”... the grass in this new world was a cause for amazement, as it rebounded from the conquistadors’ steps and erased the trace of their presence. In this great round world, all that glittered was grass and an ecosystem of such richness and diversity that could scarcely be credited.”

Savage, C. 2004. Prairie. A natural history. Greystone Books. Vancouver, B.C.

“… this country of rich grasses and vetches, watered with many lakes, rivers and springs, was known throughout the world as a hunter’s paradise. With the coming of the railroad, a change took place, and one of most fruitful agricultural portions of the American continent has replaced what was generally accepted to have been the world’s greatest hunting ground.”

Farley, F.L. 1925. Changes in status of certain animals and birds during the past fifty years in central Alberta. Canadian Field Naturalist 39:200-202.

“Festuca hallii *makes good winter pasture provided the plants can cure completely before frost, and is palatable throughout the season except when litter accumulates ... Plains rough fescue makes good native hay, producing high yields ... (and) although rhizomatous, vegetative production is slow ... Contributing to its demise is the cultivation of vast acreages that were once pristine* F. hallii *rangelands.*

Tannas, K. 2001. Common plants of the western rangeland: volume 1 grasses and grass-like species. Alberta Agriculture, Food and Rural Development. Edmonton, AB.

University of Alberta

Rough Fescue (*Festuca hallii*) Ecology and Restoration in Central Alberta

by

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This PhD dissertation is dedicated to Darcy Warner, whose patience, understanding, and love supported me during my travels and long stays in Edmonton, and innumerable hours chained to my computer.

ABSTRACT

Festuca hallii (plains rough fescue), a late-seral bunchgrass and long-lived perennial, is difficult to restore once disturbed. Once dominant in grasslands throughout central Alberta, *F. hallii* now occurs in remnants, a result of agricultural and residential development, and oil and gas exploration and development.

This research program was designed to focus on establishment of *F. hallii* to provide evidence for predicting successional trends following disturbance. Experiments assessed the reaction of *F. hallii* and competing species, such as *Poa pratensis* (Kentucky bluegrass) and *Bromus inermis* (smooth brome), to disturbed and straw-amended soil. Assessments of pipelines left to natural recovery or seeded with native hay determined if these processes aided *F. hallii* establishment. *Festuca hallii* reliance on arbuscular mycorrhizal fungi (AMF) was analyzed, to determine if topsoil storage and subsequent AMF reduction was another factor in poor recovery of *F. hallii*. A state and transition model was developed for the Rumsey Natural Area, compiling vegetation assessments of historical and recent disturbances.

Festuca hallii displayed positive responses to straw treatments, while *P. pratensis* and *B. inermis* showed little response, concluding the addition of straw as a soil amendment is a possible solution to poor establishment of *F. hallii*. When seeded as a monoculture, *F. hallii* performed best, and plant community development, from seed bank or seed rain, was better than when seeding with a mix of native species. This resulted in a recommendation to seed *F. hallii* at 15

kg/ha or less with little or no wheat grasses in the seed mix. The straw and AMF experiments had intriguing results regarding *F. hallii* use of ammonium and pH levels; both showed increased leaf lengths and biomass with reduced ammonium and lower pH. Contrary to the initial hypothesis, *F. hallii* above ground biomass, root biomass and tiller count increased with decreased AMF colonization.

Native hay cut from rough fescue grassland is a viable seed source for restoring disturbances. *Festuca hallii* appeared to recover better on plough-in pipeline right-of-ways than from seeding, most likely from remnant intact sod; therefore, narrow trenching with plough-in pipelining techniques is recommended for rough fescue grasslands.

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CHAPTER 1. ECOLOGY AND MANAGEMENT OF GRASSLANDS

1.1 Introduction

Grasslands are among the most threatened ecosystems in the world (Samson and Knopf 1994). *Festuca hallii* (Vasey) Piper (plains rough fescue) was the dominant species in grasslands from central Alberta to Western Ontario, Montana and North Dakota (Pavlick and Looman 1984). These grasslands have been reduced to remnants, a result of urban and rural development, cultivation, livestock over-grazing and oil and gas development. Grilz et al. (1994) estimated less than 5% of the original prairie dominated by *F. hallii* remains.

Festuca hallii, a late-seral bunchgrass and long-lived perennial, is difficult to restore once disturbed by oil and gas development, is reduced by moderate grazing and eliminated by heavy grazing (Looman 1969; Sinton 1980; Looman 1983). One of the largest remaining tracts of *F. hallii*-dominated grassland is located in central Alberta, Canada: 183 km^2 in a provincial protected area known as the Rumsey Natural Area and Rumsey Ecological Reserve (Rumsey Block; Figure 1.1). Ranching commenced in the Rumsey Block around 1895 and continues to this day. While protected from rural development, oil and gas exploitation is allowed in the Rumsey Block. Oil and gas activity commenced in the Rumsey Block in the 1950s, and has resulted in over 200 natural gas well sites and pipelines, in various stages of operation and reclamation. My PhD research experimented with reclamation techniques for re-establishing *F. hallii* and assessed the effects of disturbance in the Rumsey Block.

1.2 Central Alberta Grasslands

Temperate grasslands occupy about nine million km^2 or eight percent of the earth's terrestrial surface and are among the most altered ecosystems on the planet (White et al. 2000; Forrest et al. 2004). Grasslands of the Great Plains of North America originally covered about 2.6 million km^2, or approximately 14% of the continent north of Mexico (Ostlie et al. 1996; Ricketts et al. 1999).

Conventionally, these grasslands are classified into three major vegetation regions: tallgrass, mixedgrass and shortgrass prairie (Samson and Knopf 1994). Coupland and Brayshaw (1953) added a fourth, rough fescue (*Festuca* sp.) grassland, which is relatively limited in extent and is included as part of the mixedgrass prairie in some classifications. In Alberta, the Natural Regions Committee (2006) recognized two fescue-dominated grassland subregions within the Grassland Natural Region in the southern third of the province: the Northern Fescue Natural Subregion and the Foothills Fescue Natural Subregion (Figure 1.2). The Northern Fescue subregion is a narrow band stretching across the northern margin of the Grassland Natural region. Between the Grassland Natural region and the Boreal Forest in the north lies the Parkland Region, a mosaic of aspen woodlands, fescue grasslands, shrublands and wetlands on gently rolling landscape (Natural Regions Committee 2006). Within the Parkland Region, the Central Parkland Subregion extends in a 200 km-wide arc above the Grassland Natural region.

Festuca hallii is the dominant grass in undisturbed grassland in the Northern Fescue and Central Parkland subregions. The most extensive tracts of fescue grassland in these regions persist on private and adjacent public lands in hilly terrain where tillage is difficult and the predominant land use is cattle grazing. Rough fescue grassland is an important habitat for a variety of song birds, raptors and small mammals. Once prime habitat for plains bison, mule deer, white-tailed deer, moose, ruffed and sharp-tailed grouse are now common. These grasslands are some of the most productive in North America, generating deep carbon-rich Black Chernozemic soil and providing abundant high quality forage for livestock and native ungulates (Stout et al. 1981; Willms et al. 1985; Willms 1988; Willms et al. 1996).

1.3 Rumsey Block Vegetation

The knob and kettle topography of the Rumsey Block provides a variety of microclimates that support diverse vegetation and vegetation structures (Appendix

A). Grassland typically covers drier east, west and south facing slopes; depressional wetlands become sedge meadows, rush marshes or pond-like sloughs. *Populus tremuloides* occupies mesic sites near sloughs or north facing slopes (Bradley and Bradley 1977). *Salix* spp. are found bordering low wetlands, giving way to *Symphoricarpos occidentalis* Hook (western snowberry) farther up the slopes.

Festuca hallii dominates submesic to mesic sites, such as moraine plateaux and north and east facing slopes. South facing slopes and more xeric sites are dominated by *Hesperostipa curtiseta* (A.S. Hitchc.) Barkworth (western porcupine grass) communities associated with *F. hallii* and in the southern part of Rumsey Block, *Bouteloua gracilis* (Willd. *ex* Kunth) Lag. ex Griffiths (blue grama) (Wroe 1971; Bradley and Bradley 1977; Fehr 1982; Wershler and Wallis 1990). Drought-tolerant species, such as *B. gracilis, Selaginella densa* Rydb. (club moss) and *Hesperostipa comata* (Trin. & Rupr.) Barkworth, occupy disturbed drier areas, and *Distichlis stricta* (L.) Greene (salt grass) and *Pascopyron smithii* (Rydb.) A. Löve (western wheat grass) in areas with increased sodicity (Vujnovic 1998).

Many of the aspen stands have been invaded by *Bromus inermis* Leyss. ssp. *inermis* (smooth brome), a result of cattle grazing (Holcroft Weerstra 2001). Recent vegetation and range health assessments have identified the presence of *Poa pratensis* L (Kentucky bluegrass) in parts of the native grassland, and predominance on oil and gas disturbances (Eastern Slopes Rangeland Seeds Ltd. 1995; Elsinger 2009). Neither species figured prominently in pre 1982 vegetation assessments (Wroe 1971; Bradley and Bradley 1977; Fehr 1982).

1.3.1 Plains Rough Fescue

Once treated as a single species, *Festuca scabrella*, Pavlick and Looman (1984) concluded that plains rough fescue (*Festuca hallii*) and foothills rough fescue (*Festuca campestris* Rydb.) are distinct species within the Subgenus *Leucopoa*. *Festuca hallii* chromosome count is $2n = 28$, while *F. campestris* is $2n = 56$ (Pavlick and Looman 1984). *Festuca hallii* usually occurs at elevations below 800 m from central Alberta to Western Ontario, in Montana, North Dakota and Colorado, rarely, and at higher elevations (Pavlick and Looman 1984). *Festuca campestris* range includes southern British Columbia, southwestern Alberta, Montana and northeastern Washington (Pavlick and Looman 1984), usually at elevations above 800 m (Looman 1982). *Festuca hallii* was once dominant throughout the grasslands of these regions (Moss and Campbell 1947; Coupland and Brayshaw 1953). If undisturbed, *F. hallii* will dominate grassland to the exclusion of other species, possibly as a result of its litter production, which cools the ground and retains moisture, increasing soil water early in the growing season (Vujnovic et al. 2000).

Festuca hallii grows in clumps 7 to 10 cm in diameter, with often 3 to 5 and rarely up to 10 culms, which are erect or slant at an angle of 70 to 80° from the horizontal (Moss 1994). Its grey green leaf blades are always tightly rolled; its panicles are erect and florets are dull green to moderately anthocyanic. It often has short rhizomes, a feature that distinguishes it from *F. campestris*. This long-lived perennial is an erratic seed-setter, seldom producing seed and with several or many years between seeding events (Johnston and MacDonald 1967; Toynbee 1987; Romo 1996).

Temperatures near 15 °C appear to be most favourable for germination of *F. hallii* (Romo et al. 1991). Spring seeding is recommended, when seedbed temperatures are increasing rather than in autumn when they are cooling, and when higher spring soil water favours germination (Romo et al. 1991). Optimal growth and re-growth following defoliation occur near 17 °C or below, with reduced growth above that temperature (King et al. 1998). Summer dormancy is triggered by moisture stress; as shown experimentally *F. hallii* did not enter dormancy, even at 27 °C when water was non-limiting (King et al. 1998). In areas with moist summer periods, *F. hallii* may mature later in summer, even up to the latter part of July (Horton 1992; Grilz et al. 1994).

Festuca hallii is sensitive to severe defoliation, especially during the early spring, its time of initial growth and when cattle appear to prefer it to most other species (Bailey 1970; Horton 1992; Best and Bork 2003). Plains rough fescue is better suited to late summer and autumn grazing, or a deferred rotation grazing system (Horton 1992). Slogan (1997) concluded light to moderate grazing may promote *F. hallii.* A comparison of grazed and un-grazed transplanted *F. hallii* at Elk Island, found grazing reduced tillers, seed heads and overall growth (Best and Bork 2003). Defoliation by fire may benefit *F. hallii,* depending on the season. Spring burning may increase tillering and stimulate inflorescence (Gerling et al. 1995) and fall burning may have no effect (Bailey and Anderson 1978). Late spring burning may reduce foliation and frequent burning may change plant community composition away from fescue grassland to a mixed-grass prairie.

Few attempts to restore *F. hallii* plant communities in the Parkland natural regions have been successful. Gas well sites and pipelines reclaimed in the parkland had fair to poor establishment of rough fescue and other native species from reclamation seed mixes and sod salvage (Petherbridge 2000; Elsinger 2009). A restoration experiment in the grasslands of central Saskatchewan resulted in the conclusion that conserving remaining rough fescue prairie rather than restoring it would be more realistic and successful (Clark 1998). Vujnovic et al. (2002) found *F. hallii* occupied undisturbed to moderately grazed areas. Undisturbed areas of fescue grassland exhibited a centrifugal organization, with *F. hallii* at the centre or core, and its presence decreased with the amount of disturbance, especially if combined with higher moisture. They concluded *F. hallii* was not able to dominate disturbed habitats (Vujnovic et al. 2002).

1.3.2 Other Native Grassland Species

Hesperostipa curtiseta is often co-dominant with *F. hallii* (Willms et al. 1986) and with grazing may become dominant (Moss 1955; Willms et al. 1986; Pantel 2006). Bailey (1970) found *H. curtiseta* is reduced by continual grazing. Nernberg and Dale (1997) and Otfinowski et al. (2007) found *H. curtiseta* was a poor competitor when grown with *B. inermis*. Like *F. hallii, H. curtiseta* responds poorly to disturbance (Redmann and Schwarz 1986).

Wheat grasses, such as *E. trachycaulus*, *E. lanceolatus* and *P. smithii*, occur naturally in rough fescue grassland and their cultivars are used in reclamation seed mixes. Wheat grasses are strong competitors and when seeded may persist and become dominant (Hammermeister 2001; Ostermann 2001). Disturbances do not affect these species, and may actually promote their growth (Redmann and Schwarz 1986; Pantel et al. 2011). In a seed mix experiment in the Central Parkland, Bush (1998) discovered that *E. trachycaulus* and *P. smithii* were well adapted and competitive. *Pascopyrum smithii* appears in early succession, in greater abundance than other perennials and remains longer (Dormaar and Smoliak 1985; Samuel and Hart 1994). Following an assessment of seeded pipeline recovery in the Dry Mixedgrass region, Hammermeister (2001) concluded that native wheat grass cultivars, such as *E. lanceolatus* and *E. trachycaulus*, dominated plant community development and suppressed establishment of other species.

Sedges (*Carex* spp.) are common in rough fescue grassland and are affected by disturbance, disappearing and requiring many years to recover (Inouye et al. 1987; Wang et al. 2006). Samuel and Hart (1994) found *Carex* spp. recovered only after 36 years in a Wyoming rangeland recovery. Light to moderate grazing causes an increase in *Carex* spp. (Willms et al. 1986); however, continual moderate to heavy grazing will reduce it. Otfinowski et al. (2007) found *B. inermis* encroachment into mixed-grass prairie reduced abundance of *Carex* spp. Annual forbs, such as *Artemisia frigida* Willd. appear in early succession following disturbance (Samuel and Hart 1994; Nasen 2009). *Artemisia frigida* increases with grazing pressure (Dormaar and Willms 1990; Willms et al. 1990; Slogan 1997).

Bromus inermis, an important agronomic forage crop introduced from Europe and Eurasia in the late 1880s, is a prolific seed setter, rhizomatous, grows faster than many native grasses and is a long-lived perennial (Romo et al. 1990; Blankespoor and May 1996). It was once commonly used in reclamation seed mixes, especially in livestock grazing areas, such as the Rumsey Block. Once established, smooth brome spreads rapidly, suppressing the growth and abundance of native flora, reducing wildlife habitat and natural diversity, transforming diverse plant communities into virtual monocultures (Grilz et al. 1994). In an assessment of a 25 year-old pipeline in the Central Parkland, Parker (2005) found that *B. inermis* persisted where it had been introduced on the right-of-way and that it had invaded into adjacent native pastures. Brown (1997) failed to eliminate smooth brome from rough fescue grassland with fire, mowing and glyphosate treatments. *Bromus inermis* may impede the existence of native species, such as *H. curtiseta* (Otfinowski et al. 2007).

Poa pratensis, commonly included in reclamation seed mixes, is considered introduced in Alberta, possibly arriving with European settlement (Tannas 2001). It is now endemic and often establishes from the seed bank when soil is disturbed (Brown 1997; Bizecki Robson et al. 2004; Adams et al. 2005). *Poa pratensis* readily produces seed and is strongly rhizomatous, allowing it to establish and spread rapidly. Blood (1966) and Slogan (1997) discovered that *P. pratensis* replaced *F. hallii* following heavy grazing. Desserud et al. (2010) and Naeth et al. (1997) found *P. pratensis* dominating pipeline disturbed sites and Tyser and Worley (1992) found *P. pratensis* replaced native vegetation in fescue prairie along roadside disturbances. Bush (1998) found heavily grazed pasture dominated by *P. pratensis* and *B. inermis.*

Another forage species, introduced from Eurasia in the 1930s and used in early reclamation seed mixes, is *Agropyron cristatum*. It is difficult to eradicate once established. A perennial bunchgrass, several characteristics contribute to its success, including a large number of tillers and prolific seed and litter production (Henderson 2005).

1.4 Geology and Soils

Non-marine Upper Cretaceous and Lower Tertiary Paskapoo sedimentary bedrock underlie most of the Rumsey Block. Older Upper Cretaceous bedrock of the Horseshoe Canyon formation commencing south of Drumheller and extending into the upper northeastern and eastern portions (Karpuk 1995; Hamilton et al. 1999). The Ecological Reserve in the northern section is partially underlain by the Edmonton formation, extending northward to Edmonton, and, which is part of the eastern flank of the Alberta Syncline. The bedrock is buried beneath 15 to 30 m of glacial deposits (Hamilton et al. 1999).

A principle feature of over 90% of the Rumsey area is hummocky disintegration moraine topography, known as knob and kettle, which is a complex of small depressions and hills. The remaining topography includes a glacial spillway, with alluvial deposits of sand and gravel, a small esker complex of ridges in one northwest section and a gentle undulating plain in the southeast corner (Hamilton et al. 1999). The hills are moraine plateaux formed by melting glacial debris deposition and rising up to 884 m above sea level. Sloughs average 857 m above sea level. Drainage is poor and many sloughs are dry by mid-summer. In the northern section of the Rumsey Block, the slopes are moderately steep (20 to 40%), with local relief of about 15 m. Toward the south, the topography becomes more gentle, with local relief diminishing to about 7.5 m (Karpuk 1995).

Soil was formed from glacial till of the Edmonton formation, brown to grey-brown in colour, sandy in texture with a low to medium calcium carbonate content. North of the Rumsey Block and towards Edmonton are found Black Chernozem soils while Dark Brown Chernozem soils occur in the south and towards Drumheller (Karpuk 1995). On well drained sites, a 15 cm deep brown organic layer is found. Black Chernozem soils are found in moister areas and under stands of poplar (Wroe 1971). Rego-Humic Gleysols have developed in areas that are water saturated for much of the year. Saline soils occur in the southern area, in more arid grasslands. Solonetzic soils frequently occur on moderately well to imperfectly drained fine textured soils between sloughs and in the shallow, central depressions of morainal plateaus (Karpuk 1995).

1.5 Rumsey History and Land Use

The original inhabitants of the area, aboriginal First Nations, used fire to control the movement of bison and may have impacted vegetation in the area (Wroe 1971). Fur traders and bounty hunters arrived in the 17th and 18th centuries because of the abundance of wildlife. Settlers started homesteading the land in the late 1800s and early 1900s. Bison and most other large ungulates disappeared from the area by about 1890 (Farley 1925). Settlement continued throughout the 1900s, with an abatement during the 1930s depression, and by 1960 much of the native vegetation had been altered by livestock grazing or destroyed through cultivation (Bird and Bird 1967). In spring 1980, a fire swept through the north east sections of the Ecological Reserve, and prior to that 1964 was the last recorded fire (Wroe 1971). Ranching began with the Imperial Ranching Company, which grazed in the vicinity of Rumsey Block as an open range for years, starting around 1895, and continuing today (Bradley and Bradley 1977). Other major dates are found in Appendix A.

1.6 Reclamation in the Rumsey Block

Grassland restorations are often unsuccessful as a consequence of unreliable seed sources, competition from weeds and agronomic species, and variation in weather (Wilson 2002; Desserud et al. 2010). Strategies preventing or reducing competition from non-native or weedy species includes burning, grazing or mowing and applying herbicides. Patton (1988) found increased seed production in *Festuca idahoensis* Elmer (Idaho fescue), *Achnatherum nelsonii* (Scribn.) Barkworth ssp. *dorei* (Barkworth & Maze) Barkworth (columbia needle grass) and *Pseudoroegneria spicata* (Pursh) A. Löve ssp. *spicata* (bluebunch wheat grass) in the years following a burn. Herbicide application to weedy species and introduced graminoids increased native grassland production and native grass density in two studies (Wilson and Gerry 1995; Masters et al. 1996). Ewing (2002) found greater long term (3 year) survival of *F. idahoensis* transplanted plugs in herbicide treated plots.

Reclamation practices associated with oil and gas activities have evolved over the past three decades in the Rumsey Block. Oil and gas pipelines and well sites were built with a variety of soil handling techniques: full right-of-way (ROW) stripping, whereby topsoil and subsoil were stripped off a 15 m right-of-way, stored and replaced following construction; and bucket-width (25 cm) stripping with topsoil salvage with either seeding or natural recovery. Until the 1970s, pipelines or well sites received little in the way of reclamation. Early reclamation seed mixes were predominantly introduced species, such as *B. inermis* or *P. pratensis*. Later seed mixes used native grass cultivars, principally wheat grasses, such as *E. trachycaulus*, *P. smithii* and *E. lanceolatus*. More recent reclamation either involved a native grass seed mix, or natural recovery. Today a philosophy of minimum disturbance by reducing surface impacts is held by most industry players, which facilitates promotion of natural recovery of rough fescue plant communities.

Recent surveys showed 34% of grassland communities in the Central Parkland natural region were dominantly non-native. Invasive plants were found in 42% of plains rough fescue communities. Roads and pipelines cause landscape fragmentation and may contribute to invasion of grasslands by noxious weeds and non-native species, through vehicular and equipment activity and non-native seed mixes (Hume and Archibold 1986; Tyser and Worley 1992; Bradley et al. 2002).

A survey of revegetation of fourteen industrial sites in the Rumsey Block, in 1991, resulted in varied cover: persistence of wheat grasses, such as *E. lanceolatus* or *P. smithii* from seed mixes; encroachment of *Phleum pratense* L. (timothy) or *B. inermis*; and natural recovery of rough fescue and other native species (Integrated Environments Ltd. 1991). The plant species composition of the majority of disturbed sites was not similar to the adjacent native range. A few exceptions occurred on linear disturbances, in particular the two pipelines, where natural recovery resulted in encroachment of *F. hallii*, *H. curtiseta* and *S. occidentalis* (Integrated Environments Ltd. 1991).

In 1995 a vegetation inventory was conducted on 25 well sites, three pipelines and five random control sites (Eastern Slopes Rangeland Seeds Ltd. 1995). Researchers concluded disturbances favour native forbs over native grasses, especially on pipelines. Linear disturbances are encroached by rhizomatous shrub species. Non-natives, such as *Cirsium arvense* (L.) Scop., *B. inermis* and *Festuca rubra* L. were found on many of the disturbances.

Elsinger (2009) conducted a survey of 57 well sites and pipelines and found that only six resembled a rough fescue grassland community of *Festuca hallii/Stipa curtiseta*. Twenty seven sites were dominated by wheat grasses: *E. lanceolatus* and *P. smithii*; 13 by *P. pratensis* and *P. smithii*; and 11 by introduced species: *B. inermis* and *P. pratensis*.

1.7 Grassland Restoration Techniques

1.7.1 Nitrogen Reduction

The ability of many native species to out compete introduced species in nitrogen impoverished soil may provide a potential reclamation path. Nitrogen is a key element in grassland ecosystems, because of its capacity to limit primary and secondary production (Dormaar et al. 1990). Grassland ecosystems typically have low nitrogen concentrations (Risser and Parton 1982) which may improve the competitive ability of native grasses over non-native species (Tilman and Wedin 1991; Alpert and Maron 2000). Prairie soils had less available nitrogen than adjacent aspen stands in an assessment of central parkland in Saskatchewan (Wilson and Kleb 1996).

Carbon enrichment is one method to reduce soil available nitrogen, by stimulating growth of soil microorganisms, which subsequently accumulate nitrogen in their biomass making it unavailable for plants (Morgan 1994). Some researchers observed a negative response of native grasses with carbon additions. Davis and Wilson (1997) reported the death of all plants with increasing amounts of added sugar. While finding a decrease in one weedy species following carbon enrichment, Seastedt et al. (1996) and Reever Morghan and Seasdedt (1999) reported no change in a native bunchgrass. A decline in non-native species was found by Morgan (1994) after adding sugar and sawdust to soil, by Davis and Wilson (1997) after adding sugar and by Cione et al. (2002) after adding a leaf and bark mulch.

Other researchers concluded that lower nitrogen concentrations improve the competitive ability of native grasses over invasive species. Classen and Marler

(1998) demonstrated that invasive annuals, such as *Bromus* species, were favoured over native perennials at higher nitrogen concentrations. Similarly, Wilson and Gerry (1995) found native seedling density decreased with increasing nitrogen. Morgan (1994) saw a decline in non prairie species after adding sugar and sawdust to soil. Blumenthal et al. (2003) found that carbon addition facilitated prairie species establishment, increasing total prairie biomass over a two year period, whereas nitrogen addition reduced total prairie biomass in both years. The opposite effect was found for weeds. In a soil impoverishment experiment, where all organic material was removed, *F. idahoensis* plugs had a higher survival rate over three years compared to those in a fertilized plot (Ewing 2002). Redmann et al. (1993) concluded higher concentrations of available nitrogen following fire, probably increased nitrogen concentration of *F. hallii* green shoots.

1.7.2 Natural Recovery

Natural recovery is influenced by species composition of adjacent grassland, site topography and grazing conditions. In a seeding and natural recovery experiment on a well site in the northern fescue region (Neutral Hills, Alberta) a natural recovery site was affected by its low slope position with a mesic moisture regime and the proximity of non-native species in the adjacent grassland. Ten years following reclamation, cover was predominately *B. inermis* with smaller amounts of *P. pratensis*, both favouring moist locations (Fitzpatrick 2005). Ten years of recovery on one seeded block resulted in predominantly *F. hallii*, with other native species such as *H. curtiseta, A. frigida* and *E. trachycaulus* making up the majority of the rest of the species. Another block had *F. hallii* and *E. trachycaulus* and many undesirable forbs, such as *C. arvense* (Fitzpatrick 2005). Natural recovery can work on small scale disturbances, but it may take a much longer time to reach the climax plant community (Woosaree and James 2006).

Naeth et al. (1997) monitored a pipeline right-of-way and adjacent native grassland in the central parkland. Forb abundance was initially greater on the disturbed treatments than the control in the first year, then declined over the next three years. Over the same period grass abundance increased, although the species differed between disturbed and undisturbed areas. *Festuca hallii* had reduced cover in the disturbed areas, while *P. pratensis* increased over four years. *Elymus trachycaulus* was dominant on the disturbed areas and non- existent in the control.

