other Assumptions
Clear and Break	155.0	286.0	
Forage Utilization			50% use of production
Spray and Burn	60.0	93.0	
Forage Utilization			
1st 4 years			70% use of production
5th year and beyond			50% use of production
Native Check			
Treatment Area	n/a	n/a	160 acres
Forage Utilization			50% use of production
General Model Assumptions			
Treatment Area			160 acres
NPV Discount Base Rate			8%
Grazing Season			5 months
Animal Unit Month (AUM)			273 kg of forage

Table 5-2: Base case analysis comparison between the NPV generated from all forage (i.e. ANPP) and herbage only from each of 3 treatments, including the impact of varying $/AUM and changes in the duration of delay in economic production in the Clear and Break (1 or 2 years). Analysis uses *Bailey's 1984 costs* with an 8% discount rate.

Value of Forage ($/AUM)	Native Range	Clear & Break 1 Year	Clear & Break 2 Year	Spray & Burn
ANPP				
$7/AUM	$24,945	$11,493	$8,777	$27,678
$10/AUM	$35,635	$26,054	$22,174	$43,167
$13/AUM	$46,326	$40,615	$35,571	$58,657
Herbage				
$7/AUM	$15,297	$5,461	$3,135	$11,228
$10/AUM	$21,853	$17,436	$14,114	$19,667
$13/AUM	$28,408	$29,412	$25,093	$28,107

Table 5-3: Base case analysis comparison between the NPV generated from all forage (i.e. ANPP) and herbage only from each of 3 treatments, including the impact of varying $/AUM, and changes in the duration of economic production in the Clear and Break (1 or 2 years). Costs use 2006-07 data with an 8% discount rate.

Value of Forage ($/AUM)	Native Range	Clear & Break 1 Year	Clear & Break 2 Year	Spray & Burn
ANPP				
$15/AUM	$53,453	$30,738	$24,918	$64,193
$20/AUM	$71,270	$55,006	$47,246	$90,009
$25/AUM	$89,088	$79,275	$69,575	$115,825
Herbage				
$15/AUM	$32,779	$17,811	$12,827	$28,943
$20/AUM	$43,705	$37,771	$31,126	$43,009
$25/AUM	$54,632	$57,730	$49,424	$57,075

Table 5-4: Changes in NPV associated with variation in forage use within the Clear and Break and Spray and Burn treatments, including when use is derived from either all forage (i.e. ANPP) and herbage only. Comparisons are done using a base forage value of $20/AUM with changes in the duration of economic productions in the C&B (i.e. 1 or 2 years). Analysis uses 2006-07 costs with an 8% Discount Rate.

Forage Utilization and Treatment	$20/AUM
ANPP	
Native Range 50% forage use	$71,270
Clear and Break 1 year	
CB 50% use	$55,006
CB 60% use	$74,421
CB Break Even: BE= 58%,	$71,270
Clear and Break 2 year	
CB 50% use	$47,246
CB 60% use	$65,109
CB Break Even = 63%	$71,270
Spray and Burn 70% 1st 4 yrs	
70% use, then 50	$90,009
70% use, then 60	$103,902
70% use; BE= 37%	$71,270
Herbage	
Native Range 50% forage use	$43,705
Clear and Break 1 year	
CB 50% use	$37,771
CB 60% use	$53,738
CB Break Even: BE= 54%	$43,705
Clear and Break 2 year	
CB 50% use	$31,126
CB 60% use	$45,765
CB Break Even BE=59%	$43,705
Spray and Burn 70% use in 1st 4 yrs	
70% use 1st 4 years, then 50	$43,009
70% use 1st 4 years, then 60	$51,139
70% use 1st 4 years, BE= 51%	$43,705

Note 1: Break even is where the NPV of treatment equals the NPV of the Native Range under the same set of economic assumptions.
Note 2: S&B breakeven is based on 70% use in first four years and then breakeven utilization on remaining years.

Table 5-5: Sensitivity in NPV to variation in forage usage within the Clear and Break and Spray and Burn treatments, based on the use of all forage (i.e. ANPP) and herbage only. Analysis uses a forage value of $10/AUM, along with changes in the duration of economic productions in the C&B (i.e. 1 or 2 years). Analysis also uses 1979-80 costs with an 8% discount rate.

Forage Utilization and Treatment	$10/AUM
ANPP	
Native Range 50% use	$35,635
Clear and Break 1 year	
CB 50% use	$26,054
CB 60% use	$35,761
CB Break Even: BE= 60%,	
	$35,635
Clear and Break 2 year	
CB 50% use	$22,174
CB 60% use	$31,106
CB Break Even = 65%	$35,635
Spray and Burn 70% 1st 4 yrs	
70% use, then 50	$43,167
70% use, then 60	$50,114
70% use, BE= 39%	$35,635
Herbage	
Native Range 50% use	$21,853
Clear and Break 1 year	
CB 50% use	$17,436
CB 60% use	$25,420
CB Break Even: BE= 56%	$21,853
Clear and Break 2 year	
CB 50% use	$14,114
CB 60% use	$21,433
CB Break Even BE=61%	$21,853
Spray and Burn 70% use in 1st 4 yrs	
70% use, then 50	$19,667
70% use, then 60	$23,732
70% use, BE= 55%	$21,853

Note 1: Break even is where the NPV of treatment equals the NPV of the Native Range under the same set of economic assumptions.
Note 2: SB breakeven is 70% first four years and then breakeven utilization on remaining years.