Forbs are frequently pioneer species, increasing following a disturbance, and then decreasing as long lived perennials become established (Naeth et al. 1997). In natural recovery, early seral species, the first to colonize a disturbed site, may be considered weeds and may not be desirable (Woosaree and James 2006). Woosaree and James (2006) found annual weeds such as *Axyris amaranthoides* L. (Russian pigweed) and *Cleomella* spp. (stinkweed) cover reached up to 31% in the first year following seeding and was even higher in natural recovery areas. They concluded these weeds were not a concern since they were annuals and would soon be replaced by perennial grasses. Arychuk (2001) found that early seed bank species emerging in disturbed sites were weedy species, such as *Chenopodium album* L. (lamb's quarters) or *C. arvense*.

Monitoring of a pipeline reclamation project in sandy soils in the Central Parkland showed succession of species from early seral forbs to mid and late seral over five years (Woosaree 2007). In years 1 to 3the site was dominated by *A. frigida* and *Artemisia ludoviciana* Nutt. ssp. *ludoviciana*. By year five these species decreased and several naturally occurring native species appeared on the site, including *Festuca saximontana* Rydb. (Rocky mountain fescue), *Rosa woodsii* Lindl. (woodland rose), *Arctostaphylos uva-ursi* (L.) Spreng. (bearberry) and various native forbs.

1.8 Arbuscular Mycorrhizal Fungi (AMF)

1.8.1 AMF and Plant Competition

The presence of arbuscular mycorrhizal fungi (AMF) may enhance the competitive ability of mycorrhizal plants; whereas that of non-mycorrhizal plants may be reduced, as long as the plants are competing for the same limited soil resources (Reeves 1985). Mycorrhizal fungi form mutualistic associations with plant roots, obtaining organic carbon from the host plant and providing nutrients to the plant via the hyphal network (Allen and Allen 1984). Mycorrhizae may cause physiological alterations in the host plant, including increased growth rates and seed production, increased water and nutrient uptake, increased drought tolerance, changes in hormonal balance and various morphological and anatomical changes (Allen and Allen 1984).

Mycorrhizae can affect nitrogen cycling by several means. Mycorrhizae increase water flow through plants which could increase uptake of dissolved nitrates (Allen et al. 1981; Allen 1991). Arbuscular mycorrhizal hyphae have the capacity to extract nitrogen directly from soil and transfer it to the plant (Ames et al. 1987; Barea et al. 1987; Allen 1991). In a comparison of nitrogen uptake of mycorrhizal and non-mycorrhizal plants both given nitrogen fertilizer, Ames et al. (1984) determined that while both plants took up nitrogen from a fertilizer source, only mycorrhizal plants obtained nitrogen from the surrounding soil.

The role of AMF appears to increase in importance with decreasing concentrations of plant available nutrients (Read et al. 1976). In natural ecosystems where ammonium (NH_4^+) is the primary source for plant-available nitrogen, AMF may play an important role in transporting nitrogen to the rhizosphere (Ames et al. 1984). Following a nitrogen reduction experiment in native grassland, Jonasson et al. (1996) reported no change in AMF colonization of *F. rubra*, while other grasses showed a significant reduction and many forbs disappeared completely, concluding AMF assisted nitrogen uptake. Where soil nitrogen concentrations are high, e.g. cultivated lands, the role of mycorrhizae may diminish (Ames et al. 1983; Ryan and Ashe 1999).

1.8.2 AMF and Topsoil Disturbance

Disturbance, such as a stripped well site, can reduce AMF activity, causing spores to die upon exposure to air (Liberta 1981). Topsoil storage further reduces AMF presence (Allen and MacMahon 1985; DeGrood et al. 2005). While AMF spores are known to survive when buried, their presence decreases with depth of soil cover and length of storage; therefore, topsoil handling techniques may determine the extent of re-establishment of AMF microorganisms (Allen and Allen 1984;

Allen and MacMahon 1985). For up to two years of topsoil storage there may be no discernible change in AMF propagules; however, they start to decrease in the third year (Miller et al. 1985). Rives et al. (1980) demonstrated that 3-year storage of topsoil reduced viable AMF inoculum by 8 to 10 times less than undisturbed soil. Where spore counts are low, early successional non-mycotropic annuals may have an advantage over plants dependant on mycorrhizae, thus slowing the rate of secondary succession (Allen and Allen 1980).

While topsoil storage reduces AMF spores, it increases nitrogen content. Ammonium concentrations rise with increasing stored topsoil depth because of increasing anaerobic nitrification (Abdul-Kareem and McRae 1984). Conversely, nitrate (NO_3^-) concentrations do not change with depth or time of topsoil storage (Abdul-Kareem and McRae 1984).

1.8.3 AMF Influence on *Festuca hallii*

The only literature found regarding arbuscular mycorrhizal fungi infection in *F. hallii* was that of Molina et al. (1978). Following an analysis of *Festuca* spp. from western USA and Canada, they concluded that all plants examined were mycorrhizal. Their collection included *F. scabrella* from British Columbia and Alberta. Based on the elevation of their locations, these plants were probably *F. campestris* at elevations higher than 800 m and *F. hallii* at lower elevations (Looman 1982).

Two grasses, which occupy similar ecological niches to *F. hallii*, *B. gracilis* and *P. smithii*, are facultatively mycorrhizal, with variation in response to AMF infection depending on the ecological situation of the plants (Allen et al. 1984). *Festuca ovina* L. (sheep fescue) performs a similar role as *F. hallii* in natural grasslands in southwestern England, a long-lived forage producer for sheep. AMF analyses of *F. ovina* demonstrate it is facultatively mycrotrophic, acquiring AMF infection in nutrient-poor situations, and producing better growth than non-infected plants in similar nutrient conditions (Koucheki and Read 1976; Read et al. 1976).

1.8.4 AMF Influence on *Poa pratensis*

Experiments with AMF and *P. pratensis* are inconclusive, indicating it may have a facultative dependence on AMF infection or it may have a negative reaction to AMF presence. In controlled AMF inoculation experiments with tall-grass prairie species, *P. pratensis*, while being infected, had significantly less infection than native grasses (Johnson et al. 1992; Ahn-Heum et al. 2000). In a comparison of AMF in soils surrounding five grass species, Johnson et al. (1992) found that other native grasses had almost 30% more AMF than *P. pratensis*. Giovannetti et al. (1988) found *P. pratensis* had significantly less colonization of a single AMF, *Glomus monosporum*, compared to several forbs in a controlled experiment. *Poa pratensis* biomass may increase following AMF reduction. In tall-grass prairie field experiments where herbicide (Benomyl) application removed most fungi, native tall grass graminoids had significant reductions in biomass without their obligate AMF; whereas, *P. pratensis* biomass increased by up to 150% with no AMF infection (Wilson and Hartnett 1997; Hartnett and Wilson 1999). *Poa pratensis* does not always react negatively to AMF. Hart and Reader (2002) found a positive response of *Poa pratensis* to three mycorrhizae fungi families, *Glomaceae*, *Gigasporacea* and *Acaulosporaceae*, and Wetzel et al. (1996) found AMF infection in all sampled *P. pratensis* in a prairie wetland.

1.9 Summary of Knowledge Gaps

1. Prairie soils are known to be nitrogen-deficient, and as a successful prairie species, *F. hallii* should tolerate nitrogen-depleted soils, while *P. pratensis* and *B. inermis*, known to thrive in high nitrogen conditions, may not. Displaying a competitive advantage of *F. hallii*, by manipulating nitrogen content in reclaimed soils, could lead to an effective reclamation technique to assist recovery of *F. hallii* following disturbance.
2. Natural recovery is increasingly being used by industry as a reclamation technique; nevertheless, does no intervention expose a disturbance to colonization and dominance by undesirable species? Understanding what can

be expected during natural recovery will aid in predicting when the trend is toward desirable species, or when intervention is required. *Festuca hallii* responds poorly to seeding; therefore, would natural recovery, relying on seed bank propagules and intact sod, promote *F. hallii* recovery?

3. Obtaining reliable seed sources for native species is a problem for grassland reclamation. Erratic seed production of *F. hallii* makes it especially difficult to obtain seed. One possibility is utilizing native hay as seed source for reclamation. Native hay has been successfully applied as a reclamation seed source in semi-arid sites in Saskatchewan; however, it has not been attempted in the central parkland. The quality of much of the rough fescue grassland in Rumsey, a mix of desirable native species, makes it a candidate for this technique, especially when a disturbance is adjacent to high quality grassland.
4. One possibility for the poor recovery of *F. hallii* may be competition from other seeded species, especially aggressive cultivars. Success for a slow-growing species such as *F. hallii* may be increased with lower seeding rates of aggressive species or with monoculture seeding. Monoculture seeding is unlikely to result in a community resembling pre-disturbance conditions, and may result in bare ground, increasing potential for erosion and invasion by non-native species. Nevertheless, reduced competition may promote establishment of *F. hallii* if seeded as a monoculture.
5. *Festuca hallii* is known to be associated with AMF species; however, its dependence, whether obligatory or mutualistic, is unknown. While this is an intriguing ecological question, the impact of topsoil handling on fungal activity could affect recovery potential of *F. hallii*. If AMF are obligatory for *F. hallii* growth, then low levels of the fungi in stored topsoil would impede its recovery. Non-reliance on AMF would mean topsoil handling to enhance fungal activity would have little effect on *F. hallii* recovery.
6. The full impact of oil and gas activity and grazing management practices in the Rumsey Block is not known. The number of well sites, pipelines, access roads and cattle disturbances, such as dugouts and corrals, in Rumsey Block has been documented and spatially recorded. Nevertheless, effects of those

disturbances on the composition and structure of the rough fescue grasslands has not been determined. Understanding changes in state from original rough fescue grassland, as a result of disturbance, and trends over time would allow land managers to make better decisions when confronted by new developments.

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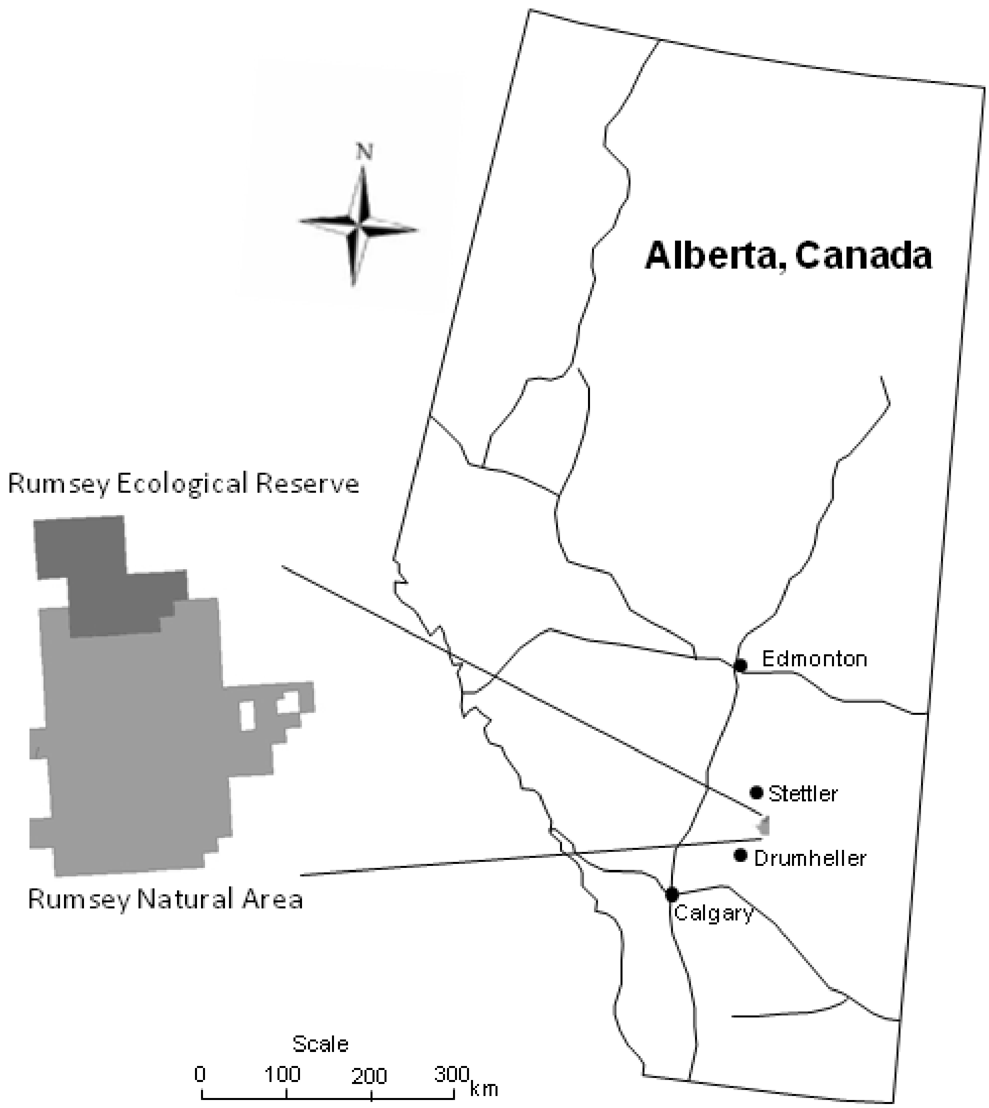

Figure 1.1 Location of the Rumsey Block, comprised of the Rumsey Ecological Reserve and Rumsey Natural Area.

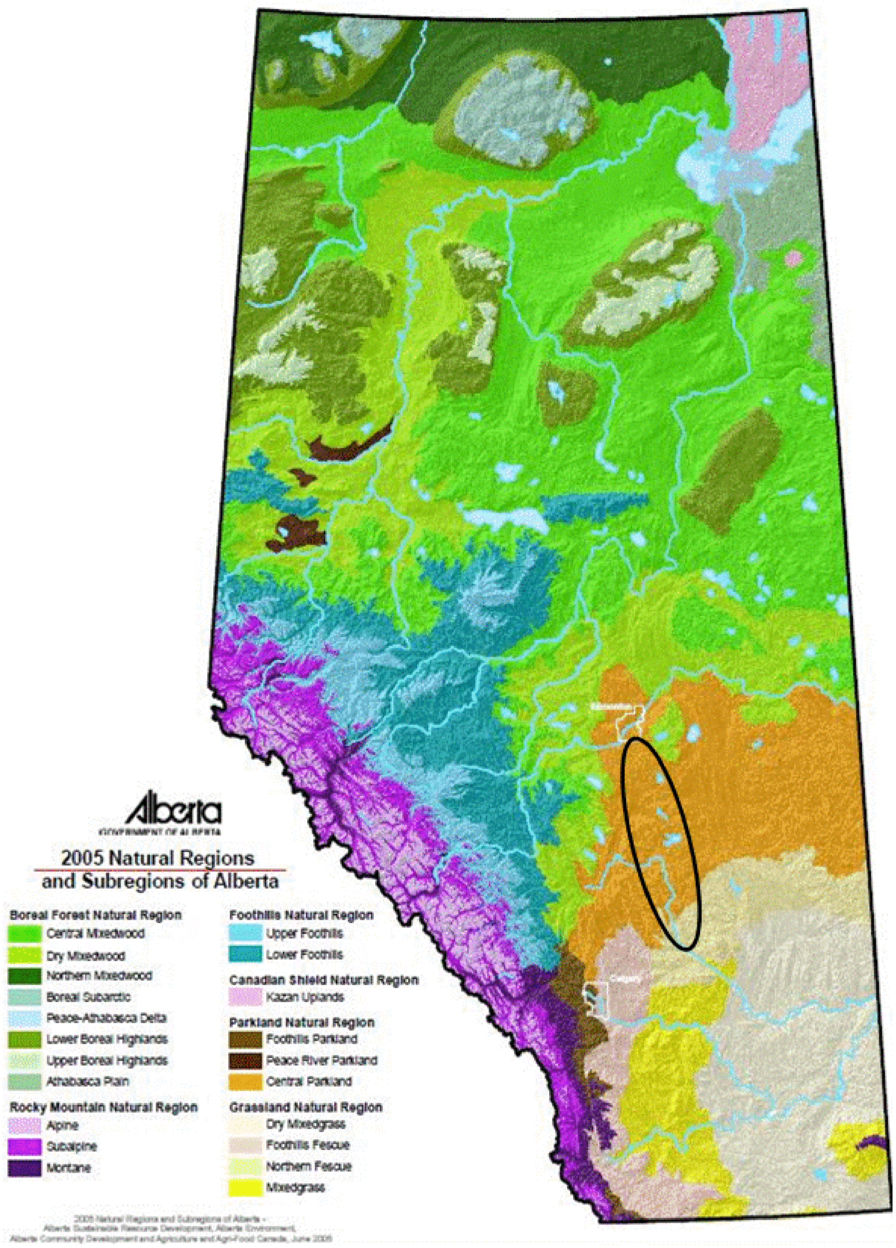

Figure 1.2 Study area (circled) encompassing the Central Parkland subregion of the Parkland Natural region and the Northern Fescue subregion of the Grassland Natural region.

CHAPTER 2. ESTABLISHMENT OF *FESTUCA HALLII* AND *POA PRATENSIS* ON DISTURBED LAND USING STRAW-AMENDED SOIL

2.1 Introduction

Grassland portions of the aspen parkland in western North America were originally fescue prairie, stretching west from central Saskatchewan to the Rocky Mountain foothills in Alberta (Coupland 1961), and dominated by *Festuca hallii* (Vasey) Piper. Rich Black Chernozemic soils in the area attracted agriculture in the late 1880s, and today, as a result of cultivation, less than 5% of the original prairie dominated by *F. hallii* remains (Grilz et al. 1994). Oil and gas development, commencing in the 1940s, combined with cultivation, resulted in small remnant parcels of fescue prairie. Once disturbed, *F. hallii* recovers poorly, and fescue prairie is often colonized by *Poa pratensis* L. to the exclusion of native species (Brown 1997; Slogan 1997; Desserud 2006).

Festuca hallii, a perennial bunch grass, is slow growing and long lived, requiring two to three years to become established from seed. This species is an erratic seed-setter, seldom producing seed and with several or many years between seeding events (Johnston and MacDonald 1967; Toynbee 1987; Romo 1996). On the other hand, *P. pratensis* is an exotic grass in North America, presumably arriving with European settlement (Tannas 2001). It is now naturalized in Alberta and often establishes from the seed bank when soil is disturbed (Adams et al. 2005). Unlike *F. hallii*, *P. pratensis* readily produces seed in its first and subsequent years, and is strongly rhizomatous, allowing it to establish and spread rapidly (Best et al. 1971; Tannas 2001).

The ability to restore fescue prairie, including *F. hallii*, has become an important consideration for oil and gas companies in Alberta, which are required by law to return disturbed land in native grassland to within 15% of pre-disturbance species distribution and cover (Alberta Environment 2010). Disturbance caused by oil and gas well site construction may initially increase nitrogen availability (Dormaar and Willms 1990) due to organic matter mineralization. Furthermore, soil admixing may decrease soil organic carbon and potassium, and increase clay content, phosphorus, pH and electrical conductivity (de Jong and Button 1973; Culley et al. 1982; Naeth et al. 1987; Hammermeister et al. 2003). These effects alter the competitive relationships among plant species and may drive succession in an undesirable direction.

One method to aid recovery of fescue grassland may be soil nitrogen impoverishment. Grassland ecosystems typically have low nitrogen concentrations (Risser and Parton 1982), which may improve the competitive ability of native grasses over non-native species (Tilman and Wedin 1991a; Alpert and Maron 2000). *Poa pratensis* leaf and root biomass increases with ammonium nitrate fertilization (Christians et al. 1979; Bowman et al. 1989). While no data were found regarding reaction of *F. hallii* to nitrogen concentrations, Smith et al. (1968) noted *Festuca campestris* Rydb. did not respond to ammonium-nitrate fertilizer, having similar dry matter yield with or without it. Johnston et al. (1967) found that increasing ammonium-nitrate fertilizer reduced the basal area and dry matter yield of *Stipa comata* Trin. and Rupr. var. *comata* (needle and thread grass) and *Koeleria macrantha* (Ledeb.) J.A. Schultes (June grass), which often co-exist with *F. hallii.* Classen and Marler (1998) found invasive annuals, such as *Bromus* species, were favored over native perennials at higher nitrate concentrations. In a soil impoverishment experiment, where all organic material was removed, Ewing (2002) found plugs of *Festuca idahoensis* Elmer had a higher survival rate over three years compared to those treated with a nitrogen-phosphorus-potassium fertilizer.

Carbon enrichment can reduce soil available nitrogen, by stimulating growth of soil microorganisms, which can immobilize nitrogen making it unavailable for plants (Morgan 1994). Wilson and Gerry (1995) found native seedling density, in sawdust-treated soil, decreased with increasing nitrogen concentrations. A decline in non-native species was found by Morgan (1994) after adding sugar and sawdust to soil, by Davis and Wilson (1997) after adding sugar, and by Cione et al. (2002) after adding a leaf and bark mulch. Davis and

Wilson (1997) reported the death of all plants with increasing amounts of added sugar, in an experiment seeding five native wetland prairie species, including *Beckmannia syzigachne* (Steud.) Fernald and *Danthonia californica* Bol., and six non-native species, including *Agrostis capillaries* L. and *Anthoxanthum odoratum* L. Seastedt et al. (1996) and Reever-Morghan and Seasdedt (1999) found a decrease in a non-native, *Lepidium densiflorum* Schrad., following carbon enrichment with sugar and sawdust, but no change in *Pascopyrum smithii* (Rydb.) A. Löve. Cione et al. (2002) found little effect of leaf and bark mulch on the establishment of seeded native shrubs such as *Artemisia californica* Less., *Salvia apiana* Jeps., and *Salvia mellifera* Greene. Blumenthal et al. (2003) discovered that sugar and sawdust addition facilitated establishment of a mix of 11 prairie grasses and forbs, including *Androopogon gerardii* Vitm., *Schizachyrium scoparium* (michx.) Nash and *Heliopsis helianthoides* (L.). The opposite effect was found for nine seeded non-native species, including *Cirsium arvense* (L.).

Prairie grasses tolerance of low nitrogen caused by carbon addition, may be the result of an arbuscular mycorrhizae fungi (AMF) mutual relationship, whereby fungi assist with nitrogen uptake, such as ammonium ($_{NH4+}$) (Molina et al. 1978; Allen 1991). On the other hand, *P. pratensis* may not be affected by AMF infection (Giovannetti et al. 1988; Johnson et al. 1992; Wilson and Hartnett 1997; Ahn-Heum et al. 2000).

While *F. hallii* and *P. pratensis* may occupy similar ecological niches, prairie evolution, including AMF infection, might allow *F. hallii* but not *P. pratensis* to tolerate nitrogen depleted soil. Finding a competitive advantage for *F. hallii* could lead to an effective reclamation technique to assist its recovery following disturbance. This research tested the hypothesis that *F. hallii* will tolerate nitrogen depleted soil, through the addition of straw as an amendment to newly reclaimed well sites, while *P. pratensis* will not. The second hypothesis is that *F. hallii* is negatively affected by admixed soil, which results in lower organic carbon and potassium, phosphorus, pH and electrical conductivity, while *P. pratensis* may tolerate such soil changes. Past failures of *F. hallii* recovery and successful invasion of *P. pratensis* after disturbance could be explained by exploring differences in soil and vegetation properties that might impact each species differently.

2.2 Methods

2.2.1 Study Sites

Three field sites were established in 2007 in central Alberta, Canada: two in the Central Parkland natural region at Ellerslie (53° 25′ N, 113° 29′ W) and Byemoor (51° 59′ N, 112° 19′ W) and one in the Northern Fescue sub-region of the Grassland natural region at Drumheller (51° 26′ N, 112° 21′ W). Elevation at each site is approximately 660, 900 and 900 m above sea level, respectively. Soils at Ellerslie are Orthic Black Chernozems while at Byemoor and Drumheller they are Dark Brown Chernozems. Native grassland vegetation in both regions is dominated by *F. hallii* associated with *Hesperostipa curtiseta* (A.S. Hitchc.) Barkworth (western porcupine grass) in the Central Parkland, and *Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths (blue grama grass) in the Northern Fescue sub-region.

During the research period between June 2007 and July 2009, temperature maximums were 36 °C, with growing season (April to October) temperatures averaging 12 °C. Average annual rainfall in this period was 510 mm at Ellerslie, 302 mm at Byemoor and 270 mm at Drumheller. Seeding occurred in July 2007. June 2007 was abnormally wet at all sites. July 2007 was abnormally dry at Byemoor and Drumheller, with normal precipitation in August. Rainfall was average in 2008 while while 2009 was abnormally dry in April, May and June at all sites (Figure 2.1).

The Ellerslie site was 1 ha in size and located in a previously cultivated area. Prior to this experiment it was fallowed for 3 years, and in May 2007 it was sprayed with glyphosate (Roundup Weathermax®) at 0.4 l/ha with a Flexicoil sprayer. Topsoil is approximately 23 cm deep. The Drumheller site was an abandoned natural gas well site (0.5 ha) in uncultivated land. In June 2007 it was cleared of topsoil, flow-line piping was removed, the well capped and topsoil replaced to a depth of approximately 11 cm. The Byemoor site (1 ha) was in uncultivated land. The site was levelled and cleared of topsoil in preparation for gas well drilling in 2006, then re-contoured in 2007 and topsoil replaced to a depth of approximately 30 cm.

2.2.2 Festuca *hallii* Seed Collection

Festuca hallii seed was collected in July 2006, in rough fescue grassland in the Rumsey Natural Area (51° 50′ N, 112° 33′ W), in central Alberta with a combine (Wintersteiger Plot Master Elite®) set at a height of approximately 30 cm. Seed was air dried at 25 °C, the chaff removed and *F. hallii* seed separated from other grasses. One hundred randomly selected seeds were distributed equally among ten closed Petri dishes on 1 mm thick germination paper, wet with distilled water and incubated in the dark at 20 °C (Romo et al. 1991). After six weeks 95% germination was achieved and the seeds were deemed viable.

2.2.3 Site Preparation

In July 2007, the three sites were treated with straw amendments. One-year-old certified weed-free straw was applied at three rates: approximately 1 kg/m^2 (high), 0.5 kg/m^2 (low) and no straw (control). Large round bales of barley straw were chopped with a Hesston 25 bale processor at Drumheller and wheat straw with a Highline BP7800 at Ellerslie and Byemoor, each running at approximately 2 km/hr. Chopped straw was spread in strips that were 6 to 7.5 m wide and 72 to 90 m long. Uneven straw applications were manually raked prior to rototilling. Straw was incorporated 8 to 10 cm into the soil with a Howard 2 m-wide rototiller, pulled by a 50 hp tractor running at 1800 rpm and 2.4 km/hr. Control strips were also rototilled. Four replications were laid out in a fully randomized design at Ellerslie and Byemoor. However, to retain equivalent plot sizes (approximately 6 x 7 m) only two replications of straw were laid out at Drumheller.

Seeding occurred in July 2007 within one week of straw application. Seed was placed at a depth of 1.2 to 1.9 cm with a Brillion® seeder and packer with double disk openers. Ambient precipitation, air and soil temperature, and relative humidity were recorded hourly on site, with a Campbell Scientific weather station at Ellerslie and Byemoor, and an Onset Hobo Micro Station at Drumheller. Rows of *F. hallii* (15.5 kg/ha, 1,300 live seeds/m) and *P. pratensis* (1.7 kg/ha, 775 live seeds/m) were seeded perpendicular to and across the straw amendment treatments. Four replications of each treatment were laid out in a fully randomized design at each site.