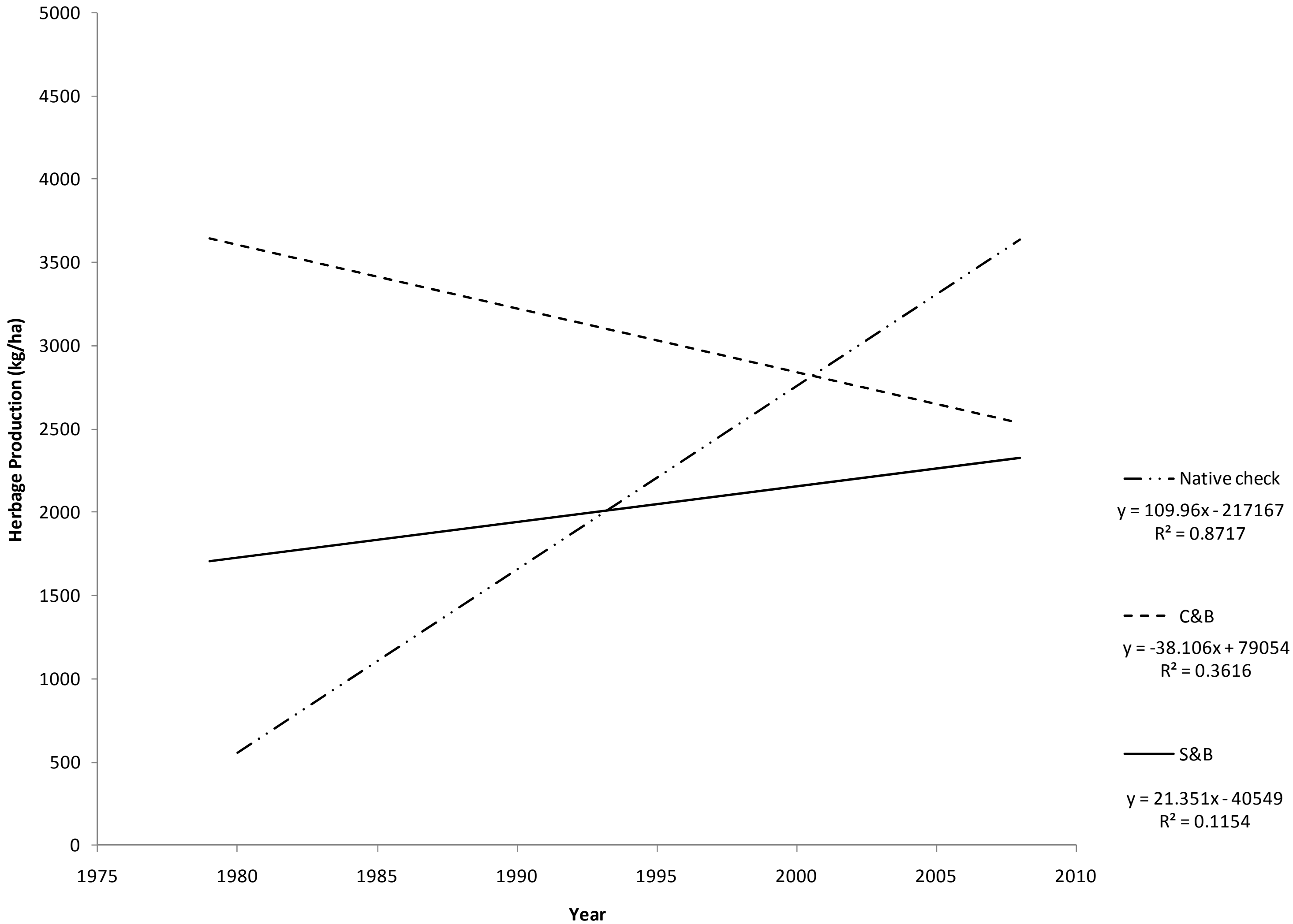

Figure 5-1: Real and predicted values of herbage production (kg/ha) for each of the three treatments (Native Check, Clear and Break and Spray and Burn) from 1980 to 2006.

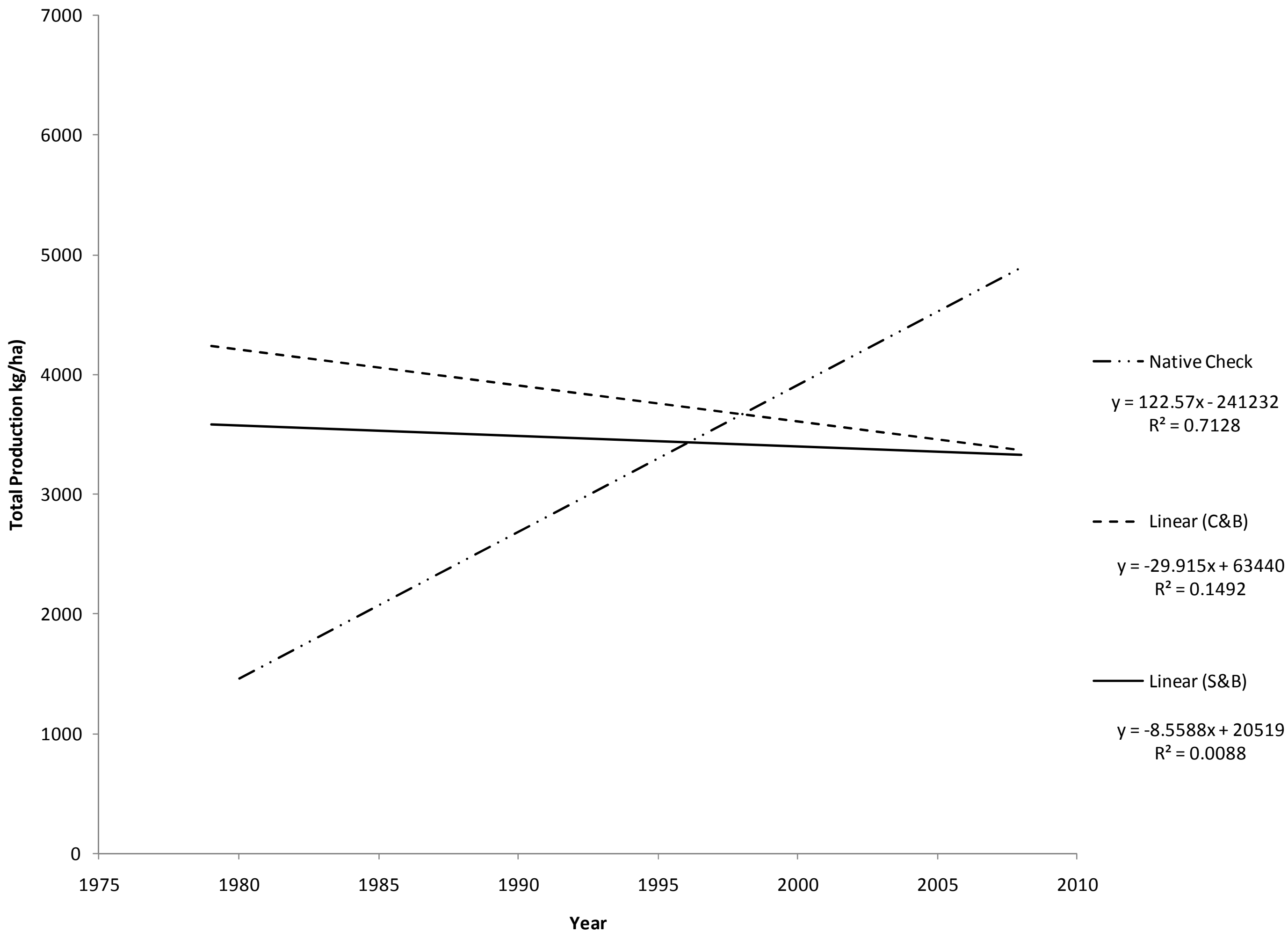

Figure 5-2: Real and predicted values of total production (kg/ha) for each of the three treatments (Native Check, Clear and Break and Spray and Burn) from 1980 to 2006.