In September 2007, all sites were mowed with a tractor-mounted hay swather, to remove volunteer wheat and barley applied with the straw. In June 2008, the Ellerslie site was mowed again, the cut biomass removed, and the site treated with a broad-leaf herbicide (Embutox 625), at rate of 0.4 l/ha using a Flexicoil sprayer to remove non-native species, such as *Cirsium arvense* (L.) Scop. (Canada thistle), *Monolepus nuttalliana* (Schultes) Greene (goosefoot) and *Convolvulus arvensis* L. (bindweed). These species produced a canopy cover that was greater than 20% and sufficiently high to alter establishment of seeded species (Stohlgren et al. 1999).

2.2.4 Experimental Design

Straw was applied in 6 to 7 by 90 m strips, in 3 rates, randomly replicated 4 times at Byemoor and Ellerslie, and 2 times at Drumheller. Seeding was applied perpendicular to the straw treatments, in 6 to 7 by 90 m strips at Byemoor and Ellerslie, and in 6 to 7 by 45 m strips at Drumheller. Seeding included 2 species, randomly replicated 4 times (Figure 2.2). This resulted in a split-block of 3 sites with 96 randomized experimental subplots at Byemoor and Ellerslie (3 [straw treatments] x 2 [species] x 16 [straw treatment by species]) and 48 at Drumheller (3 [straw treatments] x 2 [species] x 8 [straw treatment by species]).

2.2.5 Vegetation Sampling

The seeded vegetation was sampled, in late July and early August, 2008 in randomly selected subplots (6-7m^2); the same subplots were re-sampled in 2009. Seven subplots of each seeding by straw treatment were randomly selected at Byemoor and Ellerslie (42 subplots) and five at Drumheller (30 subplots) (Figure

2.2). All sampling incorporated a 50 cm wide buffer around each subplot to avoid edge effects.

In each subplot, only *F. hallii* or *P. pratensis* were sampled. Volunteer specimens of *P. pratensis* in *F. hallii* seeded subplots were ignored. No volunteer *F. hallii* was found in *P. pratensis* seeded subplots. Within each quadrat, up to five specimens of seeded *F. hallii* or *P. pratensis* were randomly selected and measured for leaf length, number of inflorescences and vegetative tillers. In each subplot, three randomly selected specimens of the seeded species were excavated with a hand shovel. For *F. hallii* cuts equivalent to the circumference of the leaf canopy, were made around the plant and to a depth of approximately 20 cm, to include most roots (Johnston 1961). For *P. pratensis,* a rhizomatous species, cuts were made around the plant, approximately half way to any adjacent *P. pratensis* plants, to encompass rhizomes, and to a depth of approximately 20 cm, to include most roots (Donkor et al. 2002; Holechek et al. 2004). Plants, including intact roots, were stored in plastic Ziploc bags at 5 °C in a refrigerator until washing. Whole plants were washed with running tap water, roots and crowns were removed and washed thoroughly to remove of all traces of soil. Leaves and roots with crowns were oven dried, separately, for 48 hours at 60 °C then weighed to determine dry biomass.

2.2.6 Soil Analyses

Soil was sampled at three locations per subplot, adjacent to the excavated plants, in both 2008 and 2009, to a depth of 10 cm using a hand auger (5 cm dia.). The three samples of each subplot were composited, resulting in 42 soil samples from Byemoor and Ellerslie, and 30 from Drumheller. Samples were stored at 5 °C in a refrigerator until they were analyzed. Volumetric soil water was measured in June, July and August in 2008 and 2009 at three random locations per subplot, using a ThetaProbe ML2x® at 0 and 15 cm depths, resulting in 126 soil water measurements from Byemoor and Ellerslie, and 90 from Drumheller.

Soil was sampled in the final year (2009) of the research in undisturbed grassland within 15 m of Byemoor and Drumheller site boundaries. Four samples were taken 25 m apart along each side of the sites with a hand auger (5 cm dia.) to a depth of 10 cm. Every two adjacent samples were combined for a total of 8 samples from each site and stored at 5 °C in a refrigerator until analyzed.

Soils were analyzed for total carbon (C) using the LECO combustion method (Nelson and Sommers 1996), total soil nitrogen (N) using the Dumas method (Bremner (1996), available ammonium (NH_4^+) using the 2.0M KCl procedure (Maynard and Kalra 1993a) and available nitrate (NO_3^-) with dilute calcium chloride solution extraction (Maynard and Kalra 1993b). Soil electrical conductivity (EC) and pH were measured with a 1:2 soil:water extraction (Carter and Gregorich 1993b; Carter and Gregorich 1993a). Available phosphate (PO_4^-) and potassium (K^+) were analyzed with a modified Kelowna extraction (Qian et al. 1994). Soil texture was determined manually (Day 1965).

2.2.7 Statistical Analyses

Data were analyzed as a fully randomized split-block design of three field sites by three treatments. The two species were not compared due to differences in growth characteristics. Data were found normally distributed with Shapiro-Wilk analysis. Variations of growth responses among sites and treatments were analyzed with multivariate general linear model (GLM). Growth responses and soil variable differences among and within sites and treatments were tested by univariate analysis of variance (ANOVA) with Tukey's post-hoc test. Differences between years were tested with student t-tests.

Relationships among growth responses and soil variables were assessed with Pearson product correlation and linear regression. Correlations were made to prior- and current-year soil variables for *F. hallii* which is slow establishing and current year for *P. pratensis* which establishes rapidly. *Festuca hallii* requires two or more years to establish. Perennial grasses store carbohydrate reserves in late summer, making prior-year water and nutrients important growth factors (Holechek et al. 2004). Johnston and MacDonald (1967) found *F. hallii* develops vegetative apices initiating floral primordia in late August to early September. *Poa pratensis* matures quickly, producing seed the first and subsequent years.

Significant variables were ranked with stepwise multiple regression. Data analyses employed IBM® SPSS® Statistics (version 18, SPSS, Chicago IL) and Microsoft® Excel© 2007 (Microsoft Corporation, Redmond, Washington).

2.3 Results

2.3.1 Straw and Site Variations

Festuca hallii responded positively to straw treatments, while there was little effect on *P. pratensis* (Table 2.1). At all sites, straw treatments increased *F. hallii* biomass, root biomass, leaf length and cover (Figure 2.3) but had no effect on *P. pratensis* (Figure 2.4). Site differences were found for both species. *Festuca hallii* 2009 biomass, root biomass, leaf length and cover were lowest at Byemoor and greatest at Ellerslie (Figure 2.3). *Poa pratensis* 2009 biomass and leaf length were greatest at Byemoor and lowest at Drumheller (Figure 2.4).

At Byemoor and Drumheller, *F. hallii* benefited from straw amendments, having greatest leaf length, root biomass and cover in high straw treatments (Figure 2.3). *Festuca hallii* shoot to root values decreased at Drumheller and Ellerslie, and increased slightly at Byemoor. Similar results were observed in high straw treatments, with greater *F. hallii* leaf length, tiller density and cover where unincorporated straw remained on the surface. No significant differences among straw treatments were found for *F. hallii* at Ellerslie. *Poa pratensis* response to straw was opposite to that of *F. hallii*, with greater leaf length and root biomass in no-straw treatments at Byemoor and Ellerslie (Figure 2.4 and Table 2.2). In 2009 *P. pratensis* did not vary among straw treatments at all sites (Table 2.1). Site differences were found for soil water, total C, total N, K, NH_4^+, NO_3^-, pH and EC (Table 2.1 and Table 2.2).

2.3.2 Soil Water

Soil water was significantly different among sites in both seeded treatments (Table 2.1 and Table 2.2), with Byemoor having the most summer soil water, and Drumheller having the least (Table 2.3). Soil water was significantly greater in high straw treatments in June 2008 at Byemoor (P=0.012) and July 2008 at Drumheller (P=0.013). In 2008 at Byemoor, June and total available soil water were significantly higher (Table 2.3) and *F. hallii* biomass, root biomass, leaf length and cover, were lower than at the other two sites (Figure 2.3). Despite having had the least 2008 mid-summer soil water (Table 2.3), Drumheller *F. hallii* root biomass was the greatest, and leaf length and cover were equivalent to the greater of the other two sites (Figure 2.3).

Festuca hallii biomass increased with previous July soil water at Byemoor and Drumheller, and cover increased at Ellerslie (Figure 2.5). No differences in soil water among straw treatments were found for *P. pratensis* in 2008 or 2009 (Table 2.4). *Poa pratensis* responded to site differences, increasing leaf length with current-year June soil water when all sites were combined (Figure 2.6). No correlation to prior-year site soil water was found. Ellerslie had the longest *P. pratensis* leaf length and highest June 2009 soil water; while Drumheller had shortest leaf length and lowest June 2009 soil water (Figure 2.4 and Table 2.4).

2.3.3 Carbon and Nitrogen

No significant differences were found for total soil C and N among the straw treatments for either *F. hallii* or *P. pratensis*; however, C and N concentrations were significantly different among sites in both seeding treatments, with Ellerslie having the highest and Byemoor the lowest (Table 2.3 and Table 2.4). With all sites combined, *F. hallii* biomass, leaf length and cover correlated positively to prior- and current-year C and N (Table 2.5). *Poa pratensis* leaf length and cover correlated positively to current-year C and N (Table 2.6). At Ellerslie which, had the highest C and N in prior and current years (Table 2.3 and Table 2.4), *F. hallii* had greatest biomass, root biomass and leaf length (Figure 2.3) and *P. pratensis* had greatest root biomass and leaf length (Figure 2.4).

2.3.4 Ammonium

In *F. hallii* treatments, NH_4^+ concentrations dropped between 2008 and 2009, significantly at Drumheller ($P < 0.001$) and Ellerslie, and were significantly different among sites in both years (Table 2.3). At Drumheller and Ellerslie NH_4^+ concentrations were greater in the high straw treatments and dropped between 2008 and 2009 (Figure 2.7); with a significant drop at Drumheller (16%, P = 0.046). Ammonium was significantly different among sites in *F. hallii* treatments in 2008 and 2009 (Table 2.1).

In *P. pratensis* treatments, NH_4^+ concentrations dropped between 2008 and 2009 at Drumheller and Byemoor and were significantly different among sites. High straw treatments had the greatest NH_4^+ concentrations at all sites and decreased between 2008 and 2009; however, the differences were not significant. No correlation was found between NH_4^+ and *P. pratensis* biomass, root biomass, leaf length or cover.

2.3.5 Nitrate

In *F. hallii* treatments, NO_3^- concentrations were similar among the straw treatments at each site, rising between 2008 and 2009, although with no statistically significant difference. Combining straw treatments, NO_3^- rose between 2008 and 2009, although there was no statistically significant difference among sites in both years (Table 2.3). No correlation between *F. hallii* biomass, root biomass, leaf length and cover and NO_3^- appeared among or within sites.

In *P. pratensis* treatments, nitrate differed significantly among sites in 2008 and 2009 (Table 2.4) and *P. pratensis* responded to varying NO_3^- across the sites. *Poa pratensis* biomass and root biomass correlated positively to current-year NO_3^-, while leaf length correlated negatively (Table 2.6**)**.

2.3.6 Potassium and Phosphate

In the *F. hallii* treatments in 2008, high straw treatments had the greatest K^+ concentrations at Drumheller and Byemoor (P=0.007), with a significant decrease at Drumheller between 2008 and 2009 ($P < 0.001$). Potassium was significantly different among sites in *F. hallii* and *P. pratensis* treatments, in 2008 and 2009, increasing between 2008 and 2009 (Table 2.3 and Table 2.4). *Festuca hallii* biomass (Figure 2.7) and leaf length increased as current-year K^+ decreased (Table 2.3).

In *P. pratensis* treatments, no difference in potassium concentrations was found among straw treatments at all sites, and potassium increased between 2008 and 2009. Site differences in K^+ concentrations were found and *P. pratensis* leaf length correlated to lower K^+ levels (Table 2.6). The lowest K^+ occurred at Ellerslie in 2008 and 2009 and next at Byemoor (Table 2.4). Phosphate concentration was significantly higher in disturbed soils than in adjacent undisturbed soils at Byemoor ($P < 0.001$) and Drumheller ($P = 0.010$; Table 2.7). At both sites, there was no significant difference in PO_4^- among straw treatments.

2.3.7 Electrical Conductivity and pH

Soil pH and EC were not affected by straw treatments in either seeding treatments at all sites. Among sites, pH and EC differed significantly in both seeding treatments (Table 2.3 and Table 2.4). Byemoor had significantly highest pH and EC and Ellerslie had the lowest. Of Byemoor sampling plots, 19% were clay textured, 54% were clay-loam and 27% were loam; at Ellerslie 100% were clay-loam; and at Drumheller 100% were loam. Byemoor soil pH ($P < 0.001$) and EC ($P = 0.003$), in both seeding treatments, were significantly higher than adjacent undisturbed grassland values (Table 2.7). At Drumheller, undisturbed grassland pH and EC were similar to that of disturbed site treatments.

Festuca hallii biomass among the sites decreased with higher pH and EC (Figure 2.8). Byemoor had significantly higher EC than the other sites and lower *F. hallii* growth parameters (Figure 2.8 and Table 2.3). *Poa pratensis* biomass increased at higher pH and EC (Figure 2.8).

2.3.8 Significant Soil Variables

The most significant variables affecting *F. hallii* biomass were prior- and current-year June and total soil water, prior-year C and N and current-year NH_4^+, K, pH, and, EC. Of these, 64% of the biomass increase could be explained by a reduction in current-year pH and EC and an increase in prior-year C ($P = 0.016$). For *F.*

hallii leaf length, the most significant variables were prior- and current-year June soil water, 2008 C and N and current-year NH_4^+, K, pH and EC. Of these, 79% of leaf length increase may be explained by a decrease in current-year EC and an increase in prior-year June soil water (P = 0.031).

The most significant variables affecting *P. pratensis* biomass were current-year June soil water, potassium, nitrate, pH, EC and total C and N. Of these, 54% of the biomass increase could be explained by an increase in current year EC and NO_3^- (P = 0.038). For *P. pratensis* leaf length, the most significant variables were current-year C and N, nitrate and potassium and 89% of the leaf length increase may be explained by an increase in C and a decrease in NO_3^- and K^+ (P = 0.001).

2.4 Discussion

The hypothesis that *F. hallii* would benefit from straw amendment was supported, although not as a result of nitrogen depletion. Straw treatments positively affected *F. hallii* growth parameters, possibly a result of increased soil water and added nutrients, such as ammonium and potassium. The mulching property of straw may have assisted *F. hallii*, similar to Johnston's (1961) observation that *F. campestris* seedlings produced the largest plants when grown on top a 5 cm litter layer. Straw treatments had little effect on *P. pratensis* growth parameters, contrary to the hypothesis about its response to nitrogen immobilization. The hypothesis about negative effects of admixed soil on *F. hallii* recovery was possibly confirmed. Soil clay content, possibly a result of admixing during restoration, adversely affected *F. hallii* growth, which responded negatively to pH above 7.2 and EC above 0.2. *Poa pratensis* had an opposite response, as it responded well to higher pH and EC.

Higher *F. hallii* growth at the driest site, Drumheller, are similar to those of Stout et al. (1981) who recorded greater *F. campestris* biomass at the drier of two sites. *F. hallii* positively responded to prior-year soil water. Mid-summer is typically when perennial grasses in northern grasslands produce and store carbohydrates for the next spring growth cycle (Holechek et al. 2004); therefore, abundant soil water would contribute to the following year's increased biomass, root biomass and leaf length. In this study, *P. pratensis* soil water responses were consistent with its growth habits, generally preferring mesic habitat and annual growth occurring in early summer (Sinton et al. 1996; Tannas 2001).

Christensen (1985) found barley straw had a higher C:N ratio than wheat straw; however, the Drumheller site, treated with barley, had lower C, N and C:N than wheat-treated sites. Ellerslie had the darkest soil, indicating higher soil organic matter (Dormaar and Willms 1998), possibly promoting growth of both species. Correlation to prior year C for both species could be explained by northern perennial grasses production and storage of carbohydrates in mid-summer, for the following year growth cycle (Holechek et al. 2004).

Ammonium is immobilized quickly during straw decomposition (Smith and Douglas 1971), which could account for decreases found between 2008 and 2009. The ammonium decrease may have involved uptake by *F. hallii*, possibly aided by AMF infection (Molina et al. 1978; Allen 1991). If *F. hallii* NH_4^+ uptake was responsible, there should have been a corresponding increase in *F. hallii* shoot to root ratios. Jarvis (1987) and Li and Redmann (1992) found *E. lanceolatus* and *Lolium perenne* L. (perennial ryegrass) allocated more to shoot growth rather than root growth with additional NH_4^+. Instead, these results are inconsistent as *F. hallii* shoot to root values decreased at Drumheller and Ellerslie, and increased slightly at Byemoor, potentially indicating a greater rate of root growth rather than shoot growth.

All treatments were rototilled, and the resulting exposure of soil to air may have resulted in mineralization and nitrification, causing an increase in soil NO_3^- concentration (Dowdell and Cannell 1975; Doran 1980; Malhi et al. 1990). Andren and Paustian (1987) found that NO_3^- is released as straw decomposes; therefore, the combination of soil nitrification and straw decomposition probably accounted for increasing NO_3^- between 2008 and 2009. *Poa pratensis* responded positively to higher nitrate, likely selecting NO_3^- rather than NH_4^+ (Darrow 1939). Tillman and Wedin (1991b) noted an inverse relationship between *P.*

pratensis root biomass and soil NO_3^- concentrations. Most NO_3^- transported by *P. pratensis* is translocated to shoots (Jiang and Hull 1999; Bushoven and Hull 2001; Jiang et al. 2002). Jiang et al. (2002) also found *P. pratensis* transports NO_3^- at a higher rate than *Festuca arundinacea* Schreb. (tall fescue) and *Lolium perenne* L. (perennial rye). Findings of longer leaf length with lower nitrate imply *P. pratensis* leaf growth assimilated nitrate.

Potassium readily leaches from straw as it decomposes and similar amounts are found in wheat and barley straw (Christensen 1985). *Poa pratensis* treatments had significantly higher K^+ than *F. hallii* treatments, indicating potential K^+ uptake by *F. hallii*, possibly aided by AMF infection (Molina et al. 1978; Allen 1991) Available K^+ was significantly lower at Byemoor than the other two sites, possibly indicating soil admixing (de Jong and Button 1973; Hammermeister et al. 2003). No references were found regarding *F. hallii* use of K, although Templeton and Taylor (1966) found little difference in yield of *F. arundinacea* with varying concentrations of K^+ fertilizer. Recommended potassium for *P. pratensis* are 160 to 240 ppm, while over 300 ppm is considered high (Robinson 1985). Monroe et al. (1969) found *P. pratensis* growth factors (clipping and root biomass, tiller density, leaf blade width and rhizome length) increased with up to 200 ppm of added potassium and decreased at 400 ppm. Byemoor and Ellerslie were within these concentrations, while Drumheller had over 300 ppm and significantly lower *P. pratensis* 2009 biomass and leaf length.

No difference was found in PO_4^- among straw treatments, leading to the conclusion that soil disturbance must have caused the differences among sites, although inconclusive results were found in the literature regarding changes in PO_4^- following disturbance. Hammermeister (2003) found no differences in PO_4^- concentrations on reclaimed well sites, while de Jong and Button (1973) found increases. No correlation was found between PO_4^- and *F. hallii* or *P. pratensis* biomass, root biomass, leaf length, or cover. Smith et al. (1968) found no difference in rough fescue basal area and dry biomass with varying PO_4^- fertilizer. Similarly, PO_4^- had no effect on *P. pratensis* growth in a fertilizer experiment by Christians et al. (1979). Phosphorous, although a key nutrient for amino acid development, is less mobile than NH_4^+ and NO_3^- (Gutschick 1981); therefore, only small amounts may have been up taken by either *F. hallii* or *P. pratensis.*

Soil pH and EC increased during well site construction, probably due to soil horizon admixing (Hammermeister et al. 2003); this was possibly the case at Byemoor. Blevins et al. (1977) suggested soil pH may decrease as N increased, consistent with these findings, where Ellerslie had the highest N concentrations and lowest pH and EC. With higher pH *P. pratensis* had greater leaf length and number, root and leaf weights and number of rhizomes (Darrow 1939); increased vigour, density and color (Skogley and Ledeboer 1968); and greatest abundance (Tilman and Olff 1991). Nevertheless, these studies analyzed *P. pratensis* reaction to acidic soils, and did not assess reaction to pH higher than 7, which was found at Byemoor. No studies were found reporting rough fescue variations in soil pH over 8. In an examination of grazing effects, a decrease in *F. campestris* cover accompanied an increase in soil pH from 5.7 to 6; however, other grazing factors were involved (Johnston et al. 1971). One possible explanation of poor *F. hallii* results in higher pH might be poor growth of various arbuscular mycorrhizae fungi spores at pH above 7 (Hall et al. 1984; Johnson et al. 1984). Molina et al. (1978) found several arbuscular mycorrhizal fungi species colonize *F. hallii*, therefore, if reliant on arbuscular mycorrhizal fungi for nutrient uptake, *F. hallii* could be negatively affected by poor arbuscular mycorrhizal fungi colonization at pH above 7.

2.5 Conclusions and Management Considerations

Field experiments are subjected to numerous uncontrolled effects on soil properties and vegetation growth. Temperature, available water, soil microbial actions, vegetation competition, to name a few, could account for variations in vegetation growth. Nevertheless, consistent correlations to various factors lead to conclusions regarding establishment of *F. hallii* and *P. pratensis*. Results for *P. pratensis* are consistent with other studies involving soil water and NO_3^-. Findings about *P. pratensis* and alkaline pH require further study, as most documented findings are limited to pH 7 or lower; the positive relationship of *P. pratensis* growth to pH above 7 could explain why *P. pratensis* can invade reclaimed disturbed grassland.

Further study on specific ammonium and potassium requirements of *F. hallii*, and *F. hallii* reaction to higher pH and EC could help explain poor recovery on disturbed sites where soil admixing may have altered natural values. While results for *F. hallii* were best for the high-straw amendment, low straw also had better results than no amendment, leading us to conclude addition of straw as a soil amendment is a possible solution to poor establishment of *F. hallii*.

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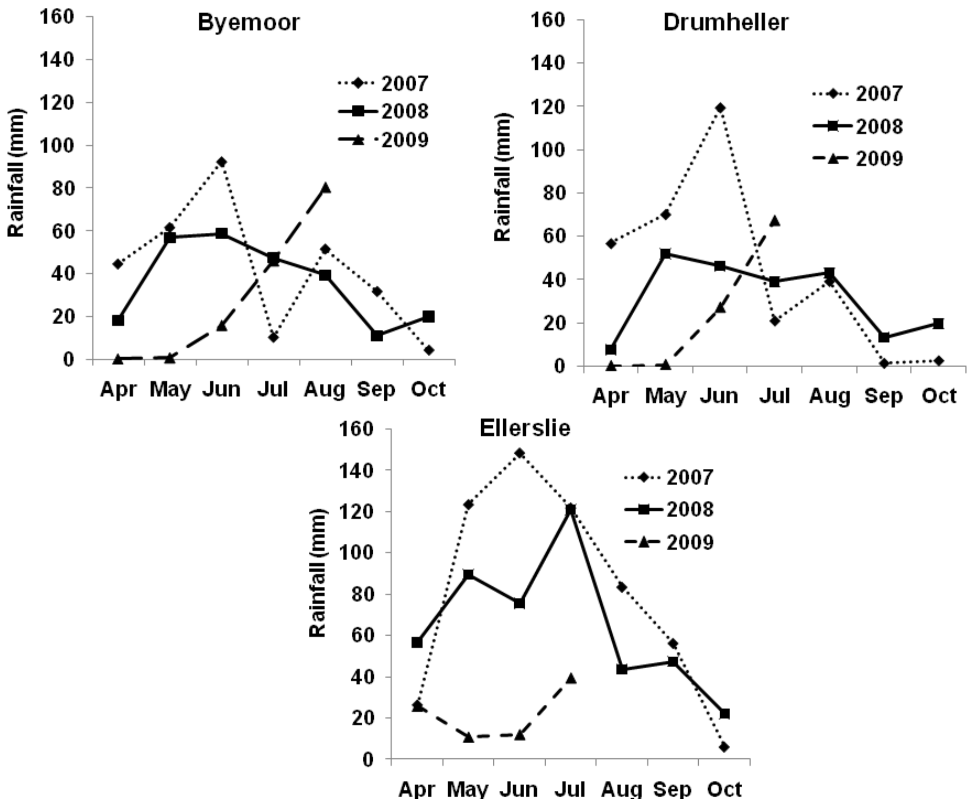

Figure 2.1 April to October rainfall at Byemoor, Drumheller and Ellerslie, in 2007, 2008 and 2009.

	Other	Fescue	Poa	Other	Poa	Fescue	Poa	Other	Poa	Fescue	Other	Fescue
None		X	X				X			X		
High					X		X			X		X
Low			X			X			X			
High					X				X	X		X
Low		X				X	X			X		
None		X	X		X					X		
High						X			X			
None		X				X	X					
Low			X		X	X			X			X
None		X	X		X		X					
High			X			X			X	X		
Low					X					X		

Figure 2.2 Schematic of experimental design showing a sample site with random subplots of seeding and straw treatments. Festuca = *Festuca hallii*, Poa = *Poa pratensis* seeding, High = high straw (1 kg/m^2), Low = low straw (0.5 kg/m^2) and N = no amendment. Strips are 5 – 7 m wide. At Byemoor and Ellerslie 7 subplots and at Drumheller 5 subplots, of each seeding/straw combination, were randomly selected for vegetation and soil sampling.

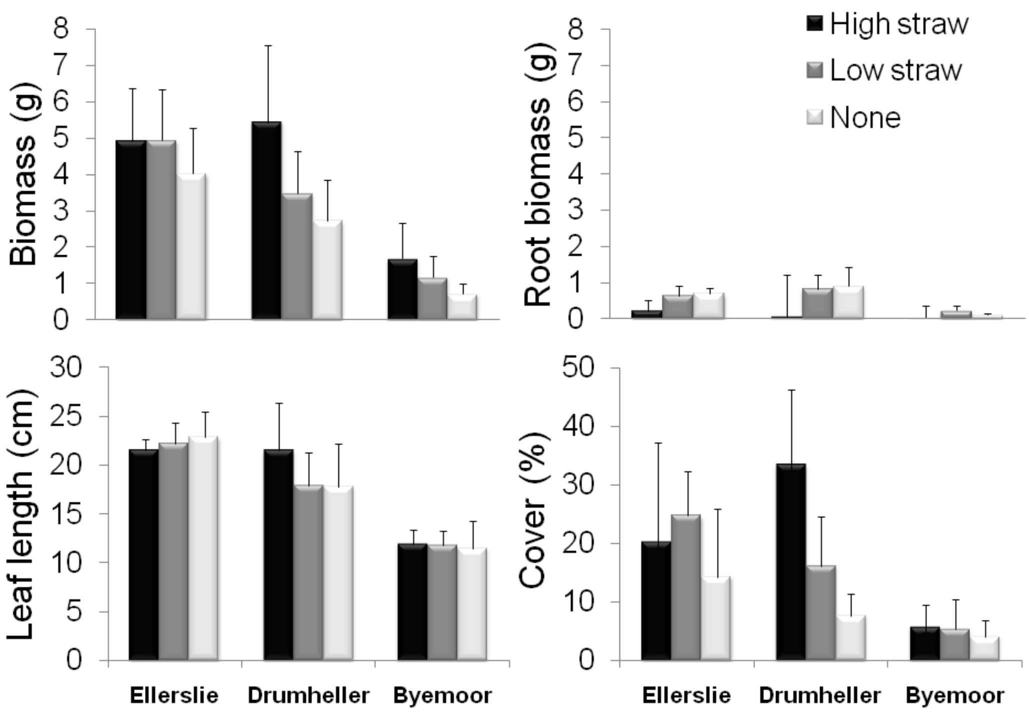

Figure 2.3 *Festuca hallii* comparison of differences among straw amendments: high straw (1 kg/m^2), low straw (0.5 kg/m^2) and no amendment (first *P* value); and field sites (second *P* value): Ellerslie, Drumheller and Byemoor, of 2009 biomass ($P < 0.001$, $P = 0.003$), root biomass ($P < 0.001$, $P = 0.001$), leaf length ($P < 0.001$, $P = 0.485$ and percent cover ($P < 0.001$, $P = 0.010$). Error bars are standard deviation.

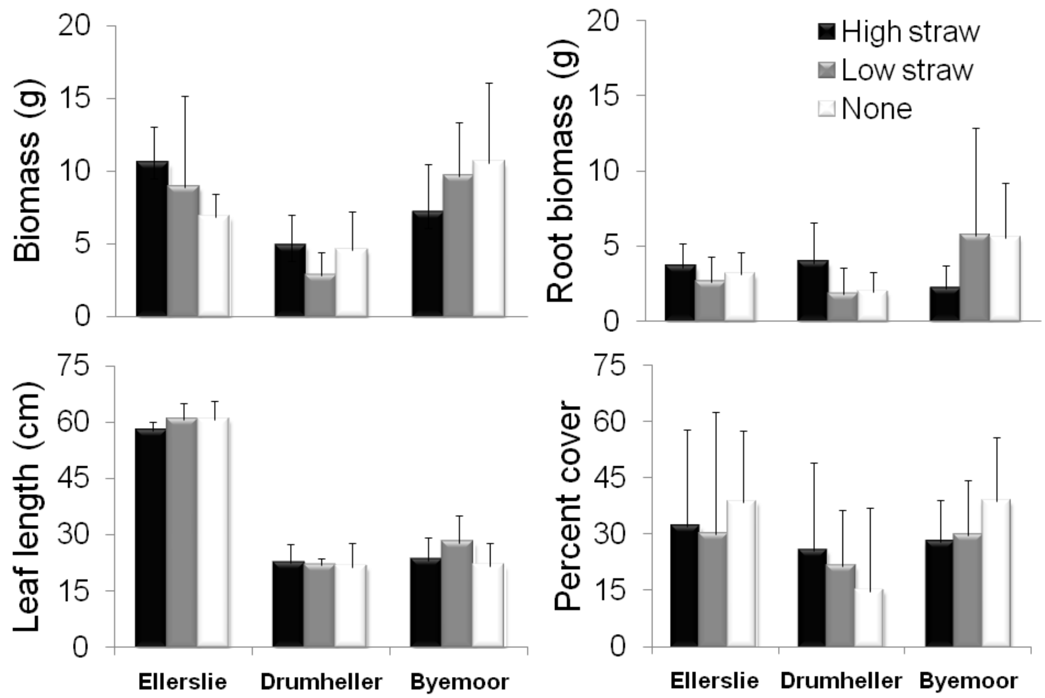

Figure 2.4 *Poa pratensis* comparison of differences among straw amendments: high straw (1 kg/m^2), low straw (0.5 kg/m^2) and no amendment (first P value); and field sites (second P value): Ellerslie, Drumheller and Byemoor, of 2009 biomass (P = 0.534, P < 0.001), root biomass (P = 0.908, P = 0.714), leaf length (P = 0.464, P < 0.001) and percent cover (, P = 0.852, P < 0.001). Error bars are standard deviation.

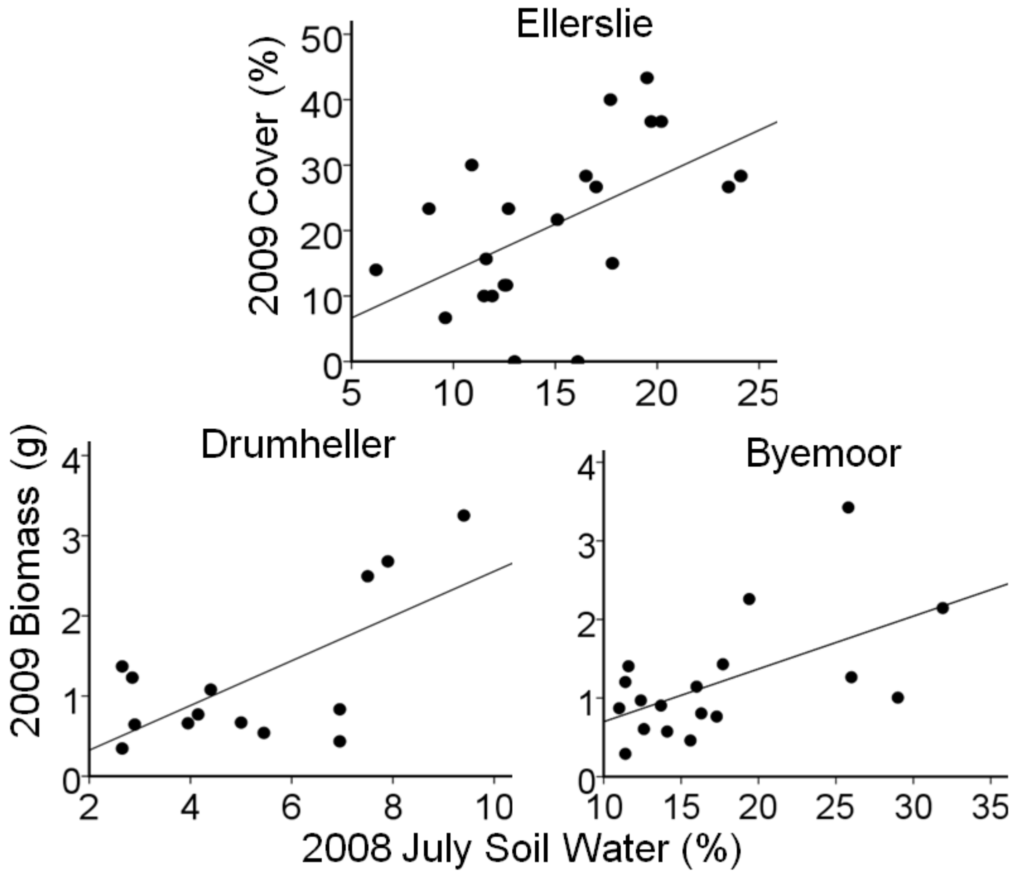

Figure 2.5 *Festuca hallii* relationship to prior-year soil water, Ellerslie ($R^2 = 0.30$, $P < 0.001$), Drumheller ($R^2 = 0.39$, $P < 0.001$) and Byemoor ($R^2 = 0.33$, $P < 0.001$).

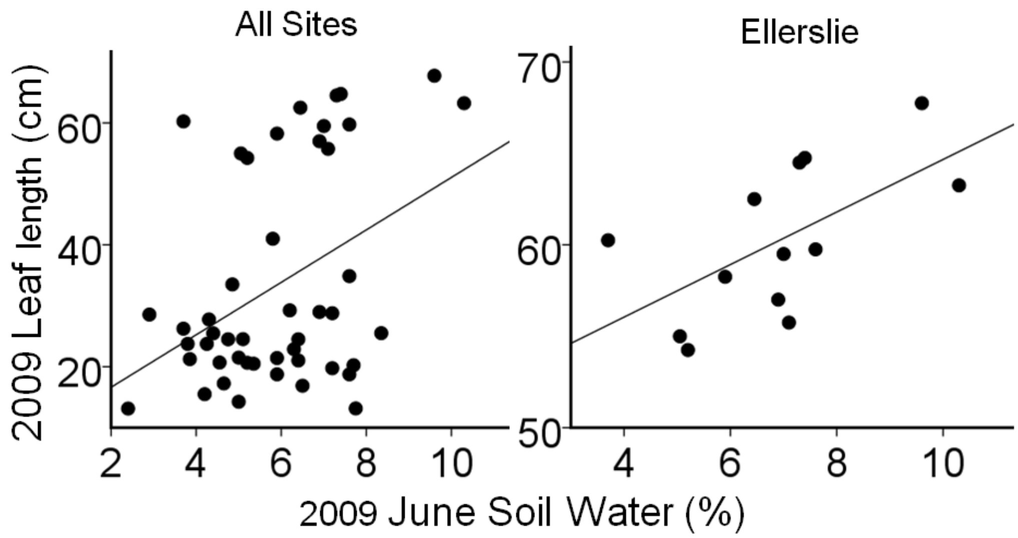

Figure 2.6 Relation of *Poa pratensis* leaf length to current-year soil water in June, at all sites (R^2 = 17, P = 0.005) and at Ellerslie (R^2 = 0.37, P < 0.001).

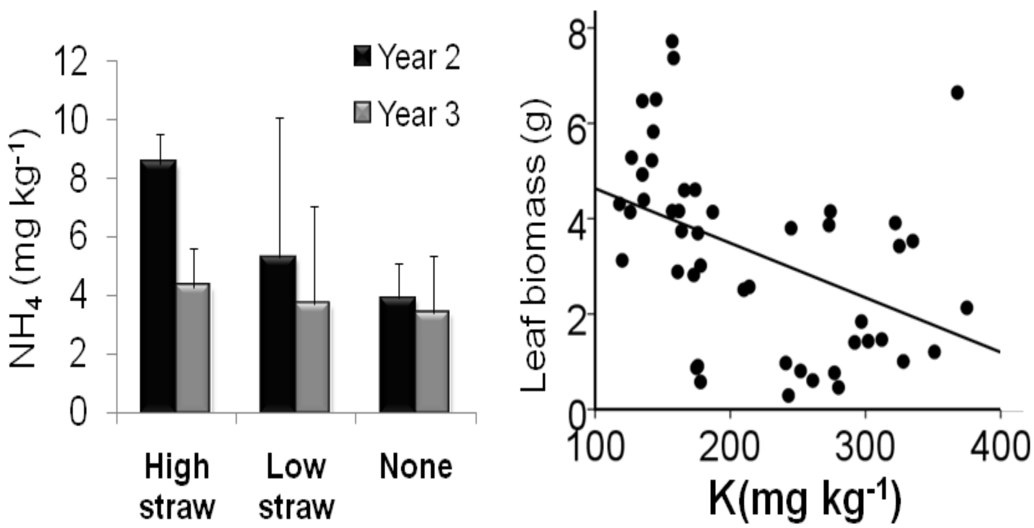

Figure 2.7 Variation in soil NH_4^+ concentrations in *Festuca hallii* subplots showing differences in straw levels (P < 0.001) and years (P < 0.001), and reaction of *Festuca hallii* biomass at all field sites to current-year soil potassium (K; R^2 = 0.19, P = 0.002).

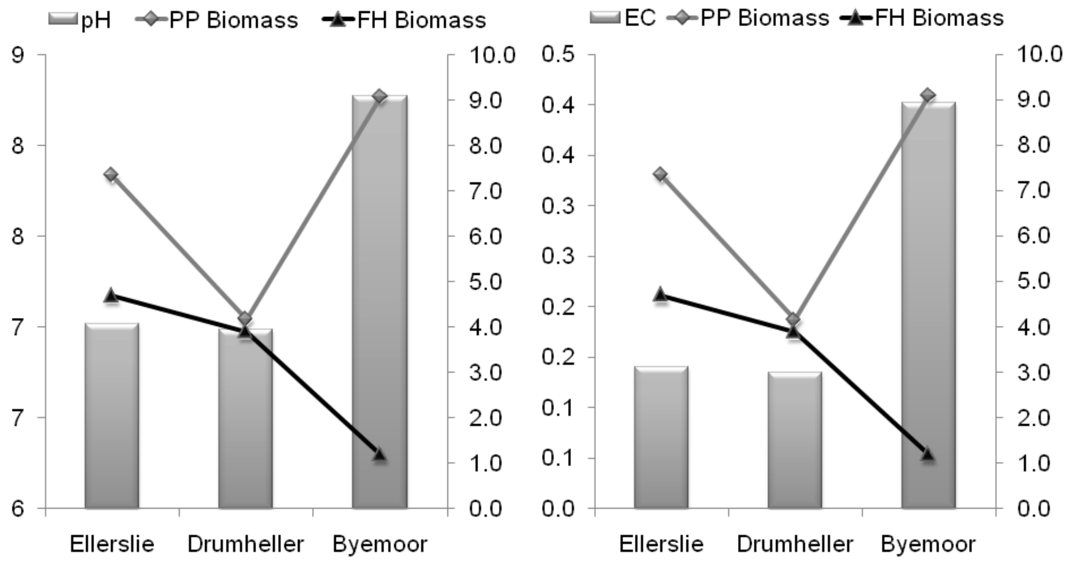

Figure 2.8 Site differences for pH and EC showing and an increase in soil pH (*P* < 0.001) and EC (*P* < 0.001) between Ellerslie, Drumheller and Byemoor with a decrease in 2009 *Festuca hallii* biomass (*P* < 0.001*)* and an increase in *Poa pratensis* biomass (*P* = 0.001).

Table 2.1 Analysis of 2009 *Festuca hallii* by site, straw and site by straw treatments showing *P* values for the sources of variation. Vegetation and soil properties were the response variables.

Response Variables		Site	Straw	Site x Straw
df		2	2	4
		Vegetation properties		
Leaf biomass	2009	**< 0.001**	**0.003**	0.279
Root biomass	2009	**<0.001**	**0.001**	**0.033**
Leaf length	2009	**< 0.001**	0.485	0.224
% Cover	2009	**<0.001**	**0.010**	**0.026**
		Soil properties		
June soil water (%)	2008	**<0.001**	0.520	0.546
	2009	**0.003**	0.127	0.588
July soil water (%)	2008	0.623	0.196	0.586
	2009	**<0.001**	0.889	0.929
Total carbon (mg/kg)	2008	**<0.001**	0.478	0.276
	2009	**<0.001**	0.133	0.736
Total nitrogen (mg/kg)	2008	**<0.001**	0.962	0.888
	2009	**<0.001**	**0.007**	0.960
Available ammonium (mg/kg)	2008	**0.002**	0.198	0.434
	2009	**<0.001**	0.222	0.105
Available nitrate (mg/kg)	2008	0.389	0.593	0.163
	2009	0.852	0.477	0.365
Available phosphate (mg/kg)	2008	**0.004**	0.696	0.480
	2009	0.219	0.924	0.813
Available potassium ($mg/kg^{0.1}$)	2008	**<0.001**	0.079	0.279
	2009	**<0.001**	0.500	0.324
pH	2008	**<0.001**	0.850	0.416
	2009	**<0.001**	0.460	0.453
Electrical conductivity (ds/m^2)	2008	**<0.001**	0.751	0.487
	2009	**<0.001**	0.955	0.975

Table 2.2 Analysis of 2009 *Poa pratensis* by site, straw and site by straw treatments showing *P* values for the sources of variation. Vegetation and soil properties were the response variables.

Response Variables		Site	Straw	Site x Straw
df		2	2	4
		Vegetation properties		
Leaf biomass	2009	**0.001**	0.534	0.555
Root biomass	2009	0.174	0.908	0.222
Leaf length	2009	< **0.001**	0.464	0.567
% Cover	2009	0.071	0.852	0.732
		Soil properties		
June soil water (%)	2008	**<0.001**	**0.001**	0.077
	2009	0.210	0.619	0.858
July soil water (%)	2008	**0.006**	**0.015**	**0.044**
	2009	**<0.001**	0.770	0.730
Total carbon (mg/kg)	2008	**<0.001**	0.350	0.400
	2009	**<0.001**	0.787	0.061
Total nitrogen (mg/kg)	2008	**<0.001**	0.617	0.726
	2009	**<0.001**	**<0.001**	**<0.001**
Available ammonium (mg/kg)	2008	0.997	0.600	0.734
	2009	0.041	0.519	0.535
Available nitrate (mg/kg)	2008	0.526	0.388	0.155
	2009	0.235	0.598	0.053
Available phosphate (mg/kg)	2008	0.486	0.155	0.950
	2009	0.725	0.633	0.863
Available potassium (mg/kg$^{0.1}$)	2008	**<0.001**	0.986	**0.001**
	2009	**<0.001**	0.271	0.087
pH	2008	**<0.001**	0.900	0.913
	2009	**<0.001**	0.260	0.183
Electrical conductivity (ds/m^2)	2008	**<0.001**	0.787	0.534
	2009	**<0.001**	0.558	0.535

Table 2.3 Mean (±1 SD) soil properties of *Festuca hallii* treatments in 2008 and 2009, analyzed by ANOVA and Tukey post-hoc tests. Letters indicate significant differences at $P \leq 0.05$.

		Ellerslie	Byemoor	Drum-heller	*P*
June soil water (%)	2008	24.0a (6)	34.0a (5)	5.3b (1)	**<0.001**
	2009	8.7a (2)	6.5 (2)	5.3b (1)	**<0.001**
July soil water (%)	2008	15.0 (5)	17.0 (7)	16.0 (4)	0.326
	2009	31.0a (2)	27.0b (5)	13.0c (2)	**<0.001**
August soil water (%)	2008	16.0a (2)	11.0b (5)	5.2b (2)	**<0.001**
C:N Ratio	2008	9.0a (1)	14.0b (5)	11.0a (1)	**<0.001**
	2009	11.0a (1)	10.0ab (0)	10.0b (1)	**0.017**
Total carbon	2008	4.2a (0)	2.6b (1)	2.6b (1)	**<0.001**
(mg/kg)	2009	5.4a (0)	2.6b (1)	2.8b (1)	**<0.001**
Total nitrogen	2008	0.47a (0)	0.2b (0)	0.2c (0)	**<0.001**
(mg/kg)	2009	0.51a (0)	0.3b (0)	0.3b (0)	**<0.001**
Available ammonium	2008	5.6ab (0)	2.8a (1)	6.1b (5)	**0.003**
(mg/kg)	2009	5.0a (1)	3.3b (2)	1.9c (0)	**<0.001**
Available nitrate	2008	8.1 (3)	6.9 (15)	4.8 (6)	0.585
(mg/kg)	2009	9.5 (5)	9.0 (15)	7.5 (4)	0.846
Available phosphate	2008	8.4a (1)	8.4b (3)	12.4c (6)	**0.004**
(mg/kg)	2009	12.0 (4)	12.0 (3)	15 (4)	**0.077**
Available potassium	2008	13.3 a(1)	27.5b (5)	28.9b (9)	**<0.001**
($mg/kg^{0.1}$)	2009	15.2 a (2)	26.3b (6)	29.3b (6)	**<0.001**
pH	2008	6.7a (0)	8.4b (1)	7.2c (1)	**<0.001**
	2009	6.7a (1)	8.4b (0)	7.5c (0)	**<0.001**
Electrical conductivity	2008	0.1a (0)	0.5b (0)	0.2a (0)	**<0.001**
(ds/m^2)	2009	0.1a (0)	0.4b (0)	0.2a (0)	**<0.001**

Table 2.4 Mean (±1 SD) soil properties of *Poa pratensis* treatments in 2008 and 2009, analyzed by ANOVA and Tukey post-hoc tests. Letters indicate significant differences at $P \leq 0.05$.

		Ellerslie	Byemoor	Drumheller	*P*
June soil water (%)	2008	21.0ab (4)	34.0b (5)	17.0a (5)	< **0.001**
	2009	6.9a (2)	5.8ab (1)	5.2b (2)	**0.025**
July soil water (%)	2008	16.0 (5)	18.0 (5)	14.0 (4)	0.129
	2009	34.0a (1)	26.0b (3)	13.0c (2)	<**0.001**
August soil water (%)	2008	18.0a (5)	9.7b (3)	4.8c (2)	<**0.001**
Total available soil	2008	46.0a (18)	60.0b (7)	36.0a (9)	<**0.001**
water[1]	2009	41.0a (3)	32.0b (3)	17.0c (5)	<**0.001**
C:N Ratio	2008	11.0 (3)	12.0 (3)	10.0 (1)	0.256
	2009	11.0a (0)	10ab (1)	9.8b (1)	**0.016**
Total carbon (mg/kg)	2008	4.9a (1)	2.7b (1)	3.3b (1)	< **0.001**
	2009	5.3a (0)	2.9b (1)	3.2b (0)	< **0.001**
Total nitrogen (mg/kg)	2008	0.4a (0)	0.2b (0)	0.3c (0)	< **0.001**
	2009	0.5a (0)	0.3b (0)	0.3b (0)	< **0.001**
Available ammonium	2008	4.7 (1)	4.2 (4)	4.5 (2)	0.857
(mg/kg)	2009	5.2a (1)	3.7b (3)	2.3b (1)	**0.001**
Available nitrate	2008	9.8a (5)	6.0ab (6)	4.7b (5)	**0.038**
(mg/kg)	2009	3.0a (2)	7.5b (7)	8.3b (3)	**0.019**
Available phosphate	2008	9.0 (1)	9.3 (3)	10.0 (3)	0.386
(mg/kg)	2009	12.0 (3)	12.0 (3)	12.0a (4)	0.943
Available potassium	2008	14.0a (2)	25.5b (5)	32.9c (7)	<**0.001**
($mg/kg^{0.1}$)	2009	15.8a (3)	30.3b (7)	35.9c (5)	<**0.001**
pH	2008	6.7a (1)	8.3b (1)	7.0a (0)	< **0.001**
	2009	7.0a (0)	8.3b (1)	7.0a (0)	< **0.001**
Electrical conductivity	2008	0.1a (0)	0.5b (0)	0.2a (0)	<**0.001**
(ds/m^2)	2009	0.1a (0)	0.4b (0)	0.1a (0)	<**0.001**

Table 2.5 Pearson product correlations of *Festuca hallii* 2009 biomass, root biomass, leaf length and cover with soil properties in 2008 and 2009.

		Biomass		Root Biomass		Leaf Length		% Cover	
		R	*P*	R	*P*	R	*P*	R	*P*
Soil water	June 2008	-0.576	**0.001**	-0.494	**0.006**	-0.664	**<0.001**	-0.226	0.229
	July 2008	-0.188	0.170	-0.146	0.291	-0.211	0.122	0.021	0.881
	Aug 2008	-0.694	**<0.001**	-0.215	0.246	-0.321	0.078	-0.248	0.179
	June 2009	0.321	**0.018**	-0.058	0.682	0.324	**0.017**	0.179	0.195
	July 2009	0.094	0.514	-0.297	**0.036**	0.028	0.847	-0.042	0.768
Total carbon	2008	0.504	**<0.001**	0.157	0.287	0.455	**0.001**	0.319	**0.024**
	2009	0.599	**<0.001**	0.210	0.162	0.603	**<0.001**	0.348	**0.016**
Total nitrogen	2008	0.561	**<0.001**	0.143	0.332	0.600	**<0.001**	0.366	**0.009**
	2009	0.616	**<0.001**	0.227	0.129	0.624	**<0.001**	0.332	**0.023**
Available ammonium	2008	0.377	**0.011**	0.441	**0.003**	0.246	0.103	0.522	**<0.001**
	2009	0.304	**0.038**	-0.030	0.844	0.283	0.054	0.289	**0.049**
Available nitrate	2008	0.085	0.560	0.081	0.584	-0.044	0.763	0.019	0.894
	2009	0.001	0.995	-0.030	0.842	-0.141	0.345	-0.250	0.090
Available potassium	2008	-0.271	0.060	0.158	0.284	-0.373	**0.008**	-0.083	0.566
	2009	-0.441	**0.002**	0.017	0.912	-0.493	**<0.001**	-0.419	**0.003**
Available phosphate	2008	0.001	0.992	0.203	0.166	0.150	0.305	0.188	0.191
	2009	-0.096	0.522	0.212	0.156	0.006	0.968	0.179	0.228
pH	2008	-0.719	**<0.001**	-0.476	**0.001**	-0.612	**<0.001**	-0.339	**0.016**
	2009	-0.729	**<0.001**	-0.381	**0.009**	-0.696	**<0.001**	-0.269	0.068
Electrical conductivity	2008	-0.636	**<0.001**	-0.441	**0.002**	-0.637	**<0.001**	-0.393	**0.005**
	2009	-0.662	**<0.001**	-0.514	**<0.001**	-0.779	**<0.001**	-0.487	**0.001**

Table 2.6 Pearson product correlations of *Poa pratensis* 2009 biomass, root biomass, leaf length and cover with soil properties in 2009.

		Biomass		Root Biomass		Leaf Length		% Cover	
		R	*P*	R	*P*	R	*P*	R	*P*
Soil water									
	June 2009	-0.074	0.623	-0.072	0.630	0.407	**0.005**	0.148	0.320
	July 2009	0.191	0.215	-0.050	0.746	0.711	**<0.001**	0.466	**0.001**
Total carbon (C)	2009	-0.107	0.499	-0.056	0.724	0.883	**<0.001**	0.444	**0.003**
Total nitrogen (N)	2009	-0.121	0.445	0.011	0.947	0.740	**<0.001**	0.410	**0.006**
Available ammonium (NH_4^+)	2009	-0.570	0.721	-0.200	0.199	0.507	**0.001**	0.228	0.142
Available nitrate (NO_3^-)	2009	0.321	0.038	0.370	**0.015**	-0.492	**0.001**	-0.247	0.110
Available potassium (K)	2009	-0.088	0.581	-0.139	0.372	-0.705	**<0.001**	-0.442	**0.003**
Available phosphate (PO_4^-)	2009	0.080	0.988	-0.061	0.696	0.013	0.932	-0.278	0.072
pH	2009	0.357	**0.020**	-0.013	0.935	-0.391	**0.010**	-0.930	0.554
Electrical conductivity	2009	0.459	**0.002**	0.130	0.406	-0.402	**0.008**	0.017	0.914

Table 2.7 Mean (±SD) soil properties of undisturbed grassland at Byemoor and Drumheller sites

	Byemoor	Drumheller
Total carbon (C) (mg/kg)	3.7 (1)	3.3 (1)
Total nitrogen (N) (mg/kg)	0.3 (0)	0.3 (0)
Available ammonium (NH_4^+) (mg/kg)	6.4 (4)	5.5 (2)
Available nitrate (NO_3^-) (mg/kg)	2.9 (1)	1.9 (1)
Available phosphate (PO_4^-) (mg/kg)	4.0 (2)	5.0 (2)
Available potassium (K^+) (mg/kg)	253 (83)	220 (38)
pH	6.4 (1)	6.6 (1)
Electrical conductivity (EC) (ds/m^2)	0.2 (0)	0.3 (0)

CHAPTER 3. RESULTS WITH *FESTUCA HALLII* SEEDING FOLLOWING DISTURBANCE, IN CENTRAL ALBERTA

3.1 Introduction

The Central Parkland natural region, the upper edge of the Northern Great Plains, has been subjected to disturbance since the late 19th century by agricultural cultivation, urban and rural residential expansion, and petroleum industry exploration and development. The Central Parkland was once comprised mainly of rough fescue grassland, dominated by *Festuca hallii* (Vasey) Piper (plains rough fescue) (Looman 1969). This long-lived bunch grass grows slowly, requiring three to five years to mature. It is an erratic seed-setter, seldom producing seed, with between 2 and 10 years between seeding events (Johnston and MacDonald 1967; Toynbee 1987; Romo 1996).