6 Synthesis

Aspen encroachment into grasslands of the Parkland has been well documented (Moss and Campbell 1947, Johnston and Smoliak 1968, Bailey and Wroe 1974, Scheffler 1976, Simonson and Johnson 2005). A primary objective of contemporary rangeland managers in the Parkland of Alberta has been to increase the availability and accessibility of preferred forage through a reduction in woody species. There are two contrasting agricultural land management strategies for dealing with aspen encroachment, including intensive and extensive aspen control. The prevailing trend in the Aspen Parkland since the turn of the 20th century has been wholesale land conversion from native ecosystems into tame pastures or intensive cropping systems. This type of intensification of agricultural systems may not be sustainable and can lead to a reduced carrying capacity through reduced organic matter and changes in soil nutrient regimes (Huang et al 2002).

The purpose of this research was to examine the long-term agronomic (forage yield and quality), ecologic (community diversity), and economic value of three contrasting brush management strategies available to producers operating in the Parkland. These strategies represent contrasting philosophies on the degree of capital inputs necessary to control aspen and restore rangeland productivity, and thereby optimize profitability. Results from Chapter 3 demonstrate the continued presence of treatment-based effects in terms of plant species richness and diversity, the relative dominance of introduced and native species, and forage production (i.e. yield and quality) differences, approximately 25 years after the implementation of landscape level treatments (C&B, S&B and NC) for aspen forest control.

Twenty-five years after the conversion of native grassland and forest into seeded agronomic communities, plant communities in the C&B remained overall lower in total species richness and diversity compared to both the S&B and NC. Nevertheless, the C&B retained a substantial presence of native species, which provides evidence of the resiliency of upland grasslands in the Aspen Parkland. Moreover, as native species may have superior adaptations compared to introduced species (Baruch et al. 1985) to local growing conditions, this may enable them to survive better despite limitations in soils or fluctuations in growing conditions (e.g. during drought). This finding may have important future management implications as native species may have a competitive advantage as conditions become more favorable for their reproduction and growth (i.e. increasing drought under climate change).

The results of this study highlight the overall role of both forested habitats, and non-grass based forage sources in contributing to forage production potential across the Parkland. Aspen groves create microclimatic conditions that retain moisture and recycle nutrients (Maini 1960) and can provide optimum forage on a season-long basis (Asamoah et al.2003). Therefore, managed grazing of shrubs in the Aspen Parkland may be a viable option for rangeland managers.

Moreover, the results of the cost: benefit analysis suggest that the additional cost associated with intensive C&B activities may not be justified. Although plagued by many assumptions, many of which are subject to debate, our analysis indicated that the S&B continued to provide the greatest NPV among the 3 landscape treatments examined. This result appeared to occur from the optimal combination of reduced input costs associated with treatment, the absence of a forage opportunity cost shortly after establishment, and the sizeable contribution of browse from forested communities. Abundant browse was typically high in forage quality, and along with forbs, provided a low cost, valuable asset for grazing. When combined, the S&B landscape treatment resulted in the greatest NPV, and did so with a lower initial investment than the conventional C&B treatment.

Future research should look to further establish the relationship between community diversity and rangeland variables of key commercial importance to ranching, including maintaining stability in production through variable growing conditions, as may occur under climate change. Additional work is needed to understand the degree to which introduced species are able to fulfill key functional roles in the ecosystem, including ensuring site stability, promoting energy flow and nutrient retention, as well as sequestering carbon. As there is often debate over the extent and conditions under which browse constitutes ‘usable’ forage by cattle, more information is needed to understand the management factors regulating browse use (e.g. stocking rate, timing of use, role of grazing systems, etc.) of aspen forest in the Parkland. A comparison of native and introduced species may also help to understand the mechanisms regulating native rangeland resistance to introduced species invasion, as well as native rangeland resilience following land conversion. All of this information will help to develop an understanding of the potential economic ramifications based on ecosystem responses, particularly in relation to global stressors such as climate change.

When one considers the scale to which arable lands have been shifted outside of their natural realm of resilience (beyond natural disturbance regimes) what impact do these land uses have on those fragmented native remnants within this intensely managed system? Is brush encroachment an adaptive measure or perhaps a defense mechanism in response to surrounding land use?

Today, the federal and provincial governments have developed programs to help producers plan for operational improvements that will reduce their environmental impact (Growing Forward 2009). More specifically the Grazing and Winter Feeding Management Stewardship Plans are designed to help producers develop actions to mitigate their highest environmental risks and to minimize their impact on the environment. Match funding grant funding to a maximum of $15,000 is available for the Grazing and Winter Feeding Management program (Growing Forward 2009). Eligible projects within this program include alternative watering systems, shelterbelt establishment using native species, fencing to protect environmentally sensitive and enhance grazing management areas, riparian health assessments and riparian restoration and native upland range establishment or restoration through the purchase and planting of native species. The final project mentioned which also receives support from Ducks Unlimited Canada is of particular interest as a cost effective “renovation” of the C&B uplands. As this research indicates the extensive conversion of native grasslands in the Aspen Parkland has not only led to lowered total and native species richness and diversity of these ecosystem but also suggests lower abundance of soil biota. Yet native species diversity of the C&B upland areas remained comparable to native grasslands suggesting that native species may have superior adaptations for survival on these sites. The observation of native species in these modified rangelands should not be overlooked and perhaps active restoration combined with the natural recovery of these areas could lead to not only more productive ecosystems but also a diverse resilient plant community well adapted to upland growing conditions

These results will have significant implications for the future management of brush in western Canada, and thus, the conservation of remaining native rough fescue grasslands. Even though the results of this study indicate that native pastures can provide a consistent level of forage, it is important to recognize the influence of introduced forages that have invaded into these native pastures. Today, the majority of the Aspen Parkland has been modified beyond natural disturbance regimes (i.e. introduction of agronomic pastures and cropping systems), which in turn, may have a strong influence on the ecology of remaining fragments of native aspen groves. Nonetheless, adaptive management and innovative grazing strategies can be used to utilize, at least in part, available shrub and tree biomass and the integration of intensive and extensive management practices can be used to encourage sustainable management.

6.1 Literature Cited

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Appendix 1. Individual crude protein analysis results for plant species from pooled samples collected across the landscape.