The ability to restore fescue prairie, including *F. hallii*, has become an important consideration for oil and gas companies in Alberta, which are required by law to return disturbed land in native grassland to within 15% of prior disturbance species cover (Alberta Environment 2010). Few attempts to restore rough fescue plant communities in the Central Parkland have been successful. Studies of gas well sites and pipelines reclaimed in the parkland in the past 5 to 27 years found fair to poor establishment of rough fescue and other native species from reclamation seed mixes and sod salvage (Petherbridge 2000; Elsinger 2009). A restoration experiment in the grasslands of central Saskatchewan concluded that conserving remaining rough fescue grassland would have greater benefit than attempting to restore it (Clark 1986).

Disturbance reclamation in native grassland includes seeding with a mix of native species, in attempts to return the site to pre-disturbance conditions. Fast growing species, such as wheat grasses, are generally included in native seed mixes to prevent wind and water erosion on newly disturbed sites (Hammermeister et al. 2003), with the expectation that slower growing bunch grasses will eventually establish. Despite inclusion of *F. hallii* in reclamation seed mixes, little was found in vegetation assessments of pipelines and well sites 7 to 30 years following reclamation (Elsinger 2009; Desserud et al. 2010).

McArthur and Wilson (1967) recognized that plant species utilize resources differently and derived the theory of r/K-selection, which differentiates species based on use of resources and habitat carrying capacity. r-selected species quickly reproduce using any available resources, while K-selected species are slower growing and rely on carrying capacity of the environment (McArthur and Wilson 1967). Ruderals, opportunistic annual forbs which readily colonize disturbed sites, occur at one end of the resource usage r/K continuum. Some species, such as *Poa pratensis* L. (Kentucky blue grass) may occupy both ends of the r/K continuum, easily establishing disturbed sites, then becoming and remaining dominant (Taylor et al. 1990). *Festuca hallii*, slow growing and eventually dominant, could be defined as a strong K-selector species. Competition may result from the ability of one or more species to utilize available resources at the expense of another (Grime 1979) or the ability of a species to maximize the ratio of resource supply to demand when resources are limited (Tilman 1988; Taylor et al. 1990). Tilman and Wedin (1991a; 1991b) found poor nitrogen competitors, such as rhizomatous *Elymus repens* (L.) Gould. (quack grass), were the first to colonize fields after abandonment, while better nitrogen competitors such as bunch grasses, required 11 to 17 years to invade fields, and up to 40 years more to dominate. They noted self recruitment of native bunch grasses was very slow. It is likely that competition from species, such as wheat grasses, impedes successful *F. hallii* establishment when seeded (Johnston 1961).

During reclamation seeding, competition for slower growing plant species such as *Festuca hallii* may be reduced with lower seeding rates of aggressive species or seeding with monocultures; however, monoculture seeding is unlikely to result in a community resembling pre-disturbance conditions. The research objective was to assess the competitive ability of *F. hallii* by comparing a monoculture seeding to seeding with a native seed mix including *F. hallii*, and to evaluate plant community development of both seeding mixes over a short period of time. The hypothesis is that seeding only a slow growing species such as *F.*

hallii may result in bare ground, increasing potential for erosion, and subsequent invasion by non-native species.

3.2 Study Sites

Three field sites were established in 2007 in central Alberta, Canada: two in the Central Parkland natural region at Ellerslie (53° 25′ N, 113° 29′ W) and Byemoor (51° 59′ N, 112° 19′ W) and one in the Northern Fescue sub-region of the Grassland natural region at Drumheller (51° 26′ N, 112° 21′ W). Elevation at each site is approximately 660, 900 and 900 m above sea level, respectively. Soils at Ellerslie are Orthic Black Chernozems while at Byemoor and Drumheller they are Dark Brown Chernozems. Native grassland vegetation in both natural regions is dominated by *F. hallii* associated with *Hesperostipa curtiseta* (A.S. Hitchc.) Barkworth (western porcupine grass) in the Central Parkland, and *Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths (blue grama grass) in the Northern Fescue sub-region.

During the research period between June 2007 and July 2009, temperature maximums were -40 to 36 °C, with growing season (April to October) temperatures averaging 11 and 12 °C. Average annual rainfall in this period was 510 mm at Ellerslie, 302 mm at Byemoor, and 270 mm at Drumheller. Seeding occurred in July 2007, when June was abnormally wet at all sites and July was abnormally dry at Byemoor and Drumheller, followed by normal precipitation in August. Rainfall was average in 2008 while 2009 was abnormally dry in April, May and June at all sites.

The Ellerslie site is 1 ha in size and located in a previously cultivated area. Prior to the experiment it was fallowed for 3 years, and then in May 2007, sprayed with glyphosate (Roundup Weathermax) at 0.4 l/ha, with a Flexicoil sprayer. Topsoil is approximately 23 cm deep. The Drumheller site is a 0.5 ha abandoned natural gas well site, in uncultivated land. In June 2007 it was cleared of topsoil, flow-line piping was removed, the well capped and topsoil replaced to a depth of approximately 11 cm. The Byemoor site is 1 ha in size, on uncultivated land, and was leveled and cleared of topsoil in preparation for gas well drilling, one year prior to this experiment. The well was not drilled, and in April 2007, the Byemoor site was recontoured and topsoil replaced to a depth of approximately 30 cm.

3.3 Materials and Methods

3.3.1 Field Methods

Festuca hallii seed was collected in July 2006, in rough fescue grassland of the Rumsey Natural Area (51° 50′ N, 112° 33′ W), in central Alberta with a combine (Wintersteiger Plot Master Elite® set at a height of approximately 30 cm. Seed was air dried at 25 °C, the chaff removed, and *F. hallii* seed separated from other grasses. One hundred randomly selected seeds were distributed equally among ten closed Petri dishes on 1 mm thick germination paper, wetted with distilled water, and incubated in the dark at 20 °C (Romo et al. 1991). After six weeks 95% germination was achieved and the seed was deemed viable.

Prior to setting up field experiments, germination and establishment of *F. hallii* was tested in the greenhouse, alone and with *P. pratensis* to determine inter- and intra-species competition. Soil was collected by hand from stored topsoil piles on a natural gas well site (0 to 50 cm depth) in the vicinity of the planned field sites, and added to simulate topsoil replacement. Seeds were randomly selected for seeding into 18 cm pots. Treatments were *F. hallii* alone (5 seeds per pot) and *F. hallii* and *P. pratensis* together (3 seeds each per pot), replicated 48 times. Pots were randomized in the greenhouse, kept at 25 °C with 16 hours of sunlight to mimic summer growing conditions. Pots were watered with tap water when the surface appeared dry, about every two or three days and randomly rotated every three weeks. Leaf length was recorded every month for six months. At six months, plants were harvested at soil level and dry biomass determined following 48 hours of oven drying at 60 °C.

The three field sites were treated with straw amendments in July 2007. One-year-old certified weed-free straw was applied at a rate of 0.5 kg/m^2 (; barley straw was applied with a Hesston 25 bale processor at Drumheller, and wheat straw with a Highline BP7800 at Ellerslie and Byemoor, each running at approximately 2 km/hr. Uneven straw applications were manually raked prior to rototilling. Straw was incorporated 8 to 10 cm into the soil with a Howard 2 m rototiller pulled by a 50 hp tractor running at 1800 rpm and 2.4 km/h.

Field sites were seeded in July 2007, at a depth of 1.2 to 1.9 cm with a Brillion® seeder and packer with double disk openers. *Festuca hallii* and a native mix were seeded in randomized strips (6 to 7 m by 70 to 90 m) with four replications. The native mix at all sites included *F. hallii*, *Koeleria macrantha* (Ledeb.) J.A. Schultes (June grass), *Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould ssp. *Lanceolatus* (slender wheat grass) and *Pascopyrum smithii* (Rydb.) A. Löve (western wheat grass). For Ellerslie and Byemoor, *Nassella viridula* (Trin.) Barkworth (green needle-grass), native to the Central Parkland region, was added. At Drumheller, two species found in the Northern Fescue sub-region were included: *Hesperostipa comata* (Trin. & Rupr.) Barkworth ssp. *comata* (needle-and-thread grass) and *Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths (blue grama) (Table 3.1).

In September 2007, all sites were mowed with a tractor-mounted Massey Ferguson model 36 hay swather, at 15 cm height, to remove opportunistic wheat and barley growing from the straw application. All sites had non-native species in the first year. At Ellerslie, non-native species, such as *Cirsium arvense* (L.) Scop. (Canada thistle), *Monolepus nuttalliana* (Schultes) Greene (goosefoot), and *Convolvulus arvensis* L. (bindweed), occurred at over 20% cover, greater than the other sites (less than 5% cover), and thus were considered great enough to alter establishment of seeded species (Stohlgren et al. 1999). As a result, the Ellerslie site was mowed again, in June 2008, and treated with a broad-leafed herbicide, Embutox 625 (Group 4), at rate of 0.4 l/ha with a Flexicoil sprayer.

Each seeded strip was divided into twelve 6 x 7 m subplots (Figure 3.1). The seeded vegetation was sampled in late July and early August, 2008 in randomly selected subplots; the same subplots were re-sampled in 2009. Twenty one subplots of each seeding treatment were selected at Byemoor and 15 at Drumheller. A 50 cm wide buffer around each plot was not sampled to avoid edge effects. In each plot, three 20 x 50 cm quadrats were randomly located to visually estimate cover of all species (Daubenmire 1959). In adjacent undisturbed grassland, approximately 15 m from the field sites, ten 20 by 50 cm subplots, 10 m apart along a 50 m transect were sampled to serve as undisturbed controls.

In 2009, 15 subplots at each site, different from the vegetation assessment subplots, were randomly selected to assess the potential effect of competition from neighbouring plants on *F. hallii*. Within each subplot between one and three of the largest *F. hallii* plants were selected, based on leaf length and basal circumference. The distance of all neighbouring plants within 30 cm was measured as maximum and minimum distances. Plants were identified as grass, forb or *F. hallii*, and classified as large, medium or small in size for their expected growth properties.

3.3.2 Statistical Analyses

Data were normally distributed and differences among *F. hallii* growth parameters and vegetation cover were analyzed by ANOVA and General Linear Model (GLM). Nearest neighbor type was converted to binary coding (0 or 1) indicating presence or absence, for each nominal class. Stepwise multiple regression was used to test the effect of each class and variable on *F. hallii* growth characteristics. Statistical analyses were performed with IBM® SPSS® Statistics (version 18, SPSS, Chicago IL). Indicator species analysis specified the dominant species of the plant communities of seeded treatments and controls. Indicator values (IV) ranges from 0 to 100, where 100 indicated a species is exclusively found in a particular group (Dufrene and Legendre 1997). Nonparametric multiple response permutation procedure (MRPP), operating on Sorenson (Bray-Curtis) distance measures, was used to evaluate significant differences between seeded treatments and controls (Zimmerman et al. 1985) using PC-ORD (version 5.31, MjM Software, Gleneden Beach OR). MRPP generates a chance corrected, within group, agreement value (A), which evaluates the difference between species composition of grouped treatments. The lower the A value, the more similar are the groups (McCune and Grace 2002).

3.4 Results

Festuca hallii growth characteristics were significantly different when grown in the greenhouse alone and with *P. pratensis* (Figure 3.2). At two months, little difference occurred; however mean, minimum and maximum leaf lengths were significantly shorter when grown with *P. pratensis* at the fourth ($P < 0.001$) and sixth months ($P < 0.001$). *Poa pratensis* had greater leaf length when grown with *F. hallii* (Figure 3.2).

Festuca hallii field basal circumference was associated with neighbouring plants ($P = 0.043$, $R^2 = 0.05$) (Figure 3.3). Multiple regression tested relationships between *F. hallii* growth characteristics and its nearest neighbors (number of neighbors, vegetation type and size, maximum and minimum distance). *F. hallii* basal circumference increased if nearest neighbors were forbs or *F, hallii* plants and minimum distance increased ($P = 0.004$, $r^2 = 0.60$); *F. hallii* leaf length increased if nearest neighbors were forbs or *F. hallii* plants ($P = 0.030$, $r^2 = 0.53$).

Combining results from the three field sites showed differences in cover of seeded species (Table 3.2). Rough fescue treatments were dominated by *F. hallii* ($P < 0.001$), while native mix treatments had greater cover of *Bouteloua gracilis* ($P < 0.001$), *Nassella viridula* ($P < 0.001$), *Elymus trachycaulus* ($P < 0.001$), *Hesperostipa comata* ($P = 0.015$), and *P. smithii* ($P < 0.001$). *Koeleria macrantha* cover was similar in both *F. hallii* and native mix treatments ($P = 0.326$). Both seeded treatments had similar incursions of native species that were not seeded (Table 3.2). Total vegetation cover was similar between treatments ($P = 0.303$). *Festuca hallii* treatments had greater bare ground ($P = 0.012$), species diversity ($P = 0.004$), and species richness ($P = 0.010$) than native mix treatments. Native mix treatments had greater litter ($P = 0.027$) and fewer non native species ($P = 0.019$) than *F, hallii* treatments (Figure 3.4).

Festuca hallii treatments were more similar to controls ($A = 0.07$, $P < 0.001$) than native-mix treatments ($A = 0.17$, $P < 0.001$). *Festuca hallii* treatments were dominated by *F. hallii* (IV = 34) and *P. pratensis* (IV = 42), with similar cover to controls for *B. gracilis*, *H. hookeri* and *P. smithii* (Figure 3.5 and Table

3.2). Native-mix treatments were dominated by *E. trachycaulus* (IV = 72) and *N. viridula* (IV = 62). Little *F. hallii* and few forbs were found in native-mix treatments (Figure 3.5 ansd Table 3.2). Controls were dominated by *Carex* spp. (IV = 99), *H. curtiseta* (IV = 92), and *F. hallii* (IV = 58). Controls had more forbs ($P = 0.001$), shrubs ($P < 0.001$), total cover ($P = 0.020$) and moss and/or lichens ($P < 0.001$) than other treatments (Table 3.2).

3.5 Discussion

Competition from neighbouring plants clearly affected *F. hallii*. Competition is the tendency of neighbouring plants to utilize the same quantum of light, ion of a mineral nutrient, molecule of water or volume of space (Grime 1977). Competition between *F. hallii* and *P. pratensis* was evident in greenhouse pots, where both species used the same resources. Tilman's resource utilization theory could apply in a greenhouse setting, as resources are limited by the size of the pots. *P. pratensis*, in this case, made better use of limited resources, than *F. hallii*.

The C-S-R triangle theory is a refinement of the r/K-selection theory, stating that the intensity of competition (C) for resources (R) declines as intensity of stress (S) and/or disturbance increases (Grime 1985). Newly reclaimed well sites, with replaced topsoil, may be considered both high disturbance and low stress. They are high disturbance because nothing remains of the original plant community, but low stress, with no existing competitors and plentiful soil nitrogen. R-selector species should become quickly established in such an environment. Without competition, K-selector species, such as *F. hallii*, may also thrive. *F. hallii* responded positively to distant neighbors, implying free space may allow it to mature slowly, with no competition for resources.

These results are similar to Johnston (1961), who observed the least vigorous *Festuca campestris* plants were those subjected to the most competition, such as those sown on established grassland. Holling (1992) and Gunderson (2000) noted ecosystem succession following disturbance is usually characterized by initial rapid colonization of fast growing species, while species that dominate in a later phase tend to have slower growth rates and survive in an arena of exclusive competition. Tilman (1982; 1990) concluded competition for nutrients is the major factor determining species composition of natural plant communities. He reported increased biomass of a bunch grass when neighbouring biomass was removed, attributing the increase to improved access to nutrients.

Festuca hallii thrived if its nearest neighbor was other *F. hallii* plants or forbs. An indication that *F. hallii* tolerates its own species in the field was confirmed in the greenhouse, where pots of only *F. hallii* grew better than those with *P. pratensis*. The slow growth trait of *F. hallii* might indicate low resource usage; thus at early stages, intra-specific competition may not occur. Similarly Moora and Zobel (1996) found that intra-specific competition did not occur when young neighbor plants (*Fragaria vesca* L.) were grown close to young target plants (*Prunella vulgaris* L.)

Poa pratensis competes successfully with native grasses in Canada and the United States (Robocker and Miller 1955; Donkor et al. 2002). These results are similar to Reader et al. (1994) who found little change in *Poa pratensis* growth parameters when examining its reaction to neighbouring species, possibly a result of its highly competitive nature. Same resource usage would imply intra-specific competition should occur when a plant grows near its own species; however, at early growth stages, this may not apply.

Festuca hallii tolerance of forbs might imply use of different resources. *F. hallii* is deep rooted (Best et al. 1971) while most forbs, especially annuals are shallow rooted; therefore, each species type may use varying nutrients at different soil depths. Grime (1977) noted disparities in the performance of neighbouring plants may arise not from competition, but from differences in their capacity to exploit features of the physical or biotic environment.

The degree of plant community development in the *F. hallii* treatment in the field was unexpected. Not only did non native species not take over, but erosion was not evident and several volunteer perennial grasses became established. *Festuca hallii* treatments were starting to resemble undisturbed controls, although some species differed, and bare ground was greater. Species richness of the *F. hallii* treatment in the third year was consistent with observations of Tilman (1997), who noted an inverse relationship between original species richness and the amount of new recruitment. The native mix treatment initially had up to seven species, while the *F. hallii* treatment was initially a monoculture. Bare ground in the *F. hallii* treatment has potential to recruit species, such as perennial grasses, (*E. dasystachyum* and *H. curtiseta)*, which established easily by the third year. These results differed from Tilman (1997) who found perennial grasses were the poorest invaders in a grassland seeding experiment.

Incursion of forbs and several native grasses into both treatments could have resulted from seed banks or seed rain from adjacent seeded treatments and native grassland. For *Festuca hallii* treatments, seed rain from adjacent seeded treatments probably contributed to the *P. smithii*, *E. trachycaulum,* and *N. viridula* cover. Seed banks or seed rain likely contributed to *H. hookeri* and *H. curtiseta* cover. Annual forbs, such as ruderal species, readily colonize disturbed areas, taking advantage of bare ground and low competition for resources (Grime 1977). Greater bare ground in the *F. hallii* treatment may have aided *F. hallii* growth, providing gaps in the vegetation canopy, thus reducing below ground competition (Cahill and Casper 2002). Higher litter in the native mix treatment may have suppressed non native species, similar to Evans and Young (1970), who found litter suppressed the growth of annual weeds.

Wheat grasses success in the native mix treatment, despite low seed mix percentages, confirms observations by Hammermeister (2001) that *P. smithii* and *E. trachycaulum* have high establishment and reproductive rates. Launchbaugh (1964) showed ultimate stand dominance of cool season grasses did not depend solely on seeding rate or seeds in mixes. In fact, the low seeding rate (15 kg/ha) might have improved wheat grass establishment, confirmed by Launchblaugh and Owensby (1970) who found establishment of cool season native grasses decreased with higher seeding rates. Although *E. trachycaulus,* an early seral species, will die out within 5 to 10 years (Desserud 2006), its initial dominance is probably sufficient to suppress slow growing species such as *F. hallii*.

3.6 Conclusions and Management Considerations

This study demonstrated the possibility of restoring rough fescue grassland species. The success of *F. hallii* with little competition, underscores the importance of reducing the amount and number of aggressive species, in rough fescue grassland reclamation seeding. Seeding rates should be no more than 15 kg/ha as bare patches will allow infill from adjacent species. While monoculture seeding of *F. hallii* is not practical due to low seed availability and high cost, seed mixes should include few or no wheat grasses, and instead a mix of other native grasses common in the area.

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Other	Fescue	Native Mix	Other	Native Mix	Fescue	Native Mix	Other	Native Mix	Fescue	Native Mix	Fescue
	X	X				X			X		
				X		X			X		X
		X			X			X			
				X				X	X		X
	X				X	X			X		
	X	X		X					X		
					X			X			
	X				X	X					
		X		X	X			X			X
	X	X		X		X					
		X			X			X	X		
				X					X		

Figure 3.1 Experimental design schematic showing seeding strips and 6 - 7 m^2 subplots. Fescue = *Festuca hallii*. See Table 3.1 for seed mixes.

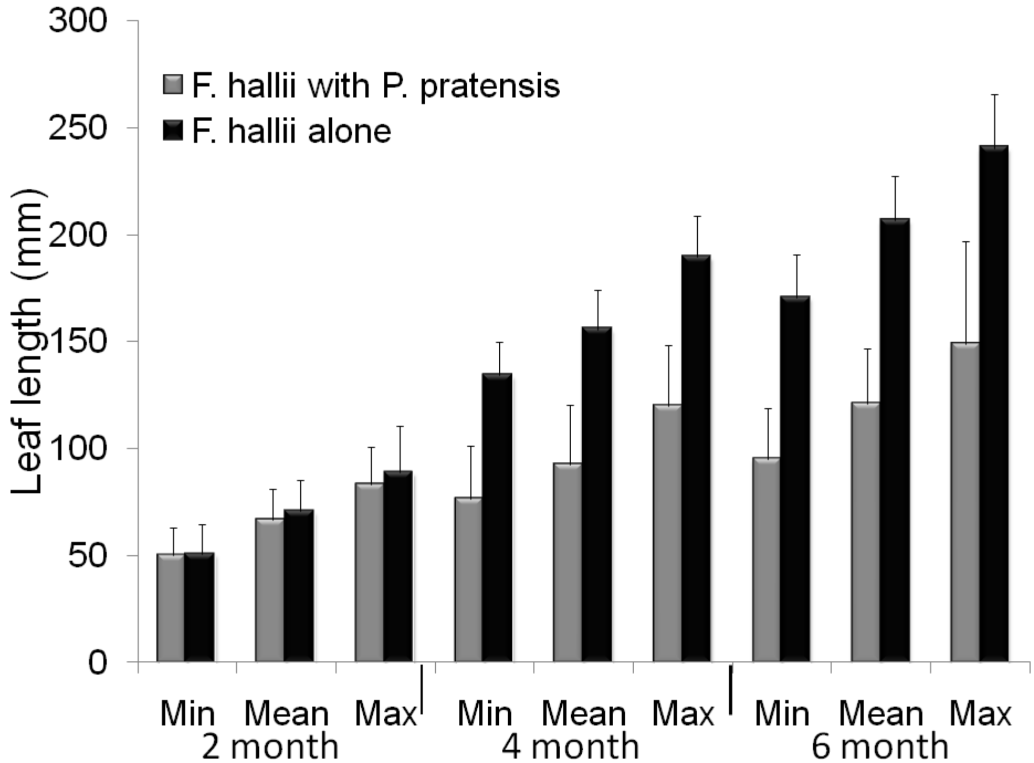

Figure 3.2 Comparison of *Festuca hallii* leaf length, grown in the greenhouse alone and with *Poa pratensis*. Error bars are standard deviation. All differences, except two-month minimum leaf lengths, were significant ($P < 0.001$). Error bars are standard deviation.

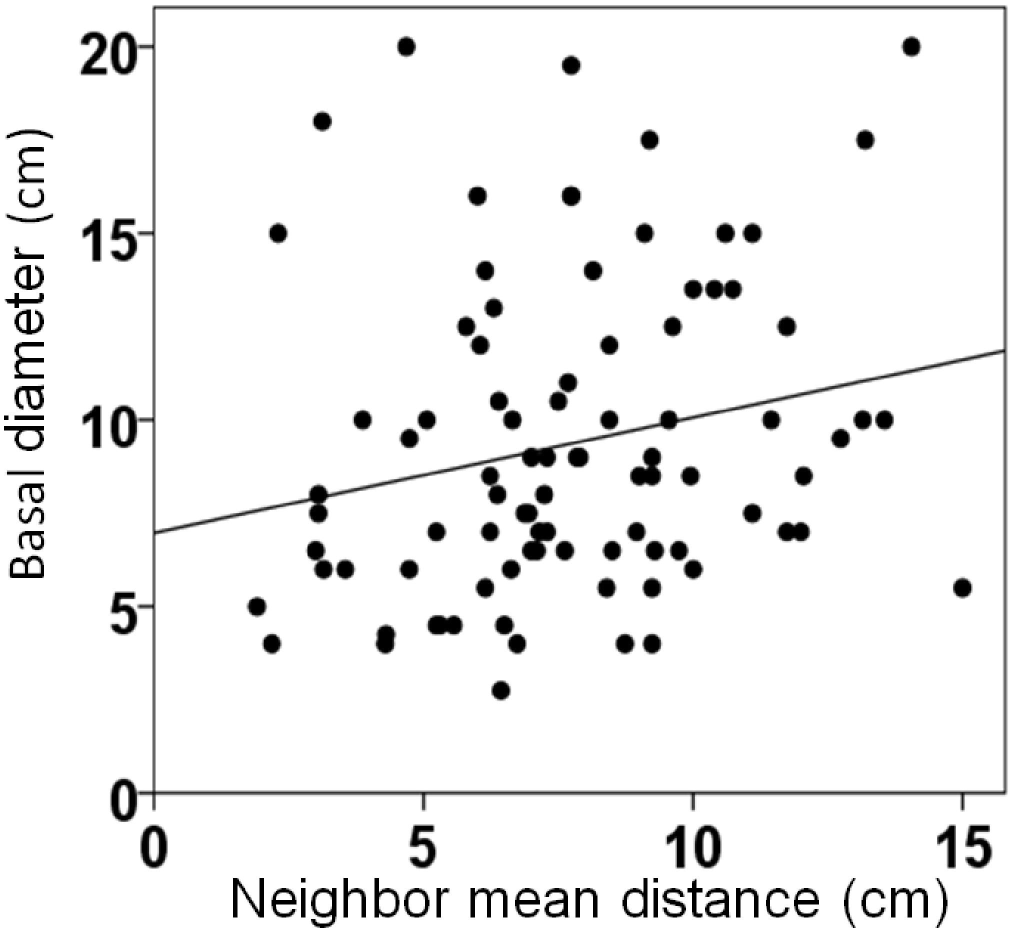

Figure 3.3 Linear regression of three-year-old *Festuca hallii* basal diameter (cm) and nearest neighbouring plants mean distances (cm), showing a trend of greater basal diameter the further away the neighbours (P = 0.043, r^2 = 0.05).

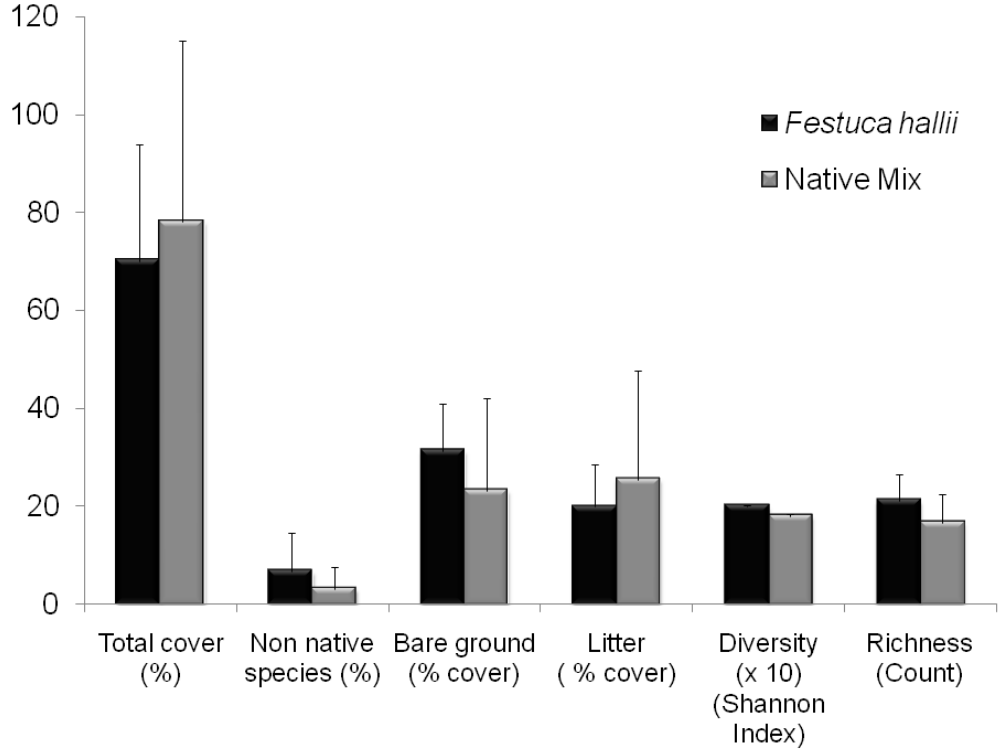

Figure 3.4 Comparison of *Festuca hallii* and native mix seeding treatments on three field sites, in year 3, showing no significant differences for total cover (P = 0.303), differences for bare ground (P = 0.012), litter (P = 0.027) and species diversity (P = 0.004) and richness (P = 0.010). Error bars are standard deviation.

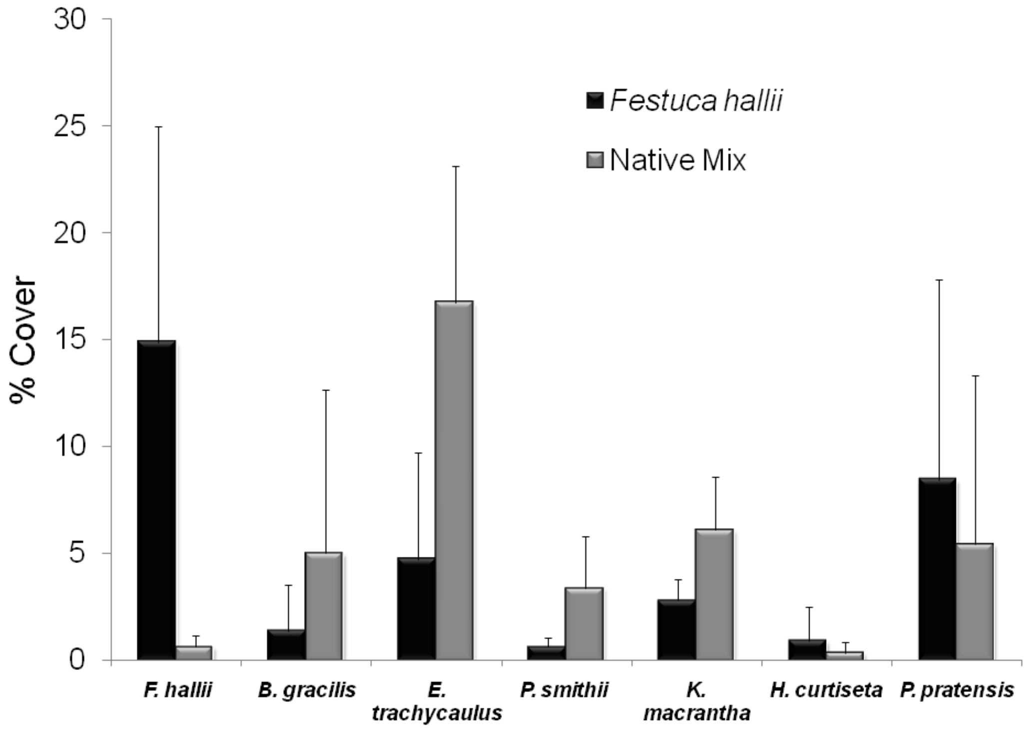

Figure 3.5 Comparison of *Festuca hallii* and native mix seeding treatments, showing third year results for seeded species *Festuca hallii*, *Bouteloua gracilis*, *Elymus trachycaulus*, *Pascopyron smithii*, *Koeleria macrantha*, *Hesperostipa curtiseta* and *Poa pratensis.* All differences in cover were significant ($P < 0.001$). Error bars are standard deviation.

Table 3.1 Seed mixes showing percent of total mix, kg/ha and seeds per m^2 per experimental design row.

	Seed mix%	kg/ha	Seeds/m^2
Ellerslie and Byemoor			
Festuca hallii[1]	99	15.5	1,373
Other native species	1		n/a
Native Mix			
Festuca hallii	20	1.3	119
Koeleria macrantha	35	2.3	809
Nassella viridula	40	2.7	107
Elymus trachycaulus	3	0.2	5
Pascopyrum smithii	2	0.1	3
Drumheller			
Festuca hallii	99	15.5	1,373
Other native species	1		n/a
Native Mix			
Festuca hallii	20	1.3	119
Koeleria macrantha	20	1.3	462
Hesperostipa comata	25	1.7	42
Bouteloua gracilis	30	2.0	366
Elymus trachycaulus	3	0.2	5
Pascopyrum smithii	2	0.1	3

[1] *Festuca hallii* seed was wild harvested and following cleaning contained trace amounts of other native species.

Table 3.2 Selected species comparing cover on plots seeded with *Festuca hallii*, a native mix and undisturbed controls, in the third year. Means with different letters within a column are significantly different as determined by one-way ANOVA and Tukey HSD. Standard deviation of means are in parentheses.

	Festuca hallii	Native Mix	Control	*P*
Bouteloua gracilis	1.4a (2)	5.0b (8)	1.9a (3)	**0.001**
Carex spp.	0a	0a	5.4b (3)	**<0.001**
Festuca saximontana	11.0a (12)	9.2a (12)	0b	**<0.001**
Elymus trachycaulus	4.7a (5)	16.7b (6)	0.5c (1)	**<0.001**
Festuca hallii	14.9a (10)	0.6b (1)	18.1a (16)	**<0.001**
Hesperostipa curtiseta	0.9a (2)	0.3a (1)	9.6b (9)	**<0.001**
Hesperostipa comata	0.1 (0)	0.4 (1)	0	**0.039**
Koeleria macrantha	2.8a (1)	6.1b (3)	4.3c (5)	**<0.001**
Nassella viridula	3.0a (4.1)	15.3b (12)	0.6c (2)	**<0.001**
Pascopyrum smithii	0.6a (0)	3.4b (2)	1.0a (2)	**<0.001**
Poa pratensis	8.4a (9)	5.4b (8)	1.1c (2)	**0.012**
Total grasses	42.1a (11)	56.6b (18)	48.7a (28)	**<0.001**
Achillea millefolium	0.2a (0.3)	0.3a (1)	0.8b (1)	**0.006**
Artemisia frigida	9.6 (12)	7.8 (12)	5.3 (5.8)	0.883
Cerastium arvense	0.2a (0.6)	0.0b	0.6a (1)	**<0.001**
Total forbs	11.0a (8)	8.0a (6)	21.4b (14)	**0.001**
Chenopodium album	0.3a (0.7)	0.1a (0.2)	1.5b (2)	**0.003**
Cirsium arvense	0.1a (0)	0.1a (0.1)	2.2b (4)	**<0.001**
Taraxacum officinale	0.2a (0.4)	0b	0b	0.285
Total introduced	2.7a (4.2)	1.3a (2)	0b	0.081
Total cover	2.8a (4)	1.7b (3)	0c	0.050
Moss and lichens	6.7a (8)	3.2b (4)	0c	**<0.001**
Bare ground	70.1a (24)	78.1a (37)	105b (37)	**0.020**
Litter	0a	0a	6.6b (7)	**<0.001**
Diversity	3.1a (10)	2.3a (19)	2.9b (3)	**<0.001**
Richness	19.9 (9)	25.5 (22)	41.8 (31)	0.085

CHAPTER 4. NATURAL RECOVERY OF PIPELINES

4.1 Introduction

Grassland restoration after industrial disturbance that removes all vegetation and soil is highly complex. Pipeline construction impacts include vegetation stripping, leaving subsoil on the surface, admixing soil horizons, compacting soils and destroying biological crusts including moss and lichens (Naeth et al. 1987; Lovich and Bainbridge 1999). Resulting disturbed land is in turn subjected to weather, increasing susceptibility to runoff and erosion, faunal disturbance and invasion by non-native plant species.

Rough fescue grassland is an important ecosystem and forage source in western Canada and a rapidly diminishing natural resource (Romo et al. 1990; Clark 1998). Although efforts have been directed to its restoration after disturbance, they have mostly been unsuccessful. Attempts, to assist pipeline restoration in fescue grasslands, included soil manipulation during construction, such as topsoil stripping and replacement, and revegetation methods such as seeding native species or non-native cover crops. Seeding native rough fescue (*Festuca hallii* (Vasey) Piper or *Festuca campestris* Rydb.) has had little documented success. Desserud et al. (2010) and Elsinger (2009) reported numerous examples of unsuccessful revegetation of rough fescue following seeding on pipelines and well sites in central and southern Alberta.

More recent restoration efforts involved minimum disturbance techniques with plough-in and narrow ditch pipelines, a 2 to 3 m width disturbance (excavation bucket width), and sites left to revegetate via natural recovery. In central Alberta, Petherbridge (2000) concluded pipeline construction with no topsoil stripping resulted in a smaller initial disturbance and conserved more of the initial plant community than conventional right-of-way (ROW) stripping, although admixing reduced emergence of some native species. The time required for perennial establishment may expose the site to erosion and non-native species (Hammermeister and Naeth 1996). Some researchers have commented that although initial species emerging during natural recovery are undesirable forbs, slower growing perennials, such as rough fescue will eventually dominate (Naeth et al. 1997; Arychuk 2001; Hammermeister 2001). Lathrop (1980) concluded age was a positive determinant for linear disturbance recovery, although extent of the disturbance had a greater negative impact.

Landscape ecology must be considered prior to determining reclamation techniques. Heavily altered landscapes, such as cultivated areas, would not be candidates for natural recovery (Prach et al. 2011). To succeed, natural recovery requires availability of diaspore sources and suitable site factors for establishment (Tischew and Kirmer 2007). In native grassland, natural recovery is dependent on recruitment from the seed bank, intact vegetation in sod segments, seed rain from nearby species and wind-born or faunal-carried seed (O'Neill 1998; Petherbridge 2000; Wang et al. 2006). Natural recovery has high potential for establishment of a diverse native plant community similar to pre-disturbance species. Bare ground is a limitation, which may facilitate establishment of non-native perennials, and increase susceptibility to soil erosion. In mixed grass prairie, Soulodre (2001) noted natural recovery resulted in moderate erosion and high diversity with mid-seral species, despite a slow progression towards pre-disturbance status.

Soil exposure following disturbance may allow seed bank propagules to germinate, which, if allowed to recover without competition from seeded species, may become established. Revel (1993) and Petherbridge (2000) found species from the seed bank of replaced sod aided in grassland restoration; however, Soulodre (2001) found fewer than expected seeds in the seed bank following disturbance in central Alberta. Bischoff (2002) concluded seed rain was more important than seed bank in restored grassland in a German floodplain.

Opinions differ on the time required to achieve natural recovery. Allen (1993) suggests return to the original ecosystem is typically a slow process not accomplished within a human lifetime. Dobson et al. (1997) estimated recovery to pre-disturbance plant cover and biomass may take 50-300 years, while Lovich and Bainbridge (1999) suggest ecosystem recovery may require over 3000 years in places like the Mojave Desert. In alpine and permafrost conditions, Jin et al.

(2008) estimated that where vegetation and soils were severely damaged, it would take 20-30 years for alpine grasslands to recover their ecological structures and biodiversity similar to the original conditions. In western Canadian grasslands, Nasen (2009) found impacts on soil and vegetation by oil and gas development persisted over 50 years, while Naeth (1985) concluded there was evidence of succession towards undisturbed vegetation on a pipeline after 26 years.

In this study, pipeline ROWs constructed in undisturbed grasslands and left to recovery naturally were evaluated. Undisturbed controls and seed bank propagules were compared to ROW species to assess the effects of age and pipeline construction methods by examining vegetation cover and biological crust conditions. The objective was to assess which construction methods would be most successful in returning grassland species.

4.2 Methods

4.2.1 Study Area

The study area, an uncultivated natural area grassland, was located in Alberta, Canada, in the Rumsey Natural Area, in the Central Parkland natural region, between latitudes 51.796° and 51.883° and longitudes 112.417° and 112.701°. Temperature maximums range from -40 to +35 °C, with growing season (May to October) temperatures averaging 13 °C. Average annual rainfall is approximately 350 mm and snowfall 100 cm. Topography is undulating, with a complex of small depressions and hills. Soils are loamy Dark Brown Chernozems on medium textured glacial till. Native vegetation on uplands and upper slopes is rough fescue grassland, dominated by *F. hallii*, *Hesperostipa curtiseta* (A.S. Hitchc.) Barkworth (western porcupine grass), *Koeleria macrantha* (Ledeb.) J.A. Schultes (June grass) and *Poa* spp. (bluegrasses). Non-native species occur on some disturbed sites, including *Cirsium arvense* (L.) Scop. (Canada thistle), *Taraxacum officinale* G.H. Weber ex Wiggers, *Chenopodium album* L. (lamb's quarters), *Tanacetum vulgare* L. (tansy), *Crepis tectorum L.* (hawksbeard) *and Gutierrezia sarothrae Pursh) Britton & Rusby* (broomweed).

4.2.2 Study Design

Three natural gas pipeline ROW segments, each with 15 cm diameter pipes, were studied between 2007 and 2010. Pre-disturbance sampling on each ROW was conducted at construction times and restoration monitoring commenced the first growing season following construction (Prach et al. 2001).

In September 2007, the first pipeline was installed using a plough-in technique. A plough, with an 80 cm-wide bucket, created a narrow trench, the width of the bucket; 15 cm-diameter pipe was fed into the trench the same day; and soil and sod was allowed to fall back into place. Vegetation was assessed along the ROW route in September 2007 prior to pipeline construction to serve as a control. Three 50 m long transects were located along the proposed pipeline trench at three aspects: south-facing, north-facing and a crest. On each transect five 20 by 50 cm rectangular quadrats (Daubenmire 1959), were located 10 m apart, and within each quadrat, foliar cover of all species, and litter and bare ground, were visually assessed.

In November 2008, the second pipeline was installed with a SpiderPlow© trencher; whereby sod is cut and parted, soil displaced, the flexible pipe threaded and sod and soil allowed to fall back into place immediately following pipe lowering (SpiderPlow 2008). Native vegetation differed, with the southwest end being hummocky and dominated by *F. hallii* and *H. curtiseta,* while the northeast end was an upland plain with mainly *F. hallii* and *Poa* spp. Since natural recovery could be influenced by different species, each end of the ROW was treated as different sites. In July 2009 and 2010, vegetation was assessed at two locations on the ROW approximately two km apart. Vegetation was assessed along the ROW trench and 2 m parallel from the trench, in undisturbed grassland as a control. Fifty meter long transects, were located on the pipeline trench, along the most disturbed area, at south-facing, north-facing and crest aspects. On each transect, five 20 by 50 cm rectangular quadrats, were set 10 m apart and within each, foliar cover of all species, and litter and bare ground cover, were visually assessed.

Foliar cover of all species, litter and bare ground cover, from 10 other pipeline ROWs and adjacent undisturbed controls in the same region, collected by Elsinger (2009), were added to the data set for analyses. These included plough-in pipelines, constructed similarly to the two ROWs described above; pipelines from which topsoil was stripped the width of the pipeline trench, stored and replaced following pipe installation; and ditch witch construction where a trencher destroys sod causing a narrow, major disturbance (Table 4.1). On each pipeline ROW, a 30 m transect was positioned along the trench, the area of highest disturbance and another 15 m from the ROW, in undisturbed grassland to serve as a control. On each transect, ten 20 by 50 cm rectangular quadrats, 3 m apart were laid. Within each quadrat, foliar cover of all species and litter and bare ground, were visually assessed (Elsinger 2009).

Seed bank samples were collected at each quadrat on each ROW, immediately following construction in November 2008. Samples, approximately 36 cm^2 and 6 cm deep, were hand-cut from the soil surface with a 6 cm wide spade. Vegetation was cut to about 3 cm from the sample ground surface and removed. Litter, surface seeds and inflorescences were retained, the samples sealed in plastic bags and frozen at -5 °C until assessment (Coffin and Lauenroth 1989). Thawed samples were hand crushed, passed through a coarse sieve to remove twigs, stones, plant crowns and roots, and spread 2 cm thick over 3 cm of potting soil (1:4 vermiculite and peat) in 10 x 15 x 5 cm trays. The trays were kept in greenhouse conditions to simulate summer growing conditions (temperature average of 25 °C and ambient light average of 16 hours per day), and watered as needed with tap water. Emerging seedlings were enumerated and removed once identified, for a three month period. Any unidentified plants remaining after three months were grown for another two months to facilitate identification. Percent of each species was calculated based on total number of germinated species.

4.3 Statistical Analyses

Data from the three transects at each site were averaged. Mean cover for all plant species was calculated. Data of all sites were normally distributed; therefore, ANOVA, with Tukey post-hoc tests, was used to compare mean cover among plant community groups, and t-tests were used for comparison of bare ground and litter between ROW and control plots**.**

Two-way cluster analysis using Ward's method was used to classify ROW and control plots, with a cut-off at 50% of information remaining. Cluster analysis is based on the concept of grouping points representing individuals with similar characteristics in mathematical space (Kent and Coker 1992). Indicator species analysis (ISA) identified dominant species of plant communities derived from cluster analysis groupings. Indicator values (IV) ranged from 0 to 100, where 100 indicated a species was exclusively found in a particular group (Dufrene and Legendre 1997). Non-metric multidimensional scaling (NMS) was used to ordinate and display the separation of native grassland controls to ROW species, and to confirm cluster analysis groupings (McCune and Grace 2002). Key species, as identified by ISA, and significant factors, such as bare ground, were overlain on NMS diagrams to display trends. Sorenson (Bray-Curtis) distance measure was used to calculate statistical significance of axes with a final run of 500 iterations using three axes and a randomly selected starting configuration.

ROW plots were grouped according to dominant vegetation species for further analyses. T-tests were used to indicate significant differences between and among groups. Pearson product correlation and linear regression were used to display relationships between species and ROW characteristics. Nonparametric multiple response permutation procedure (MRPP), operating on Sorenson (Bray-Curtis) distance measures, was used to evaluate significant differences between ROW groups and controls (Zimmerman et al. 1985). MRPP generates a chance-corrected within-group agreement value (*A*), which evaluates the difference between species composition of grouped plots. The lower the *A* value, the more similar are the groups (McCune and Grace 2002).

PC-ORD (ver. 5.31, MjM Software, Gleneden Beach OR) was used to classify and ordinate. PASW (ver. 18.0, SPSS, Chicago IL) and Excel (ver. 2007, Microsoft, Redmond WA) were used for statistical analyses.

4.4 Results

4.4.1 Right-of-Way Recovery

Two plant communities were isolated on the ROWs (Figure 4.1). One was dominated by *Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould ssp. *lanceolatus* (northern wheat grass) and *Pascopyron smithii* (Rydb.) A. Löve (western wheat grass) (Wheat Grass community); and the other by *Poa* spp. and *F. hallii* (Fescue-Bluegrass community) (Table 4.2).

NMS ordination confirmed ROW cluster classification, with three axes explaining 89% of the variation and stress of 5.8, indicating good ordination with no risk of false inferences (McCune and Grace 2002). *Elymus lanceolatus* and *P. smithii* trended toward Wheat Grass communities, while *F. hallii* and *Poa* spp. trended toward the Fescue-Bluegrass community (Figure 4.2; Table 4.2). The Wheat Grass community generally had older sites than the Fescue-Bluegrass community (Table 4.1).

ROW age was not correlated to any species, except moss and lichens, which increased significantly with age (Figure 4.3). Bare ground, although not significant, decreased with age (Figure 4.3). Pipeline construction techniques affected *F. hallii* cover which was significantly greater on plough-in pipelines, and of wheat grasses which was significantly greater with ditch-witching and topsoil-stripping (Figure 4.4). Seventy percent of Fescue-Bluegrass ROWs were plough-ins. Two topsoil-strip pipelines in this community were the oldest (Table 4.1). All Wheat Grass community ROWs were topsoil-striped or ditch-witched.

4.4.2 Right-of-Way Recovery Compared to Controls

Festuca hallii, moss and lichen and litter cover were significantly greater on controls than ROWs. *Elymus lanceolatus*, *Elymus trachycaulus* (Link) Gould ex Shinners ssp. *subsecundus* (Link) A.& D. Löve (slender wheat grass), *P. smithii* and bare ground cover were significantly lower on controls than ROWs. Forb and shrub cover and species diversity and richness were similar between controls and ROWs (Figure 4.5).

Controls associated with ROW communities were dominated by *F. hallii* (Table 4.2), except for three controls of the Fescue-Bluegrass community, which were co-dominated by *Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths (blue grama). The Wheat Grass community was significantly dissimilar to its controls ($A = 0.11$, $p < 0.001$). *Elymus lanceolatus* and *P. smithii* were significantly greater on ROWs than on controls, while *F. hallii*, *H. curtiseta* and moss and lichens were significantly less on ROWS than controls. Species with similar abundance in Wheat Grass community ROW and controls included *K. macrantha*, *Carex* spp. (sedges), *Poa pratensis* L. (Kentucky bluegrass) and *Artemisia frigida* Willd. (fringed sage) (Table 4.2).

Fescue-Bluegrass ROW community and controls were similar ($A = 0.03$, $p = 0.09$), although Fescue-Bluegrass controls had significantly more *F. hallii* and less *E. lanceolatus* and bare ground ($P = 0.025$) (Table 4.2). Various species with similar abundance in Fescue-Bluegrass ROW and controls were *K. macrantha*, *Carex* spp., *H. curtiseta, A. frigida* and moss and lichens (Table 4.2).

NMS ordination of the ROWs confirmed classification and MRPP analyses. Wheat Grass community ROW and controls were clearly separated, with 86% of variation explained on two axes and a stress of 10, indicating good ordination (McCune and Grace 2002) (Figure 4.6). Moss and lichens, *F. hallii* and litter, correlated with controls; while *E. lanceolatus*, *P. smithii*, *A. frigida* and bare ground, correlated with ROWs. Fescue-Bluegrass ROW and controls were not as clearly separated, indicating some degree of similarity. Eighty-nine percent of the variance was explained on three axes with a stress of 5.3, a good ordination (McCune and Grace 2002). Bare ground, *P. smithii* and *Poa* spp. trended towards ROWs, *B. gracilis* trended towards controls, and *E. trachycaulus* and *F. hallii* trended towards both ROWs and controls. Ordination indicated similarity of three ROWs and controls: 1991-1, 1999-2 and 2009-1 (Figure 4.7).

Comparison of years one and two of three plough-in pipeline ROWs, indicated resistance of *F. hallii*, resilience of *H. curtiseta*, *P. smithii* and *E. lanceolatus. Koeleria macrantha* was neither resistant nor resilient. *Poa* spp. and *E. trachycaulus* were opportunistic (Figure 4.8). Litter increased between years 1 and 2, while shrubs, forbs and bare ground remained the same (Figure 4.9). *K. macrantha*, *H. curtiseta* and litter were significantly greater on controls than ROWs; (Figure 4.8 and Figure 4.9). *Poa* spp., *E. lanceolatus, E. trachycaulus* and bare ground ($P < 0.001$) were significantly greater on ROWs than on controls (Figure 4.8 and Figure 4.9).

4.4.3 Seed Banks

The seed bank had fewer species than years 1 and 2 of the ROWs. Annual forbs dominated, including *Androsace septentrionalis* L. (fairy candelabra), *A. frigida* and *Vicia americana* Muhl. ex Willd. ssp. *Americana* (American vetch). Grasses which emerged from the seed bank were *Poa* spp., *F. hallii*, *E. lanceolatus*, *H. curtiseta*, *K. macrantha* and *B. gracilis.* In all cases, except *Poa* spp., seed bank values were less than ROW cover. Species missing from the seed bank, but found on ROWs were *Carex* spp., *E. trachycaulus* and *P. smithii*.

4.5 Discussion

Natural recovery may be a viable disturbance restoration mechanism in native grassland, resulting in plant communities similar to original grassland. These results confirm Prach et al. (2001) recommendations that landscape ecology must be suitable for natural recovery to succeed. Rough fescue grassland species occurred on over half of the sites and similarity between Fescue-Bluegrass communities and controls indicated success of natural recovery. Pipeline construction techniques and degree of disturbance also affected natural recovery success. In this study, plough-in pipeline construction reduced disturbance intensity with retention of much of the original sod, including presumably viable perennial grass roots. Plough-in pipelines also had the fewest non-native species, the greatest rough fescue recovery, and were similar to native grassland controls. Maki (1989) also found plough-in construction can avoid induced germination of non-native species and other researchers noted plough-in pipelines had comparable species richness to undisturbed prairie when revegetated by natural recovery (Petherbridge 2000; Elsinger 2009; Desserud et al. 2010).

Wheat grass-dominated pipelines were associated with the greatest disturbance in this study, top-soil-strip or ditch-witch construction techniques. Elsinger (2009) observed ditch-witch construction left a poor growth medium for plants to establish through natural recovery. Despite being some of the oldest pipelines their covers were dissimilar to controls and still dominated by wheat grasses. The disturbance extent and ensuing bare ground would provide suitable habitat for wheat grass establishment, having high germination rates, rapid establishment and rapid ground cover production. These findings are similar to Hammermeister (2001) who suggested aggressive wheat grasses may suppress establishment of other species, and Desserud et al. (2010) who observed wheat grasses dominated top-soil-stripped pipelines over ten years of age. Ostermann (2001), who found rhizomatous non-native and native grasses dominating the trenches of 12 year old pipelines in mixed grass prairie and foothills grasslands, suggested wheat grasses got an early start, and remained dominant. Seed banks on the newly constructed plough-in pipelines yielded few perennial grasses, and many annual forbs, in contrast to species composition found on the recovering pipelines. This may have been due to soil admixing; whereby collected soil samples included subsoil, with fewer seeds. These results are similar to those of Soulodre (2001), who found fewer than expected seeds in the seed bank, and to Petherbridge (2000) and Hammermeister (2001), who found more seedlings emerged in soil collected from stripped compared to plough-in treatments, attributing the difference to admixing of soil in plough-in construction. Despite reported dominance of annual forbs in early disturbances (Naeth et al. 1997; Arychuk 2001; Hammermeister et al. 2003), perennial graminoids (e.g. rough fescue) remained on the plough-in ROWs in the first year, most likely due to intact sod.