Growth Form	Scientific Name	Common name	Crude Protein (%)
Forb	*Vicia americana*	American vetch	16.2
	Taraxacum officinale	Dandelion	12.8
Graminoid	*Hesperostipa curtiseta*	western porcupine	4.4
	Bouteloua gracilis	blue grama grass	11.4
	Koeleria macrantha	June grass	4.0
	Agropyron trachycaulum var. unilaterale	bearded wheatgrass	5.6
	Poa pratensis	Kentucky bluegrass	6.5
	Bromus inermis	smooth brome	8.6
	Agropyron smithii	western wheatgrass	9.0
	Festuca hallii	plains rough fescue	5.5
Shrub	*Rubus idaeus*	red raspberry	17.0
	Rosa sp	Rose	11.1
	Populus tremuloides	trembling aspen	13.8
	Populus balsamifera	balsam poplar	15.8
	Amelanchier alnifolia	saskatoon berry	9.0
	Rosa arkansana	prairie rose	10.0
	Symphoricarpos occidentalis	western snowberry	11.3

Appendix 2. Mean percent cover estimates of shrubs by landscape treatment.

Shrubs			Mean Percent Cover by Treatment		
Species List	Common name	Origin	Native	Spray & Burn	Clear & Break
Actaea spp.	Baneberry	N	0.0	0.0	0.0
Amelanchier alnifolia	Saskatoon	N	2.3	0.9	0.53
Artemisia frigida	Pasture Sage	N	2.8	1.8	4.6
Artemisia ludoviciana	Prairie Sage	N	0.5	0.6	0.3
Cornus stolonifera	Red Osier Dogwood	N	0.1	0.1	0.0
Elaeagnus commutata	Wolf Willow	N	0.2	0.3	0.9
Lonicera dioca	Twining Honeysuckle	N	0.3	0.1	0.0
Populus balsamifera	Balsam Poplar	N	0.1	1.0	0.0
Populus tremuloides	Trembling Aspen	N	2.7	8.8	1.1
Prunus virginiana	Choke Cherry	N	0.2	0.2	0.0
Ribes	Gooseberry	N	1.0	2.0	0.2
Rosa acicularis	Prickly Rose	N	1.5	3.7	0.8
Rosa sp.	Rose	N	5.8	6.1	1.3
Salix sp.	Willow	N	0.5	0.2	0.1
Shepherdia canadensis	Buffalo Berry	N	0.2	0.4	0.2
Symphoricarpos occidentalis	Western Snowberry	N	7.7	14.1	14.3

Appendix 3. Mean percent cover estimates of grasses by landscape treatment.

Grasses			Mean Percent Cover by Treatment		
Species List	Common name	Origin	Native	Spray & Burn	Clear & Break
Agropyron cristatum	Crested Wheatgrass	I	0.0	0.1	0.0
Agropyron repens	Quackgrass	I	0.0	0.0	0.0
Agrostis stolonifera	Redtop	I	0.1	0.0	0.0
Bromus inermis	Smooth Brome	I	9.8	10.4	15.0
Dactylis glomerata	Orchard Grass	I	0.0	0.8	0.0
Elymus junceus	Russian Wild Rye	I	0.0	0.0	0.2
Poa pratensis	Kentucky Bluegrass	I	12.6	17.0	12.8
Agropyron dasystachyum	Northern Wheatgrass	N	0.7	0.1	0.1
Agropyron smithii	Western Wheatgrass	N	0.7	0.3	0.6
Agropyron trachycaulum var. trachy	Wheatgrass	N	0.0	0.0	0.0
Agropyron trachycaulum var. unilat	Bearded Wheatgrass	N	0.6	1.3	2.8
Agrostis scabra	Ticklegrass	N	0.1	0.2	0.3
Bouteloua gracilis	Blue Grama Grass	N	0.5	0.6	0.6
Bromus ciliatus	Fringed Brome	N	0.0	0.1	0.0
Calamagrostis canadensis	Marsh Reed Grass	N	0.4	0.0	0.0
Calamovilfa longifolia	Sandgrass	N	0.0	0.0	0.0
Carex sp.	Sedges	N	6.6	8.0	5.9
Danthonia parryi	Parry's Oatgrass	N	0.0	0.0	0.0
Deschampsia caespitosa	Tufted Hair Grass	N	0.1	0.0	0.1
Festuca hallii	Plains Rough Fescue	N	0.2	0.4	0.3
Festuca saximontana	Sheep Fescue	N	0.0	0.2	0.1
Glyceria grandis	Tall Manna Grass	N	0.0	0.0	0.0

Appendix 3. continued

Grasses			Mean Percent Cover by Treatment		
Species List	Common name	Origin	Native	Spray & Burn	Clear & Break
Helictotrichon hookeri	Hooker's Oatgrass	N	0.1	0.0	0.0
Hierochloe odorata	Sweetgrass	N	0.0	0.0	0.1
Hordeum jubatum	Foxtail Barley	N	0.0	0.0	0.0
Koeleria macrantha	June Grass	N	1.8	0.8	0.8
Phalaris arundinacea	Canary reed grass	N	0.0	0.1	0.0
Poa palustris	Fowl Bluegrass	N	0.2	0.7	0.0
Stipa comata	Spear Grass	N	0.1	0.0	0.3
Stipa curtiseta	Western Porcupine Grass	N	2.8	3.2	2.4
Stipa viridula	Green Needle Grass	N	0.2	0.2	0.1

Appendix 4. Mean percent cover estimates of forbs by landscape treatment.

Forbs			Mean Percent Cover by Treatment		
Species List	Common name	Origin	Native	Spray & Burn	Clear & Break
Chenopodium album	Lamb's Quarters	I	2.4	0.0	0.0
Circium arvense	Canada Thistle	I	0.1	0.2	1.1
Galeopsis tetrahit	Hemp Nettle	I	0.2	0.1	0.0
Medicago sativa	Alfalfa	I	0.0	0.0	0.3
Plantago major	Common Plantain	I	0.0	0.0	0.0
Polygonum convolvulus	Wild Buckwheat	I	1.75.2	0.0	0.1
Sonchus arvensis	Perennial Sow Thistle	I	0.1	0.0	0.0
Taraxacum officinale	Dandelion	i	0.8	1.1	0.1
Achillea millefolium	Yarrow	n	0.4	1.5	1.4
Androsace septentrionalis	Fairy Candelabra	n	1.2	0.3	0.1
Anenome canadensis	Canada Anenome	n	0.1	0.5	0.5
Anenome cylindrica	Long Fruited Anenome	n	0.1	0.0	0.1
Anenome patens	Prairie Crocus	n	0.1	0.1	0.1
Antennaria parvifolia	Pussytoes	n	0.4	0.1	0.7
Aster ciliolatus	Lindley's Aster	n	0.0	0.2	0.0
Aster conspicuus	Showy Aster	n	0.0	0.4	0.1
Aster ericoides	Tufted White Prairie Aster	n	0.9	0.1	0.1
Aster laevis	Smooth Aster	n	0.2	0.2	0.3
Astragalus miser	Vetch	n	0.0	0.0	0.0
Astragalus sp.	Vetch	n	0.0	0.0	0.0
Astragalus striatus	Purple Milk Vetch	n	0.0	0.1	0.0