Other studies including Revel (1993) and Petherbridge (2000), attest to the value of the seed bank in replaced sod for fescue grassland restoration. By the second year, perennial grasses not found in the ROW seed bank appeared, likely resulting from seed rain originating in adjacent grassland, which, as Desserud et al. (2010) suggested, would occur on narrow-width disturbances such as a plough-in pipeline. Diaspore sources may be more important than seed bank in grassland recovery (Kirmer and Mahn 2001; Tischew and Kirmer 2007).

Litter, a critical grassland component, was reduced on all ROWs compared to undisturbed grassland. Plant litter helps conserve soil water by reducing soil temperature and evaporation, and insulating soil against incident radiation by reducing light and temperature at the soil surface (Hopkins 1954; Johnston et al. 1971; Willms et al. 1985; Willms et al. 1986). Litter may have assisted plant community development in Fescue-Bluegrass ROWs in this study, by regulating soil water as Willms et al. (1986) and Foster and Gross (1998) suggest, and by providing micro-climates for various seeds as found by Molofsky and Augspyrger (1992). Conversely, lack of litter on Wheat Grass ROWs may have impeded plant community development. *Festuca hallii* recovery on ROWs with greater litter in this study is consistent with findings by Willms (1988), who observed reduced rough fescue growth when litter was removed.

Despite evidence of grassland recovery on some ROWs, bare ground was apparent on all ROWs, although bare ground decreased as ROWs aged. This is consistent with findings by Hammermeister (2001), who found bare ground was significantly greater on natural gas well sites than undisturbed controls, and by Ostermann (2001), Desserud (2010), and Elsinger (2009), who discovered bare ground persisting on pipelines in central and southern Alberta. Bare ground indicates little biological crust, an important component of grasslands. Biological crusts consist of microorganisms (e.g. algae, cyanobacteria) and nonvascular plants (e.g. mosses and lichens) that grow on or just below the soil surface (Pellant et al. 2005). They are important as cover and in stabilizing soil (Belnap and Gillette 1998), and may increase or reduce water infiltration or enhance soil water retention (Pellant et al. 2005). Moss and lichen cover responded more to age than pipeline construction method, occurring on older ROWs, not on newly constructed pipelines, similar to what Petherbridge (2000) reported on a pipeline work zone shortly after construction. Complete elimination of mosses was reported from top-soil-stripped ROWs in Solonetzic and Chernozemic mixed grass prairie (Naeth 1985; Petherbridge 2000; Soulodre 2001). Ostermann (2001) reported the same for a top-soil-stripped 12 year old pipeline in mixed grass prairie. In this study, moss and lichens disappeared from most disturbed sites, and as Evans and Bishop (1999) concluded, they require many years to recover.

Does natural recovery require 50 years as suggested by Dobson (1997) or over 100 years as claimed by Allen (1993), to return a disturbance to its original plant community? Plough-in pipelines showed evidence of recovery, with plant communities similar to original grassland within 2 to 23 years. Moss and lichens were starting to appear on the older pipelines; however, bare ground remained, indicating lack of a key feature of grassland ecosystems, biological crust.

4.6 Conclusions and Management Considerations

These results support the importance of minimum disturbance in grasslands. Intact sod reduces soil exposure, preventing invasion by rhizomatous species and allowing other native species to propagate the narrow disturbance through seed rain or existing propagules. Rough fescue, in particular, appears to recover better on plough-in pipelines ROWs than from seeding, most likely from remaining intact sod. Retaining grassland sod through plough-in and keeping disturbance as narrow as possible is critical to successful restoration. Narrow trenching with plough-in pipelining techniques is recommended for grasslands.

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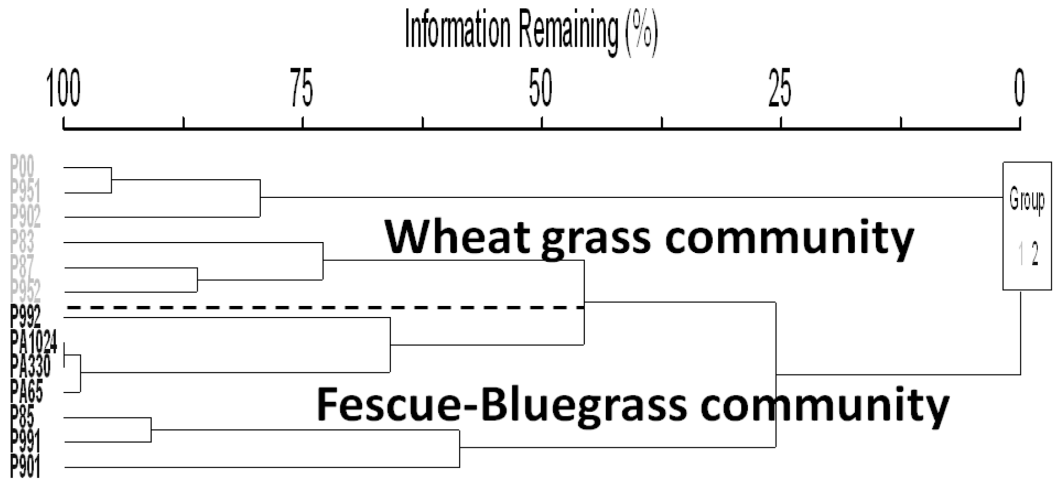

Figure 4.1 Two-way cluster dendogram showing classification of ROW plots into two plant communities with approximately 50% of information remaining: Wheat grass and Fescue-bluegrass.

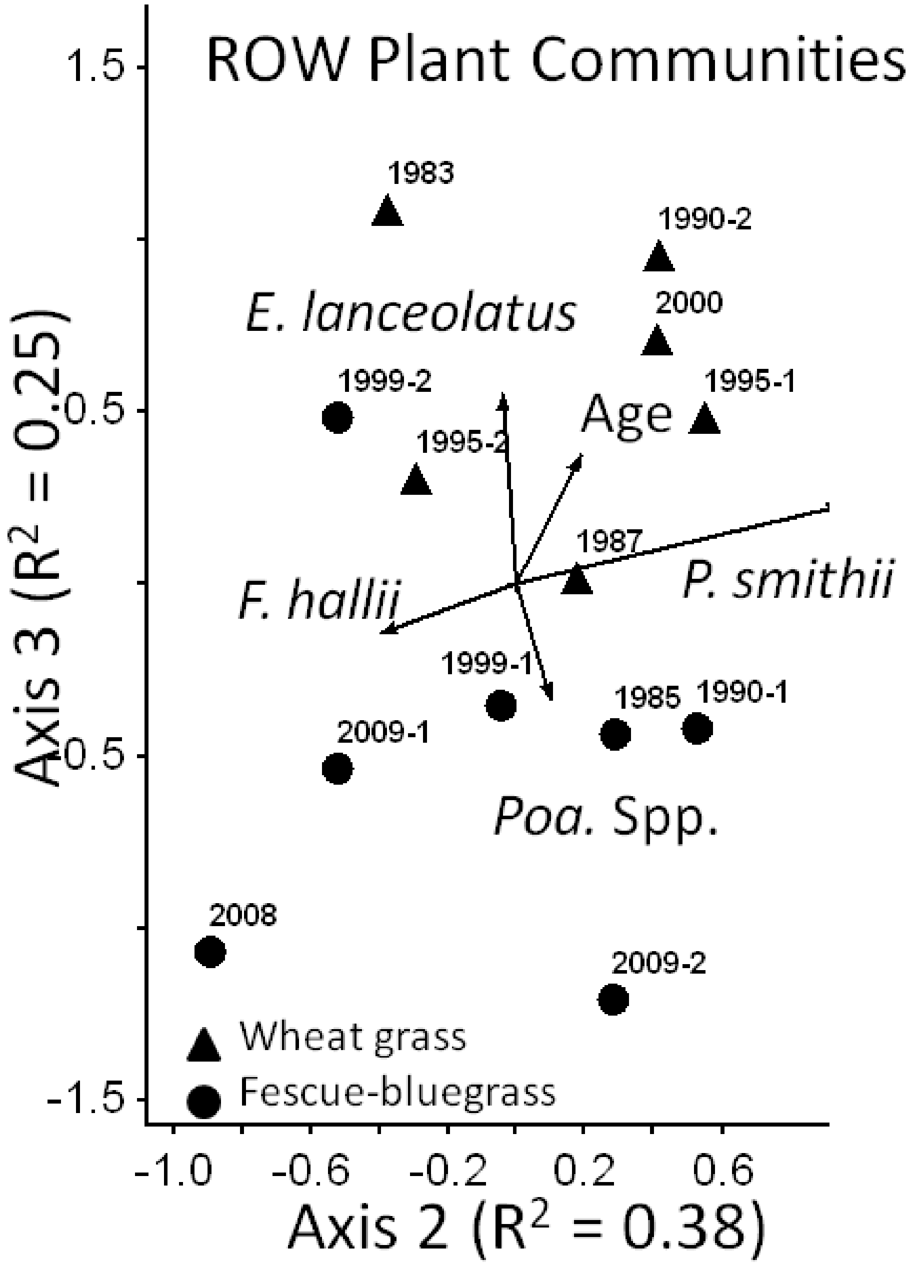

Figure 4.2 NMS ordination of ROW plots with two plant communities represented by different symbols (Wheat grass and Fescue-bluegrass). Overlays indicate significant trends along axes 2 and 3 for ROW age, *Festuca hallii*, *Elymus lanceolatus*, *Pascopyron smithii* and *Poa* species.

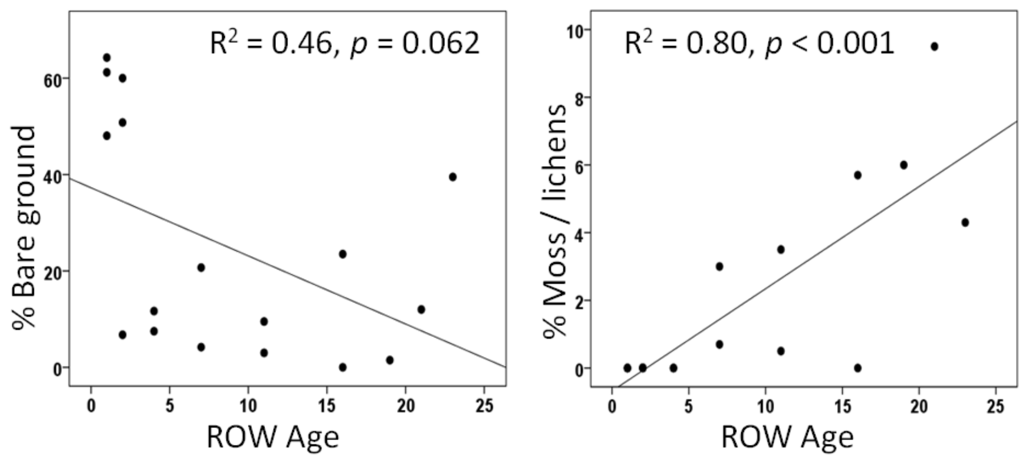

Figure 4.3 Linear regression of ROW age with bare ground (R^2 = 0.46, *P* = 0.062) and moss and lichens cover (R^2 = 0.80, *P* < 0.001).

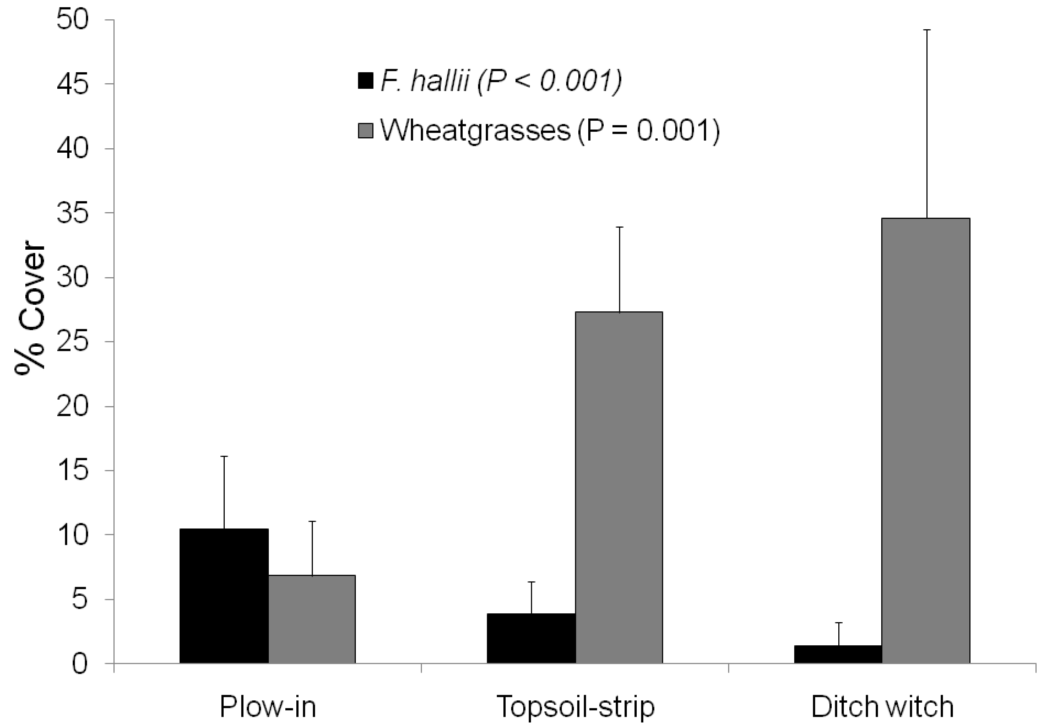

Figure 4.4 *Festuca hallii* and wheat grass (*Elymus lanceolatus, Elymus trachycaulus, Pascopyron smithii)* cover in relation to pipeline construction techniques. *Festuca hallii* (*P* < 0.001) decreased with degree of disturbance, while wheat grasses (*P* = 0.001) increased.

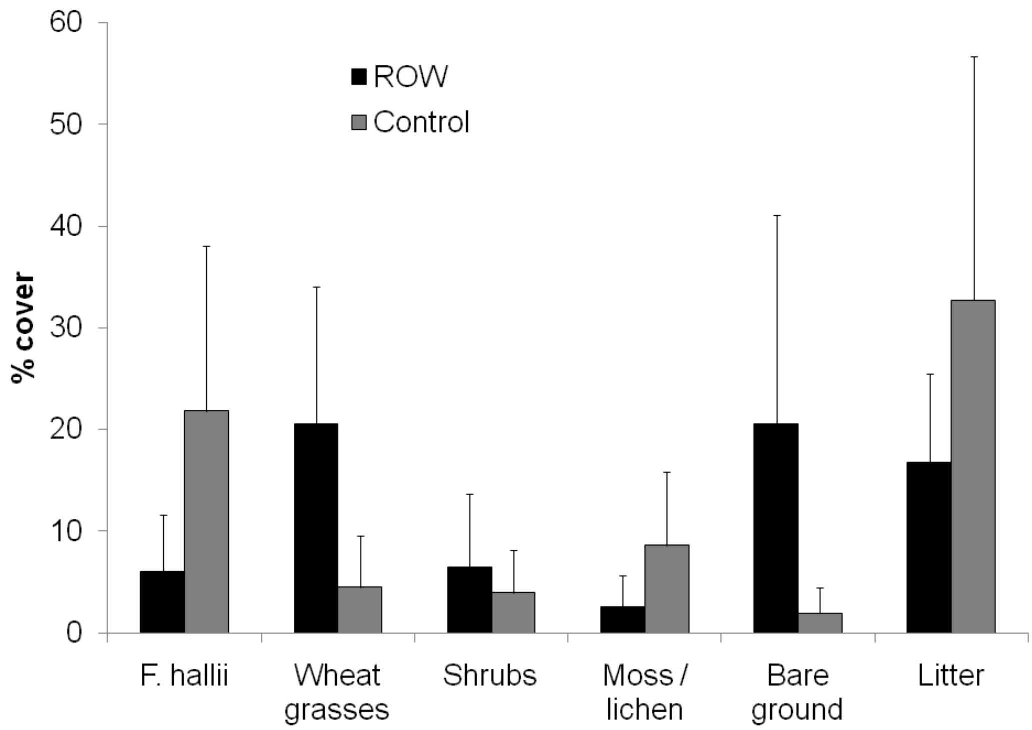

Figure 4.5 Comparison of all control and ROW plots showing mean percent cover of *Festuca hallii* (P = 0.003), wheat grasses (P < 0.001), shrubs (P = 0.279), moss/lichen (P = 0009), bare ground (P < 0.001) and litter (P = 0.011). Error bars are standard deviation.

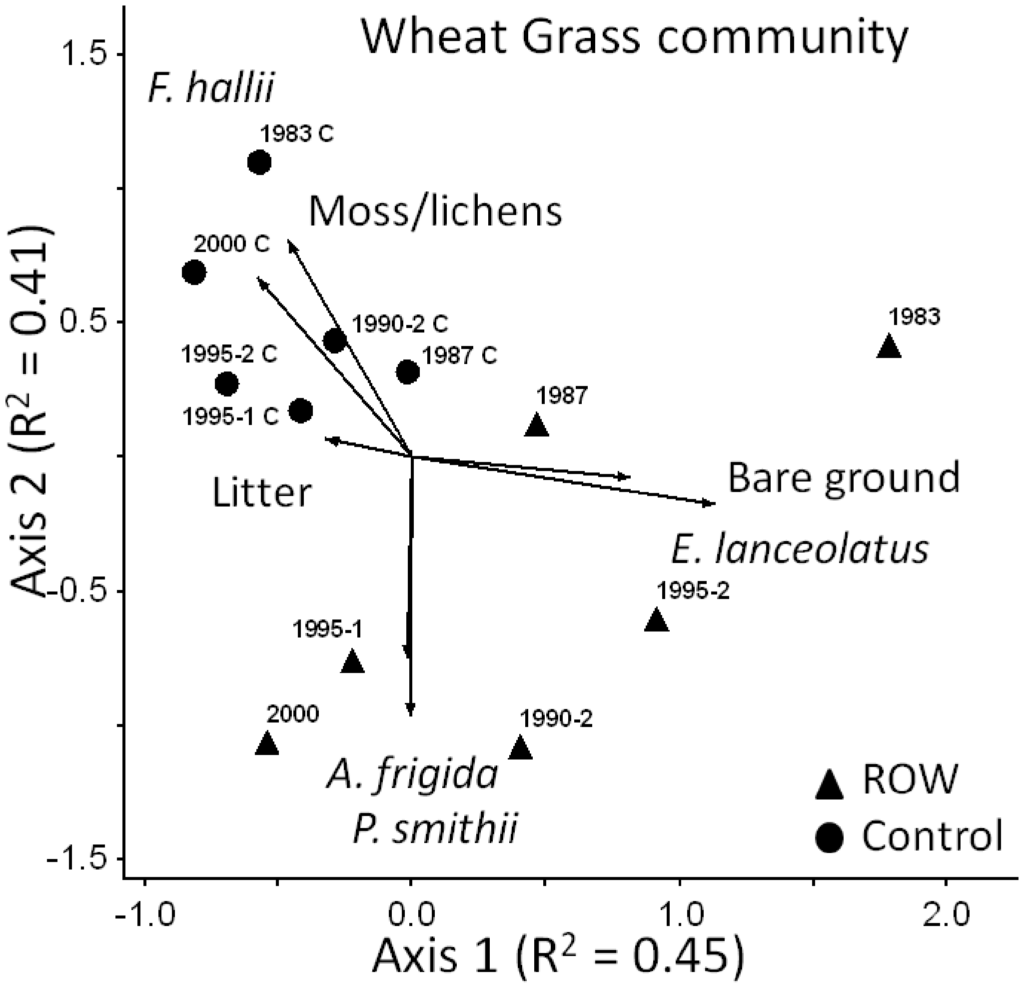

Figure 4.6 NMS ordination of Wheat Grass community ROW and corresponding control plots. Overlays indicate significant trends along axes 1 and 2 indicate significant trends for *Festuca hallii*, moss and lichens and litter towards control plots, bare ground, *Elymus lanceolatus* and *Artemisia frigida* towards ROW plots.

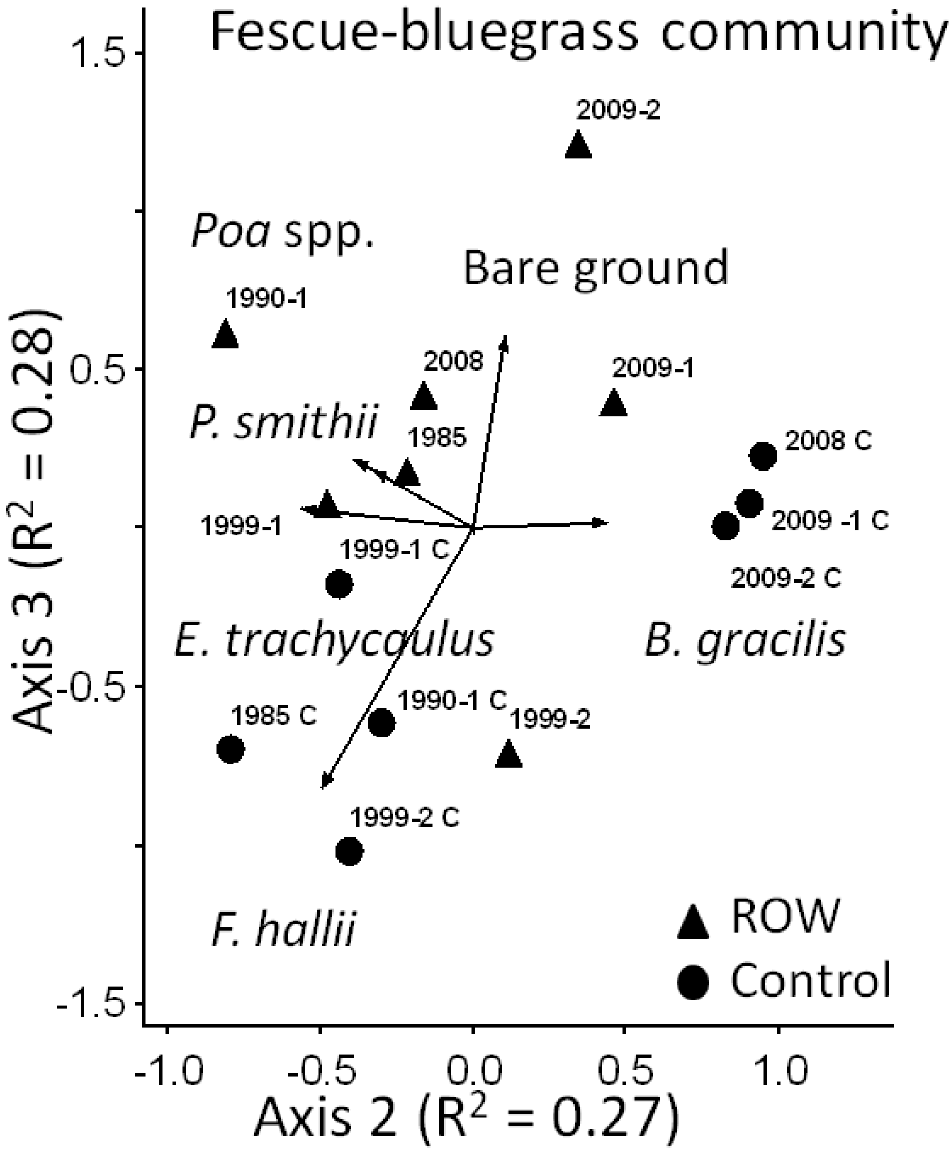

Figure 4.7 NMS ordination of Fescue-bluegrass community ROW and corresponding control plots. Overlays indicate significant trends along axes 2 and 3, indicate significant trends for *Bouteloua gracilis* towards control plots, bare ground, *Pascopyron smithii* and *Poa* spp. towards ROW plots and *Festuca hallii* and *Elymus trachycaulus* towards both ROW and controls.

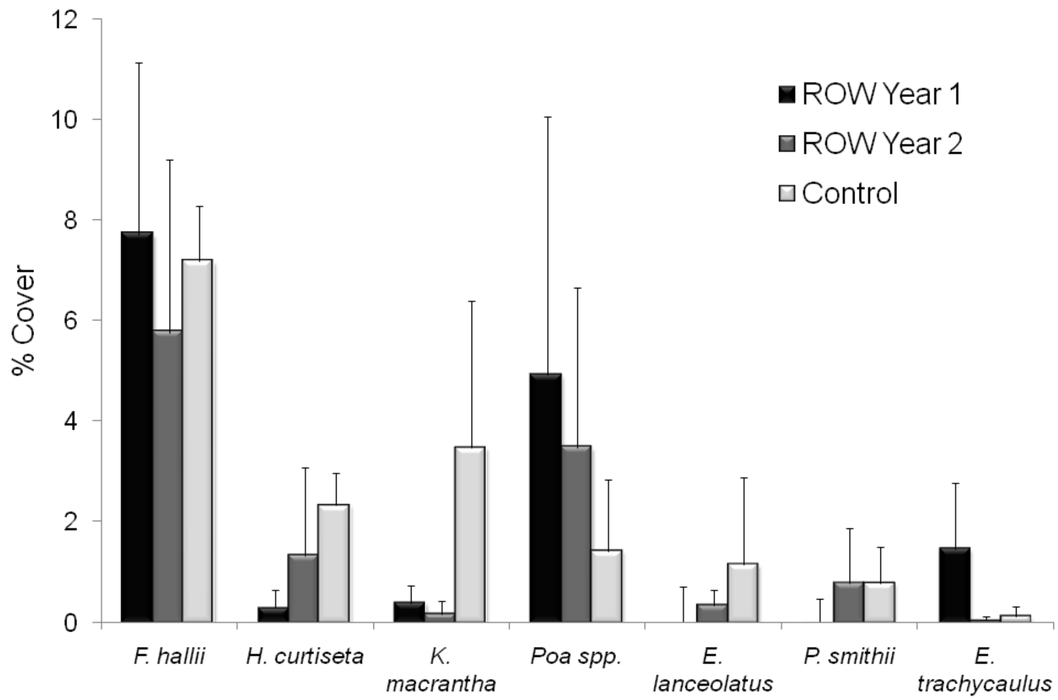

Figure 4.8 Comparison of species on newly constructed plough-in pipelines in years 1 and 2 with corresponding controls: *Festuca hallii* (*P* = 0.213), *Hesperostipa curtiseta* (*P* = 0.001), *Koeleria macrantha* (*P* < 0.001), *Poa* spp. (*P* = 0.024), *Elymus lanceolatus* (*P* < 0.001), *Pascopyron smithii* (*P* = 0.113) and *Elymus trachycaulus* (*P* = 0.018).