Appendix 4. continued

Forbs			Mean Percent Cover by Treatment		
Species List	Common name	Origin	Native	Spray & Burn	Clear & Break
Campanula rotundifolia	Bluebells	n	0.0	0.0	0.0
Cerastium arvense	Chickweed	n	0.2	0.2	0.0
Circium flodmanii	Flodman's Thistle	n	0.0	0.2	0.1
Comandra umbellata	Bastard Toadflax	n	0.3	0.2	0.3
Crepis tectorum	Annual Hawksbeard	n	0.3	0.0	0.0
Disporum trachycarpum	Fairybells	n	0.0	0.0	0.0
Epilobium angustifolium	Fireweed	n	0.0	0.0	0.0
Equisetum arvense	Common Horsetail	n	0.0	0.0	0.0
Erigeron glabellus	Smooth Fleabane	n	0.2	0.0	0.0
Erysimum cheiranthoides	Wormseed Mustard	n	0.0	0.0	0.0
Fragaria virginiana	Strawberry	n	2.7	8.2	11.5
Gaillardia aristata	Brown eyed Susan	n	0.0	0.0	0.0
Galium boreale	Northern Bedstraw	n	1.2	1.5	0.5
Galium triflorum	Sweet Scented Bedstraw	n	0.2	0.0	0.0
Gentianella amarella	Northern Gentian	n	0.0	0.0	0.0
Geum aleppicum	Yellow Avens	n	0.3	0.3	0.4
Geum macrophyllum	Large Leaved Avens	n	0.1	0.0	0.0
Geum triflorum	Three Flowered Avens	n	0.3	0.1	0.0
Habenaria hyperborea	Northern Green Bog Orchid	n	0.0	0.0	0.0
Hedysarum alpinum	Alpine Hedysarum	n	0.2	0.3	0.4
Heterotheca villosa	Hairy Golden Aster	n	0.0	0.0	0.0
Heuchera richardsonii	Richardson's Alumroot	n	0.0	0.0	0.0

Appendix 4. continued

Forbs			Mean Percent Cover by Treatment		
Species List	Common name	Origin	Native	Spray & Burn	Clear & Break
Lactuca tatarica (pulchella)	Common Blue Lettuce	n	0.2	0.0	0.0
Lathyrus ochroleucus	White Peavine	n	0.7	0.2	0.1
Lathyrus venosus	Purple Peavine	n	0.6	0.2	0.2
Lygodesmia juncea	Skeletonweed	n	0.1	0.0	0.1
Maianthemum canadense	Wild Lily of the Valley	n	0.1	0.1	0.0
Melilotus officinalis	Yellow sweet clove	n	0.0	0.0	0.0
Mentha arvensis	Wild Mint	n	0.2	0.1	0.0
Mitella nuda	Bishop's Cap	n	0.0	0.0	0.0
Muhlenbergia richardsonis	Mat Muhley	n	0.0	0.0	0.1
Orthocarpus luteus	Owl's Clover	n	0.0	0.0	0.0
Oxytropis sericea	Early Yellow Locoweed	n	0.1	0.0	0.0
Penstemon glaber	Smooth penstemon	n	0.0	0.0	0.0
Pentstemon gracilis	Lilac Flowered Beardtongue	n	0.0	0.0	0.0
Pentstemon procerus	Slender Blue Beardtongue	n	0.0	0.0	0.0
Petalostemon purpureum	Purple Prairie Clover	n	0.0	0.0	0.0
Petasites sagittatus	Arrow Leaved Coltsfoot	n	0.0	0.0	0.0
Phlox hoodii	Moss Phlox	n	0.0	0.0	0.0
Polygonum pensylvanicum	Smartweed	n	0.0	0.1	0.0
Potentilla anserina	Silverweed	n	2.7	0.0	0.5
Potentilla arguta	White Cinquefoil	n	0.0	0.2	0.0
Potentilla concinna	Early Cinquefoil	n	0.1	0.1	0.0
Potentilla gracilis	Graceful Cinquefoil	n	0.0	0.1	0.0
Potentilla pensylvanica	Prairie Cinquefoil	n	0.1	0.1	0.1
Psoralea agrophylla	Silverleaf Psoralea	n	0.1	0.0	0.0
Pyrola asarifolia	Pink Wintergreen	n	0.0	0.2	0.0
Rubus idaeus	Red Raspberry	n	1.5	1.9	0.0

Appendix 4. continued

Forbs Species List	Common name	Origin	Mean Percent Cover by Treatment Native	Spray & Burn	Clear & Break
Schizachne purpurascens	False Melic	n	0.3	0.3	0.2
Senecio eremophilus	Cut leaved Ragwort	n	0.0	0.0	0.0
Sisyrinchium montanum	Blue Eyed Grass	n	0.3	0.0	0.0
Smilacina stellata	Star Flwrd Solomon's Seal	n	0.4	0.9	0.3
Solidago Canadensis	Canadian Goldenrod	n	0.5	0.1	0.3
Solidago missouriensis	Low Goldenrod	n	1.0	0.4	0.5
Sphaeralcea coccinea	Scarlet Mallow	n	0.0	0.0	0.0
Stachys palustris	Marsh Hedge Nettle	n	0.2	0.0	0.1
Thalictrum venulosum	Veiny Medow Rue	n	2.9	1.1	0.1
Thermopsis rhombifolia	Buffalo Bean	n	0.4	0.6	0.7
Utrica dioica	Stinging Nettle	n	0.0	0.0	0.0
Vicia Americana	American Vetch	n	1.3	0.6	0.2
Viola adunca	Early Blue Violet	n	0.5	0.8	0.4
Viola Canadensis	Western Canadian Violet	n	0.2	0.0	0.0
Viola renifolia	Kidney leaved violet	n	0.2	0.0	0.0
Zigadenus elegans	Smooth/White Camas	n	0.0	0.0	0.0
Zizia aptera	Heart Leaved Alexander	n	0.0	0.0	0.0

Appendix 5. Mean percent cover of shrubs by treatment and landscape position.

		Bottom Forest			North Forest			Grassland		
Species List	Origin	Native	S&B	C&B	Native	S&B	C&B	Native	S&B	C&B
Actaea spp.	n	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Amelanchier alnifolia	n	2.9	0.7	0.1	3.9	1.9	1.3	0.0	0.0	0.2
Artemisia frigida	n	0.0	0.2	0.3	0.1	0.0	1.0	8.3	5.3	12.6
Artemisia ludoviciana	n	0.0	0.0	0.1	0.0	0.1	0.6	1.4	1.7	0.4
Cornus stolonifera	n	0.2	0.0	0.0	0.1	0.4	0.0	0.0	0.0	0.0
Elaeagnus commutata	n	0.3	0.1	0.4	0.2	0.4	1.9	0.2	0.5	0.5
Lonicera dioca	n	0.1	0.0	0.0	0.7	0.2	0.0	0.0	0.0	0.0
Populus balsamifera	n	0.2	1.0	0.0	0.0	1.9	0.0	0.0	0.0	0.0
Populus tremuloides	n	6.0	10.2	2.3	2.1	16.1	0.9	0.0	0.0	0.0
Prunus virginiana	n	0.4	0.2	0.0	0.2	0.4	0.1	0.0	0.0	0.0
Ribes	n	1.7	2.1	0.1	1.3	3.8	0.4	0.0	0.1	0.0
Rosa acicularis	n	3.3	3.6	0.3	1.1	7.5	2.1	0.0	0.2	0.2
Rosa sp.	n	5.7	8.2	0.7	9.6	8.3	1.9	2.0	1.9	1.2
Salix sp.	n	0.1	0.3	0.2	1.5	0.4	0.1	0.0	0.0	0.0
Shepherdia canadensis	n	0.1	0.6	0.0	0.4	0.7	0.7	0.0	0.0	0.0
Symphoricarpos occidentalis	n	10.6	14.3	17.8	8.3	16.7	19.2	4.1	11.3	5.8

Appendix 6. Mean percent cover of grasses by treatment and landscape position.

		Bottom Forest			North Forest			Grassland		
Species List	Origin	Native	S&B	C&B	Native	S&B	C&B	Native	S&B	C&B
Agropyron cristatum	i	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
Agropyron repens	i	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Agrostis stolonifera	i	0.4	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Bromus inermis	i	21.4	14.6	20.2	6.0	12.3	14.3	2.0	4.2	10.3
Dactylis glomerata	i	0.0	0.8	0.0	0.0	1.5	0.0	0.0	0.0	0.0
Elymus junceus	i	0.0	0.0	0.3	0.0	0.0	0.2	0.0	0.0	0.0
Poa pratensis	i	16.4	14.3	18.2	3.9	12.9	10.3	17.5	23.9	9.7
Agropyron dasystachyum	n	0.0	0.0	0.0	0.3	0.0	0.0	1.8	0.4	0.4
Agropyron smithii	n	0.0	0.0	0.0	0.0	0.0	0.0	2.2	1.0	1.7
Agropyron trachycaulum var. trachycaulum	n	0.1	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0
Agropyron trachycaulum var. unilaterale	n	0.3	1.1	3.3	0.1	0.9	3.8	1.5	1.9	1.2
Agrostis scabra	n	0.1	0.2	0.3	0.3	0.1	0.6	0.0	0.2	0.1
Bouteloua gracilis	n	0.0	0.0	0.0	0.0	0.0	0.0	1.6	2.0	1.7
Bromus ciliatus	n	0.1	0.2	0.0	0.0	0.0	0.1	0.0	0.0	0.0
Calamagrostis canadensis	n	1.1	0.1	0.0	0.2	0.0	0.0	0.0	0.0	0.0
Calamovilfa longifolia	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Carex sp.	n	4.1	7.3	4.8	4.8	5.6	4.4	11.2	11.0	8.5
Danthonia parryi	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Deschampsia caespitosa	n	0.0	0.0	0.0	0.4	0.0	0.2	0.0	0.0	0.1
Festuca hallii	n	0.0	0.0	0.0	0.0	0.0	0.1	0.6	1.1	0.7
Festuca saximontana	n	0.0	0.1	0.1	0.0	0.3	0.1	0.0	0.3	0.0
Glyceria grandis	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Helictotrichon hookeri	n	0.0	0.0	0.0	0.4	0.1	0.0	0.0	0.1	0.0
Hierochloe odorata	n	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Hordeum jubatum	n	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0

Appendix 6. continued

		Bottom Forest			North Forest			Grassland		
Species List	Origin	Native	S&B	C&B	Native	S&B	C&B	Native	S&B	C&B
Koeleria macrantha	n	0.0	0.0	0.0	0.0	0.0	0.1	5.5	2.3	2.3
Phalaris arundinacea	n	0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Poa palustris	n	0.7	2.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0
Stipa comata	n	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	1.0
Stipa curtiseta	n	0.0	0.0	0.3	0.0	0.0	0.1	8.4	9.5	6.8
Stipa viridula	n	0.2	0.0	0.1	0.0	0.0	0.0	0.3	0.6	0.1

Appendix 7. Mean percent cover of forb species by treatment and landscape position.

		Bottom Forest			North Forest			Grassland		
Species List	Origin	Native	S&B	C&B	Native	S&B	C&B	Native	S&B	C&B
Chenopodium album	i	0.0	0.0	0.0	7.3	0.0	0.0	0.0	0.1	0.0
Circium arvense	i	0.3	0.7	3.0	0.1	0.0	0.2	0.0	0.0	0.0
Galeopsis tetrahit	i	0.3	0.3	0.1	0.2	0.0	0.0	0.0	0.0	0.0
Medicago sativa	i	0.0	0.1	0.	0.0	0.0	0.3	0.0	0.0	0.0
Plantago major	i	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Polygonum convolvulus	i	0.0	0.0	0.2	5.2	0.0	0.0	0.0	0.0	0.0
Sonchus arvensis	i	0.3	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0
Taraxacum officinale	i	1.2	1.6	0.2	1.1	1.4	0.1	0.0	0.2	0.0
Achillea millefolium	n	0.3	2.1	1.6	0.0	1.2	2.0	0.9	1.2	0.8
Androsace septentrionalis	n	0.0	0.1	0.0	3.5	0.0	0.1	0.2	0.7	0.1
Anenome canadensis	n	0.2	1.0	1.2	0.1	0.5	0.2	0.0	0.0	0.0
Anenome cylindrica	n	0.0	0.0	0.0	0.1	0.0	0.1	0.1	0.0	0.1
Anenome patens	n	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.4	0.3
Antennaria parvifolia	n	0.0	0.1	1.0	0.0	0.0	1.0	1.1	0.1	0.0
Aster ciliolatus	n	0.0	0.1	0.0	0.0	0.3	0.0	0.0	0.0	0.0
Aster conspicuus	n	0.0	0.6	0.0	0.0	0.6	0.2	0.0	0.0	0.0
Aster ericoides	n	0.0	0.0	0.0	0.2	0.0	0.0	2.6	0.2	0.3
Aster laevis	n	0.2	0.2	0.2	0.2	0.2	0.5	0.2	0.0	0.1
Astragalus miser	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Astragalus sp.	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Astragalus striatus	n	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.0
Campanula rotundifolia	n	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0
Cerastium arvense	n	0.0	0.0	0.0	0.3	0.0	0.0	0.4	0.5	0.1
Circium flodmanii	n	0.0	0.3	0.2	0.0	0.2	0.0	0.0	0.2	0.0
Comandra umbellata	n	0.0	0.0	0.0	0.0	0.0	0.0	1.0	0.5	0.9
Crepis tectorum	n	0.1	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0

Appendix 7. continued

		Bottom Forest			North Forest			Grassland		
Species List	Origin	Native	S&B	C&B	Native	S&B	C&B	Native	S&B	C&B
Disporum trachycarpum	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Epilobium angustifolium	n	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
Equisetum arvense	n	0.0	0.0	0.1	0.	0.0	0.0	0.0	0.0	0.0
Erigeron glabellus	n	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.1	0.1
Erysimum cheiranthoides	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fragaria virginiana	n	3.9	10.8	13.1	4.2	13.2	21.4	0.0	0.6	0.2
Gaillardia aristata	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Galium boreale	n	1.4	1.6	0.1	2.0	2.8	1.2	0.3	0.2	0.2
Galium triflorum	n	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0
Gentianella amarella	n	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
Geum aleppicum	n	0.5	0.2	0.8	0.3	0.6	0.4	0.0	0.0	0.0
Geum macrophyllum	n	0.2	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0
Geum triflorum	n	0.0	0.0	0.0	0.0	0.1	0.0	0.8	0.2	0.1
Habenaria hyperborea	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Hedysarum alpinum	n	0.4	0.2	0.6	0.1	0.5	0.7	0.0	0.0	0.0
Heterotheca villosa	n	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1
Heuchera richardsonii	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Lactuca tatarica (pulchella)	n	0.4	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0
Lathyrus ochroleucus	n	0.9	0.3	0.1	1.3	0.4	0.2	0.0	0.0	0.0
Lathyrus venosus	n	0.4	0.2	0.4	1.4	0.4	0.0	0.0	0.0	0.0
Lygodesmia juncea	n	0.0	0.0	0.1	0.2	0.0	0.0	0.1	0.0	0.1
Maianthemum canadense	n	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0
Melilotus officinalis	n	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0
Mentha arvensis	n	0.5	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Mitella nuda	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Muhlenbergia richardsonis	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2
Orthocarpus luteus	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Oxytropis sericea	n	0.1	0.0	0.1	0.0	0.0	0.0	0.2	0.0	0.0
Penstemon glaber	n	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0

Appendix 7. continued

Species List	Origin	Bottom Forest Native	Bottom Forest S&B	Bottom Forest C&B	North Forest Native	North Forest S&B	North Forest C&B	Grassland Native	Grassland S&B	Grassland C&B
Pentstemon gracilis	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pentstemon procerus	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Petalostemon purpureum	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Petasites sagittatus	n	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Phlox hoodii	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Polygonum pensylvanicum	n	0.0	0.2	0.0	0.0	0.0	0.0	0.1	0.0	0.0
Potentilla anserina	n	0.0	0.0	1.4	8.0	0.0	0.1	0.1	0.1	0.1
Potentilla arguta	n	0.0	0.2	0.1	0.0	0.1	0.0	0.1	0.2	0.0
Potentilla concinna	n	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.1
Potentilla gracilis	n	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.1
Potentilla pensylvanica	n	0.0	0.1	0.0	0.0	0.0	0.0	0.3	0.1	0.4
Psoralea agrophylla	n	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.1
Pyrola asarifolia	n	0.0	0.5	0.0	0.0	0.2	0.0	0.0	0.0	0.0
Rubus idaeus	n	1.5	2.9	0.0	3.0	2.8	0.1	0.1	0.0	0.0
Schizachne purpurascens	n	0.0	0.4	0.0	0.8	0.5	0.5	0.0	0.0	0.0
Senecio eremophilus	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Sisyrinchium montanum	n	0.0	0.0	0.0	1.0	0.1	0.0	0.0	0.0	0.0
Smilacina stellata	n	0.7	0.9	0.3	0.4	1.7	0.5	0.0	0.1	0.0
Solidago canadensis	n	0.9	0.2	0.6	0.6	0.2	0.2	0.1	0.0	0.1
Solidago missouriensis	n	0.0	0.0	0.0	0.3	0.0	0.0	2.8	1.3	1.4
Sphaeralcea coccinea	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stachys palustris	n	0.2	0.1	0.0	0.0	0.0	0.0	0.4	0.0	0.1
Thalictrum venulosum	n	1.5	1.4	0.0	7.2	1.9	0.2	0.1	0.1	0.0
Thermopsis rhombifolia	n	0.0	0.0	0.0	0.5	0.1	0.3	0.7	1.6	1.7
Utrica dioica	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Vicia americana	n	1.7	0.9	0.3	1.9	0.9	0.2	0.2	0.1	0.0
Viola adunca	n	0.3	1.1	0.5	1.2	1.1	0.6	0.0	0.1	0.1
Viola canadensis	n	0.0	0.1	0.0	0.6	0.1	0.0	0.0	0.0	0.0
Viola renifolia	n	0.1	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0
Zigadenus elegans	n	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Zizia aptera	n	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix 8. Summary of Field 9A (native) data assessing biomass, species richness and diversity of native, introduced and all plant species at the Kinsella Research Station in the Aspen Parkland of central Alberta in 2006.

Biomass	Landscape Position	Results	SE
Grass	Grassland	n/a	n/a
	North Forest	589.2 kg/ha	153.7
	Bottom Forest	934.8 kg/ha	278.2
Forb	Grassland	n/a	n/a
	North Forest	339.6 kg/ha	126.7
	Bottom Forest	337.3 kg/ha	119.5
Shrub	Grassland	n/a	n/a
	North Forest	1332.4 kg/ha	190.9
	Bottom Forest	846.4 kg/ha	205.8
Herbage	Grassland	n/a	n/a
	North Forest	928.8 kg/ha	229.7
	Bottom Forest	1238.4 kg/ha	265.5
Mean Total DM Production	Grassland	n/a	n/a
	North Forest	2261.2 kg/ha	226.0
Species Richness			
Native species	Grassland	n/a	n/a
	North Forest	10.4	1.2
	Bottom Forest	11.3	1.3
Introduced species	Grassland	n/a	n/a
	North Forest	1.5	0.3
	Bottom Forest	2.9	0.6
Total species	Grassland	n/a	n/a
	North Forest	11.9	1.2
	Bottom Forest	14.2	1.2
Species Diversity			
Native species	Grassland	n/a	n/a
	North Forest	1.6	0.2
	Bottom Forest	1.9	0.1
Introduced species	Grassland	n/a	n/a
	North Forest	0.2	0.1
	Bottom Forest	0.4	0.1
Total species	Grassland	n/a	n/a
	North Forest	1.7	0.2
	Bottom Forest	1.9	0.1

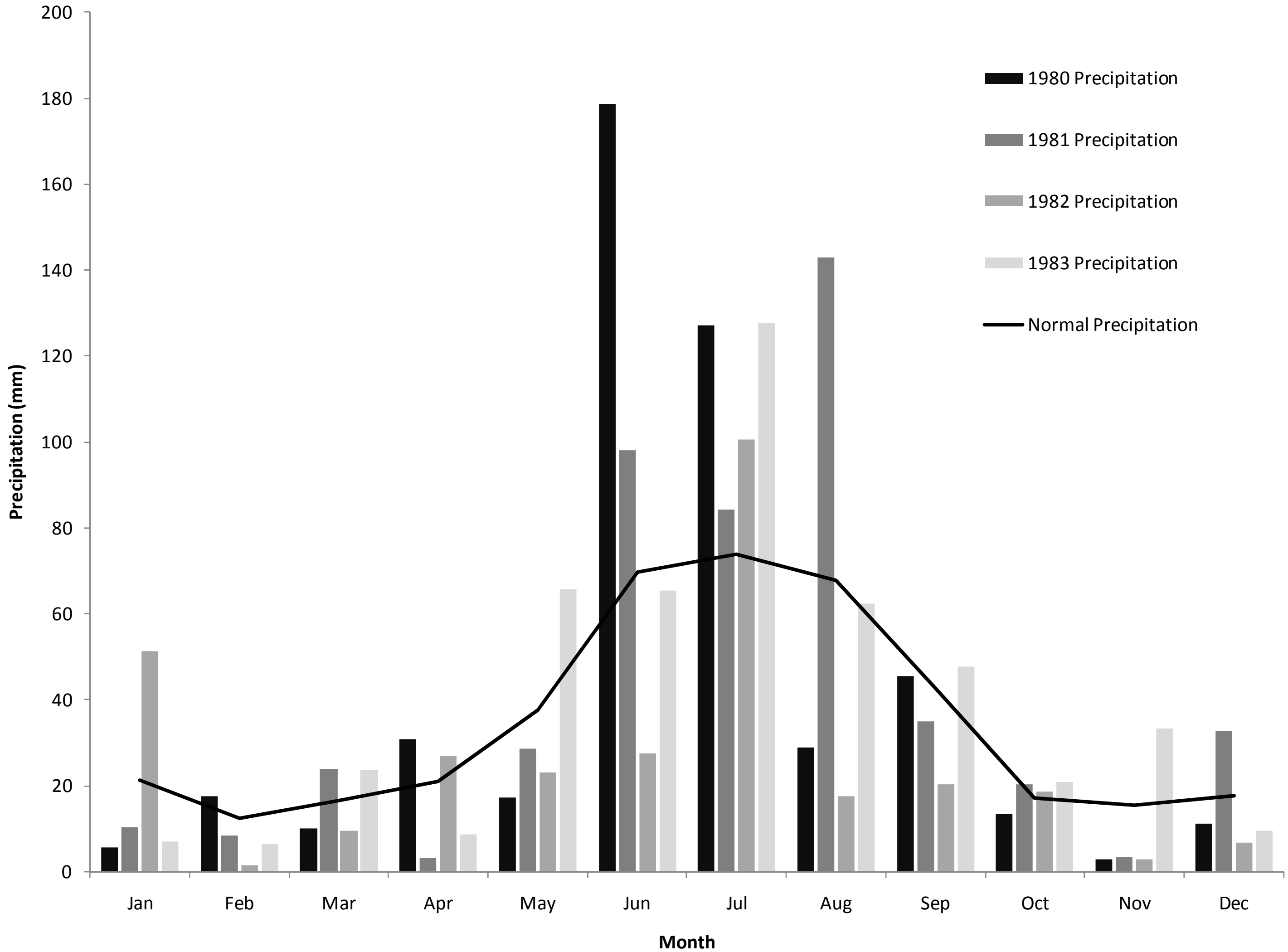
Appendix 9. Estimates of ANPP for each year (1979-2006) used in the NPV analysis. Data from1984 through 2004 were derived through regression models.

	Treatment ANPP Estimates (kg/ha)		
Year	Native Check	S&B	C&B
1979	1338	1375	4239
1980	1460	1487	4209
1981	1599	2120	5115
1982	1807	4305	4111
1983	1778	4383	3179
1984	1951	3539	4089
1985	2073	3530	4059
1986	2196	3521	4029
1987	2318	3513	3999
1988	2441	3504	3969
1989	2563	3496	3939
1990	2686	3487	3909
1991	2809	3479	3880
1992	2931	3470	3850
1993	3054	3461	3820
1994	3176	3453	3790
1995	3299	3444	3760
1996	3421	3436	3730
1997	3544	3427	3700
1998	3667	3419	3670
1999	3789	3410	3640
2000	3912	3402	3610
2001	4034	3393	3580
2002	4157	3384	3551
2003	4279	3376	3521
2004	4402	3367	3491
2005	3235	2258	2578
2006	5885	4310	4355

Appendix 10. Estimates of herbage for each year (1979-2006) used in the NPV analysis. Data from1984 through 2004 were derived through regression models.

	Treatment Herbage Production Estimates (kg/ha)		
Year	Native Check	S&B	C&B
1979	449	1705	3642
1980	559	1726	3604
1981	669	656	4342
1982	752	2186	3490
1983	947	2562	2706
1984	999	1812	3452
1985	1109	1833	3413
1986	1219	1854	3375
1987	1329	1876	3337
1988	1438	1897	3299
1989	1548	1918	3261
1990	1658	1940	3223
1991	1768	1961	3185
1992	1878	1982	3147
1993	1988	2004	3109
1994	2098	2025	3071
1995	2208	2046	3032
1996	2318	2068	2994
1997	2428	2089	2956
1998	2538	2111	2918
1999	2648	2132	2880
2000	2758	2153	2842
2001	2868	2175	2804
2002	2978	2196	2766
2003	3088	2217	2728
2004	3198	2239	2689
2005	2607	1788	2181
2006	4087	2656	3129

Appendix 11: Comparison of mean monthly precipitation for 1980-83 to regional norms (Environment Canada1949-2004)



Appendix 12: Comparison of mean monthly temperature for 1980-83 to regional norms (Environment Canada1949-2004)