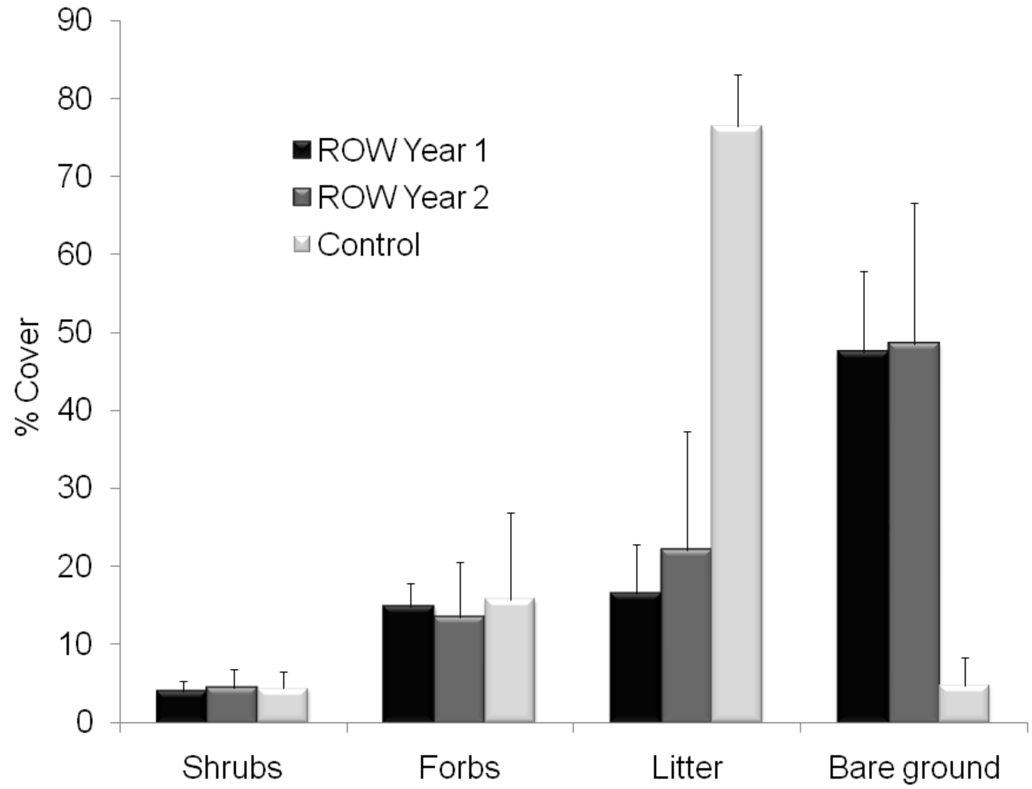

Figure 4.9 Comparison of shrubs (P = 0.188), forbs (P = 0.522), litter (P < 0.001) and bare ground (P < 0.001) on newly constructed plough-in pipelines in years 1 and 2 with corresponding controls.

Table 4.1 ROW sites showing year built, age when vegetation sampled, plant community classification, and, construction method. Numbers following age indicate different ROWs constructed in the same year. Construction method is explained in the text.

First Growing Season	Age	Plant Community	Construction Method
2009-2	2	Fescue-bluegrass	Plough-in
2009-1	2	Fescue-bluegrass	Plough-in
2008	2	Fescue-bluegrass	Plough-in
1999-1	7	Fescue-bluegrass	Plough-in
1999-2	7	Fescue-bluegrass	Plough-in
1990-1	16	Fescue-bluegrass	Topsoil-strip
1985	21	Fescue-bluegrass	Topsoil-strip
2000	6	Wheat Grass	Topsoil-strip
1995-1	11	Wheat Grass	Topsoil-strip
1995-2	11	Wheat Grass	Topsoil-strip
1990-2	16	Wheat Grass	Ditch-witch
1987	19	Wheat Grass	Topsoil-strip
1983	23	Wheat Grass	Ditch-witch

Table 4.2 ROWs grouped by dominant species and corresponding controls, mean percent cover (standard deviation) and differences between ROW and control groups tested by Indicator Species Analyses (IV) and t-tests of significance. Only species with greater than 1% cover are shown. Age is mean age of ROW construction.

	Wheat Grass Community (n=6; age = 14)					Fescue-Bluegrass Community (n=7; age = 8)				
	Row		Control			ROW		Control		
	Mean	IV	Mean	IV	*P*	Mean	IV	Mean	IV	*P*
Carex spp.	7.4 (4)	49	7.8 (2)	51	0.870	2.4 (3)	26	2.9 (4)	23	0.784
Elymus lanceolatus	15 (10)	78	4.4 (4)	22	**0.035**	3.1 (2)	67	0.9 (1)	19	**0.016**
Elymus trachycaulus	3.3 (4)	69	0.7 (1)	11	0.147	3.0 (4)	70	0.7 (1)	11	0.174
Festuca hallii	2.6 (2)	12	16 (7)	86	**0.002**	8.9 (6)	78	27 (20)	75	0.044
Hesperostipa curtiseta	7.6 (7)	26	17 (7)	69	**0.037**	8.9 (7)	46	7.8 (6)	47	0.769
Koeleria macrantha	6.9 (4)	35	9.5 (4)	58	0.327	2.0 (2)	36	2.0 (2)	43	0.992
Pascopyrum smithii	13 (12)	59	1.7 (3)	4	**0.050**	5.1 (5)	73	0.9 (2)	11	0.077
Poa pratensis	1.8 (3)	32	1.0 (2)	12	0.568	5.6 (6)	14	1.6 (3)	1	0.134
Poa spp.	3.4 (4)	30	2.2 (2)	33	0.511	7.8 (6)	79	2.1 (3)	15	0.052
Total grasses	67 (7)		68 (9)		0.913	47 (22)		52 (29)		0.799
Achillea millefolium	1.0 (1)	43	0.6 (1)	24	0.450	1.7 (2)	52	1.1 (1)	34	0.424
Artemisia frigida	16 (8)	59	11 (5)	41	0.232	4.5 (3)	60	3.0 (4)	29	0.463
Total forbs	32 (7)		32 (8)		0.961	23 (14)		21 (13)		0.695
Rosa arkansana	1.0 (1)	14	2.5 (3)	48	0.280	2.3 (2)	63	0.8 (1)	19	0.077
Symphoricarpos occidentalis	1.0 (2)	10	0.6 (1)	13	0.746	7.9 (8)	49	3.6 (5)	18	0.250
Total shrubs	2.0 (3)		3.4 (3)		0.397	10 (8)		4.4 (5)		0.131
Moss/lichen	3.3 (3)	15	15 (5)	82	**0.020**	1.9 (3)	21	3.3 (3)	46	0.438
Bare ground	14 (15)		1.5 (2)		0.064	26 (24)		2.3 (3)		**0.025**
Litter	14 (3)		16 (4)		0.370	19 (11)		43 (29)		0.068
Diversity	2.4 (0)		2.6 (0)		0.930	2.4 (0)		2.3 (0)		0.699
Richness	22 (6)		27 (4)		0.167	25 (8)		25 (5)		0.951

CHAPTER 5. *FESTUCA HALLII* RESPONSE TO ARBUSCULAR MYCORRHIZAE FUNGI

5.1 Introduction

Arbuscular mycorrhizal fungi (AMF) infect the roots of most terrestrial plants, providing their hosts with soil resources in exchange for photosynthate (Allen 1991; Smith and Read 2008). The interaction between plants and AMF has traditionally been regarded as a mutualism, in that both partners benefit from the association (Allen and Allen 1984; Allen 1991). AMF colonization may improve competition of late successional species over early colonizers, by assisting uptake of nutrients in infertile soils (Janos 1980; Allen and Allen 1984). Nevertheless, the costs and benefits of maintaining a symbiosis with AMF can differ significantly among plants, and resulting plant responses can vary widely (Johnson et al. 1997; Klironomos 2003).

Festuca hallii (Vasey) Piper, a late successional species, hosts various AMF species, including *Glomus fasciculatus* (Thaxter) Gerd. & Trappe, *Glomus macrocarpus* var. *macrocarpus* Tul. and Tul. and *Glomus scrobiculata* Trappe (Molina et al. 1978). *Festuca hallii* is the dominant species in rough fescue prairie grassland, originally stretching west from central Saskatchewan to the Rocky Mountain foothills in Alberta (Coupland 1961). A long-lived perennial, *F. hallii* will eventually dominate grassland to the exclusion of other species (Vujnovic et al. 2002), but can recover poorly if disturbed (Elsinger 2009; Desserud et al. 2010). The ability to restore fescue prairie, including *F. hallii*, has become an important consideration for oil and gas companies in Alberta, which are required by law to return disturbed land to equivalent land capability (Alberta Government 2000), and in native grassland to within 15% of prior disturbance species cover.

Oil and gas well site construction results in major disturbance to one hectare of land or more. Topsoil is stripped from the entire area, stored for the duration of well construction or longer, then replaced and re-seeded. Major soil disturbances usually reduce organic matter, including AMF populations and spores. AMF viability decreases the longer topsoil is stored (Rives et al. 1980; Gould and Liberta 1981; Liberta 1981).

No research assesses whether the association between *F. hallii* and AMF is obligate or beneficial for the plant. If *F. hallii* benefits from AMF, then topsoil stripping and storage with subsequent decline in AMF, could reduce its successful establishment. The objective of this research was to determine if arbuscular mycorrhizal fungi has a positive effect on *F. hallii* emergence and establishment.

5.2 Materials and Methods

5.2.1 Field Methods

To establish AMF inoculums from field soil, *F. hallii* plants were transplanted from native rough fescue grassland in the Rumsey Natural Area, in central Alberta. Ten mature *F. hallii* plants, with canopy diameters over 15 cm, were dug up with a large spade, soil cut in a circumference equal to the leaf canopy and to a depth of approximately 50 cm to obtain most of the root mass (Best et al. 1971; Aiken and Darbyshire 1990; Willms and Fraser 1992). Plants were immediately placed in pots, with their roots and surrounding soil, transported within six hours to a greenhouse and grown under greenhouse conditions for six months.

Ten three-year old *F. hallii* plants were collected from field sites in a different experiment, in which *F. hallii* was seeded on reclaimed natural gas well sites (Chapter 2). Plants were excavated with a hand shovel with cuts equivalent to the circumference of the leaf canopy made around the plant and to a depth of approximately 20 cm, to include most roots (Johnston 1961). These plants served as samples of AMF colonization in field conditions.

Festuca hallii seed was collected in July 2006, in rough fescue grassland in the Rumsey Natural Area, with a Wintersteiger Plot Master Elite combine, cutting at a height of approximately 30 cm. The combine was equipped with a straight cut header, a standard rasp bar cylinder, a 6 mm upper sieve and an adjustable lower sieve that was almost closed for native grasses. Seed was air dried at 25 °C, the chaff removed and *F. hallii* seed separated from other grasses and stored at 15 °C. Each year, prior to greenhouse experiments, 100 randomly selected seeds were distributed equally among 10 closed petri dishes on 1 mm thick germination paper, wet with distilled water and incubated in the dark at 20 °C (Romo et al. 1991). After 6 weeks 85-95% germination was achieved.

5.2.2 Growth Chamber Methods

To evaluate *F. hallii* with or without AMF, an experiment was established in growth chambers, using fungicide to kill AMF. The experiment was conducted twice, in 2008 and again in 2009, with a different fungicide each time, to evaluate any potential fungicide effects. Soil, including *F. hallii* roots, was extracted from pots containing the wild-harvested plants and utilized as an AMF inoculum (Elmes et al. 1983; Jarstfer and Sylvia 1993; Marler et al. 1999). Ten cm of sterile potting soil (40% peat moss, 40% Terra Green©, 10% vermiculate and 10% perlite) was placed in the bottom of the pot, followed by 5 cm of field inoculum soil and covered with 2 cm of sterile potting soil. The field inoculum soil was left intact to avail developing roots the maximum AMF infection potential. In 2008 and 2009, 20 and 18 pots, respectively, were seeded with five, healthy and viable-looking *F. hallii* wild-harvested seeds.

Pots were placed in a controlled growth chamber set at 16 hours of light at 20 °C and 8 hours of darkness at 15 °C, and watered with de-ionized water as needed. During the second experiment, a problem with automatic timers in the growth chamber resulted in lights being on for 24 hours and temperature set at 20 °C for a four week period. The result was greater plant leaf length, biomass and root biomass than the previous experiment, over the same time period.

5.2.3 Fungicide Methods

Fungicide was applied to half of each growth chamber seeded treatment to establish a non-mycorrhizal treatment in previously infected soil (Marler et al. 1999). In 2008, the fungicide Fenaminosulf (4-Dimethylaminobenzenediazo-sulfonic Acid Sodium Salt (Lesan)) was used. It was known to negatively affect *Glomus* spp. (Sylvia and Schenck 1983), the AMF genus most likely to colonize

F. hallii (Molina et al. 1978). Fenaminosulf, dissolved in de-ionized water, was applied at a rate of one gram per litre, 200 ml per pot. In 2009, Rovral® (Iprodione, 240 g/l, with 1,2-benzisothiazolin-3-one at 0.014% as a preservative), proven to reduce AMF infection (Gange et al. 1990; Ganade and Brown 1997) was applied. Rovral dissolved in de-ionized water, to maintain a pH between 5.5 and 7 as required for Rovral efficacy (Bayer Crop Science 2011) was applied at a rate of 3.5 g/l, 20 ml per pot (Moora and Zobel 1996). Each fungicide was applied once a week for three weeks prior to seeding, at seeding time and then at weeks 2, 3, 7 and 15 after seeding. De-ionized water, an amount equivalent to the volume of dissolved fungicide, was applied to non-fungicide treatments to maintain equal soil water additions. At week seven, plants were thinned to the two plants with the longest leaf length and most tillers. Plants were harvested at week 22, and leaf and root lengths were measured.

After harvest, soil from pots was composited, two pots per sample, and analyzed for chemical properties by a commercial laboratory. Soil available ammonium (NH_4^+-N) was analyzed using the 2.0*M* KCl procedure (1993b). Soil available nitrate (NO_3^-) was extracted using a dilute calcium chloride solution (1993a). Soil electrical conductivity (EC) and pH were measured in a 2:1 soil to water solution (1993b; 1993a). Available phosphate (PO_4^-) and potassium (K^+) were analyzed with a modified Kelowna extraction (Qian et al. 1994). Aerobic bacteria waere measured by the heterotrophic plate count, pour plate method and anaerobic bacteria by standard plate count method (Health Canada 2001).

5.2.4 Glucosamine Assay.

To assess AMF colonization of *F. hallii* roots, a glucosamine assay was conducted. The assay is based on the fact that chitin, a polymer of N-acetyl-β-D-glucosamine, is a component of the cell wall of the majority of fungi and is not found in plant cells (Bartnicki-Garcia 1968; Wessels 1993; Nilsson and Bjurman 1998). Glucosamine indicates the presence of chitin and may be used to determine AMF biomass (Vignon et al. 1986; Sylvia 1994; Ekblad and Nälsholm 1996; Appuhn et al. 2004; Appuhn and Joergensen 2006).

Roots, from the growth chamber and field specimens, were separated from the plant crown and washed thoroughly with distilled water to remove all soil, pebbles and non-root vegetative particles. Careful washing of roots is essential to eliminate external sources of chitin, such as non-mycorrhizal fungi, bacteria, arthropods and other soil invertebrates (Jarstfer and Miller 1985; Ekblad and Nälsholm 1996). Prior to processing, roots were stored at -18 °C for 3 weeks. Roots were oven dried at 96 °C for 48 hours (Nilsson and Bjurman 1998) and then ground with a mortar and pestle to < 0.5 mm particle size.

A modified version of glucosamine assay techniques was followed (Nilsson and Bjurman 1998; Braid and Line 1981; Hepper 1977). A maximum of 200 mg was selected from each ground root sample, placed into threaded glass test tubes and threads wrapped with three rounds of Teflon (polytetrafluoroethylene) tape. Five ml of 6 N HCl (first dilution) were added to samples weighing between 100 and 200 mg (large sample), and two ml were added to samples weighing less than 100 mg (small sample). Test tubes were tightly capped, the samples hydrolyzed at 96 °C in an oven for 48 hours and then cooled to ambient temperature. Five ml of de-ionized and distilled water were added to the large samples and two ml to the small samples (second dilution).

Carefully avoiding solid particles, two ml of diluted hydrolysate were taken from large samples and one ml from small samples and placed in clean test tubes. Samples were evaporated in a 50 °C water bath, assisted by air injection. Compressed air was gently introduced into the test tubes with a Pasteur pipette attached with plastic tubing to an air source. Evaporated precipitate was rehydrated with 5 ml of de-ionized and distilled water (third dilution), then one ml was extracted and placed in Teflon wrapped threaded test tubes. A glucosamine standard was prepared with 50 µg/ml glucosamine; one sample with 100% glucosamine, one with 100% distilled water and five with graduated dilutions. To one ml samples and glucosamine standards, 0.25 ml of 4% acetylacetone solution (4% volume acetylacetone in 1.25 N sodium carbonate) were added. Test tubes were tightly capped and bathed in a 100 °C water bath for one hour, then cooled to ambient temperature in a cool water bath. Two ml of ethanol were added to each sample, which were shaken with an agitator for five seconds to dissolve the precipitate, then, 0.25 ml of Ehrlich reagent (1.6 g of N-N-dimethyl-P-aminobenzaldehyde in 60 ml of 1:1 ethanol and concentrated HCl) were added and shaken with an agitator for five seconds (Nilsson and Bjurman 1998).

Colorimetric assay yields statistically distinct levels of infection, especially at high levels of infection where histological methods may be difficult to perform (Hepper 1977; Bethenfalvay et al. 1981). Readings of absorbance were taken at A530 nanometers with a Spectronic-2 spectrophotometer, using glucosamine S_0 standard to zero the spectrometer. Readings were compared to a standard curve made from the glucosamine standard and used to calculate the amount of glucosamine per dry gram of root, carefully incorporating all dilutions into the calculation. Total µg in sample = µg glucosamine per 1 ml sample x (third dilution ml) x (first dilution ml + second dilution ml) / hydrolysate extraction ml; µg glucosamine per g root = Total µg in sample / g root

5.2.5 Statistical Analyses

Shapiro-Wilk analyses showed the data to be normally distributed. Growth responses and soil variable differences between treatments were analyzed by Student t-tests. Relationships among growth responses, glucosamine and soil variables were assessed with Pearson product correlation and linear regression. Significant variables were ranked with stepwise multiple regression. Data analyses employed IBM® SPSS® Statistics (version 18, SPSS, Chicago IL) and MS Excel (version 2007, Microsoft, Redmond WA).

5.3 Results

Leaf length (P = 0.002), number of tillers (P = 0.007), above ground biomass (P = 0.037), root biomass (P = 0.049) and root to leaf length ratio (P = 0.009) were significantly greater in Fenaminosulf treatments than non-fungicide treatments (Figure 5.1 and Table 5.1). Root length (P = 0.035) and root to leaf length ratio (P = 0.030) were significantly greater in Rovral treatments than non-fungicide treatments (Table 5.1). No significant differences were found for Fenaminosulf and Rovral glucosamine levels between fungicide and non-fungicide treatments; however, glucosamine levels differed across treatments. Higher glucosamine levels correlated to lower above ground biomass ($R^2 = 0.46$, $P = 0.003$) and root biomass (transformed by natural logarithm) ($R^2 = 0.42$, $P = 0.005$) in Fenaminosulf treatments (Figure 5.2). A similar negative relationship was found between glucosamine concentrations and tiller density ($R^2 = 0.25$, $P = 0.048$) in Rovral treatments (Figure 5.2). Three-year old field specimens had 1.7 times more glucosamine than 22 week old greenhouse plants.

Aerobic bacterial colonization was significantly higher in the Fenaminosulf treatment than the control ($P = 0.050$) as was anaerobic bacteria, although the difference was not statistically significant (Table 5.1). In the Rovral treatment, aerobic bacteria were similar between treatments; however, anaerobic bacteria were significantly ($P < 0.001$) higher in the fungicide treatment than the control (Table 5.1). No correlation was found to any *F. hallii* growth parameters with aerobic or anaerobic bacteria.

In Fenaminosulf treatments, ammonium (NH_4^+) ($P = 0.011$), nitrate (NO_3^-) ($P = 0.003$) and phosphate (PO_4^-), pH ($P = 0.007$) were significantly lower than in non-fungicide treatments (Figure 5.3; Table 5.1). Longer leaf length correlated to lower ammonium ($R^2 = 0.77$, $P = 0.023$), nitrate ($R^2 = 0.67$, $P = 0.046$) and pH ($R^2 = 0.76$, $P = 0.023$). No similar correlations were found in Rovral treatments.

5.4 Discussion

These results support those of Molina et al. (1978) that *F. hallii* is colonized by AMF in the field. However, contrary to the hypotheses, *F. hallii* growth parameters increased in one of the fungicide treatments and with low AMF colonization. Many researchers found AMF assisted nutrient uptake (Allen et al. 1981; Sundaresan et al. 1988; Allen 1991; Allen 1993); although some concluded AMF can suppress plant growth relative to non-mycorrhizal plants, which is supported here (Koide 1985; Johnson et al. 1997; Klironomos 2003). Hays et al. (1982) found lower shoot length in 6 week-old *Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths that had been infected with AMF in an experiment with varying concentrations of nitrogen and phosphorous. Although *B. gracilis* responded to increased nutrients with increased shoot length, AMF infected plants had slower growth and less tillering than uninfected plants, confirming findings by Miller (1987).

Ammonium positively affected shoot and root growth of perennial cool-season grasses. For example, *Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould ssp. *lanceolatus* and *Lolium perenne* L. allocated more to shoot growth than root growth with additional ammonium (Jarvis 1987; Li and Redmann 1992). In this experiment, lower ammonium, nitrate and phosphate concentrations in Fenaminosulf treated soils, were probably due to plant uptake and assimilation, with corresponding greater leaf length and biomass. These results correspond to those of Smith et al. (1968), who found *Festuca campestris* Rydb., a species similar to *F. hallii*, yield (kg/ha) increased with nitrogen and phosphorous fertilizers. AMF assists host plants with nutrient incorporation, including phosphorous (Allen et al. 1981; Smith et al. 1986) and ammonium, which is often immobile and difficult for plant absorption (Smith 1980; Ames et al. 1983). These results were inconclusive regarding potential nutrient uptake assistance by AMF, since plants with greatest biomass and tiller densities also had the lowest AMF colonization.

Aerobic and anaerobic bacteria colonization were higher in both fungicide treatments than in non-fungicide treatments. Plants may associate with soil aerobic bacteria, for example *Festuca rubra* L. (Elo et al. 2006) and *Festuca ovina* L. (Lawley et al. 1982). Bacteria may contribute nitrogen to plants in return for amino-acid exudates (Lawley et al. 1982). In turn, plants may promote soil bacterial growth through organic material released into their rhizosphere (Griffiths et al. 1992). AMF colonization may positively affect soil bacteria, as Andrade et al. (1997) found AMF root colonization resulted in an increase of bacterial colony forming units in the hyphosphere and the presence of *Pseudomonas* spp. in the rhizosphere. This experiment may have confirmed this association, since lower bacterial colonization occurred where AMF infection was higher, possibly indicating hyphosphere and rhizosphere associations. Fewer bacteria would remain in soil once plants, with AMF infection, were removed from the experimental units with soil attached to roots.

The amount of associated bacteria varies with species. For example, Griffiths et al. (1992) found *Festuca arundinacea* Schreb. supported less bacteria than *Poa annua* L. and *Poa pratensis* L. In addition to nutrient cycling, soil microorganisms are involved in hormone production, plant pathogens, detoxification and other functions which influence plant growth (Ames et al. 1987). Fenaminosulf is toxic to several species of bacteria including *Rhizobium* sp. (Tu 1980) and *Xanthomonas campestris* (He et al. 2010) when applied to above ground leaf and fruit; however, no literature was found on its effect on soil bacteria. Kazempour and Elahinae (2007) found Rovral had no effect on bacteria, such as *Pseudomonas fluorescens* and *Bacillus cereus* (aerobic bacteria). No research was found on the interaction of Rovral and anaerobic bacteria. In this experiment, neither fungicide had effect on total soil bacterial colonies; however, their effect on individual species was not evaluated.

Unlike the Fenaminosulf treatment, the experiment with Rovral resulted in differences only in root growth. This did not allow duplication of the results; therefore, additional research would be required to confirm the reaction of *F. hallii* to AMF. Research using sterile soils or hydroponics with AMF inoculants could provide more exact results. Field experiments tracking *F. hallii* growth over several years, with and without AMF could provide insights into AMF effects over time; however, the difficulty of isolating environmental factors in the field would reduce the success of such experiments.

5.5 Conclusions and Management Considerations

This research was based on juvenile plants, when plant requirements for carbon to promote growth may outweigh the nutrient benefits AMF might afford. As

Johnson et al. (1997) suggest, while mycorrhizal associations are generally mutualistic, they may also be parasitic, when the stage of plant development makes costs greater than benefits. Carbon drain by AMF may have diminished young plant growth, in the slow-growing *F. hallii*.

These results are intriguing and may indicate reduced AMF in stored topsoil may not affect *F. hallii* recovery. Other factors must be in play to answer the puzzle of why *F. hallii* fails to return on disturbed sites.

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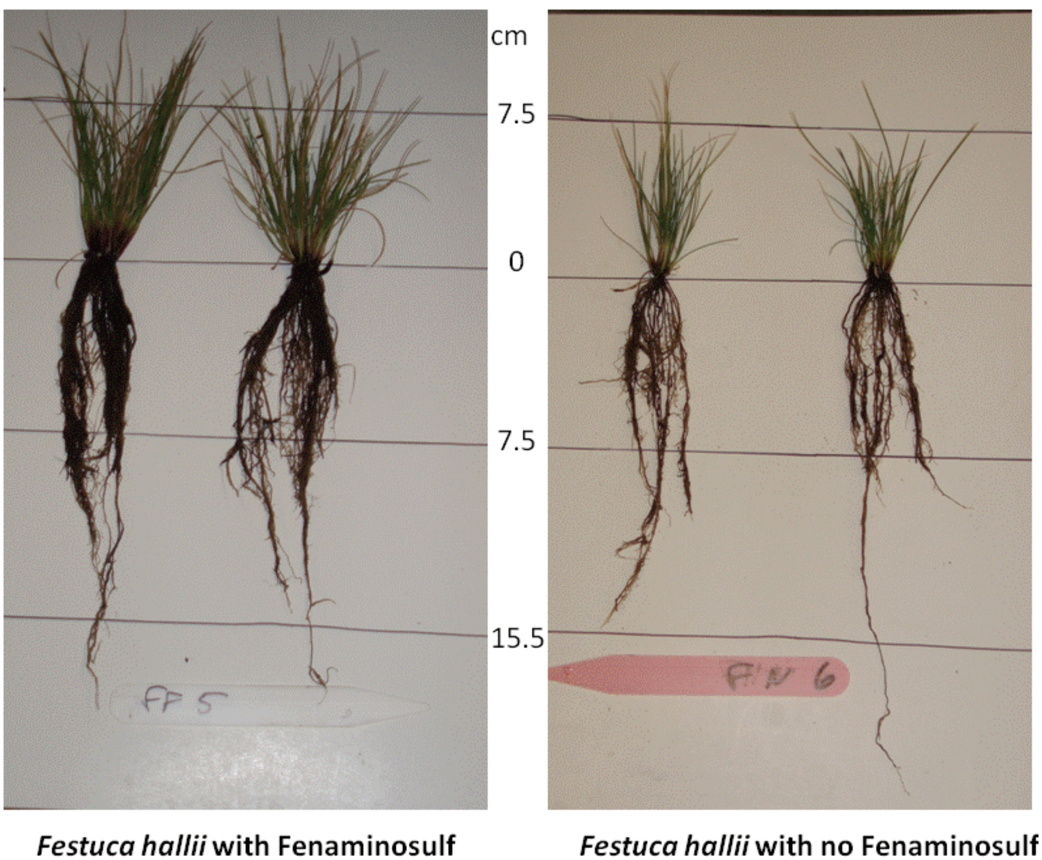

Figure 5.1 *Festuca hallii* plant samples showing differences in leaf length, tiller count and root length, between treatments with and without fungicide (Fenaminosulf).

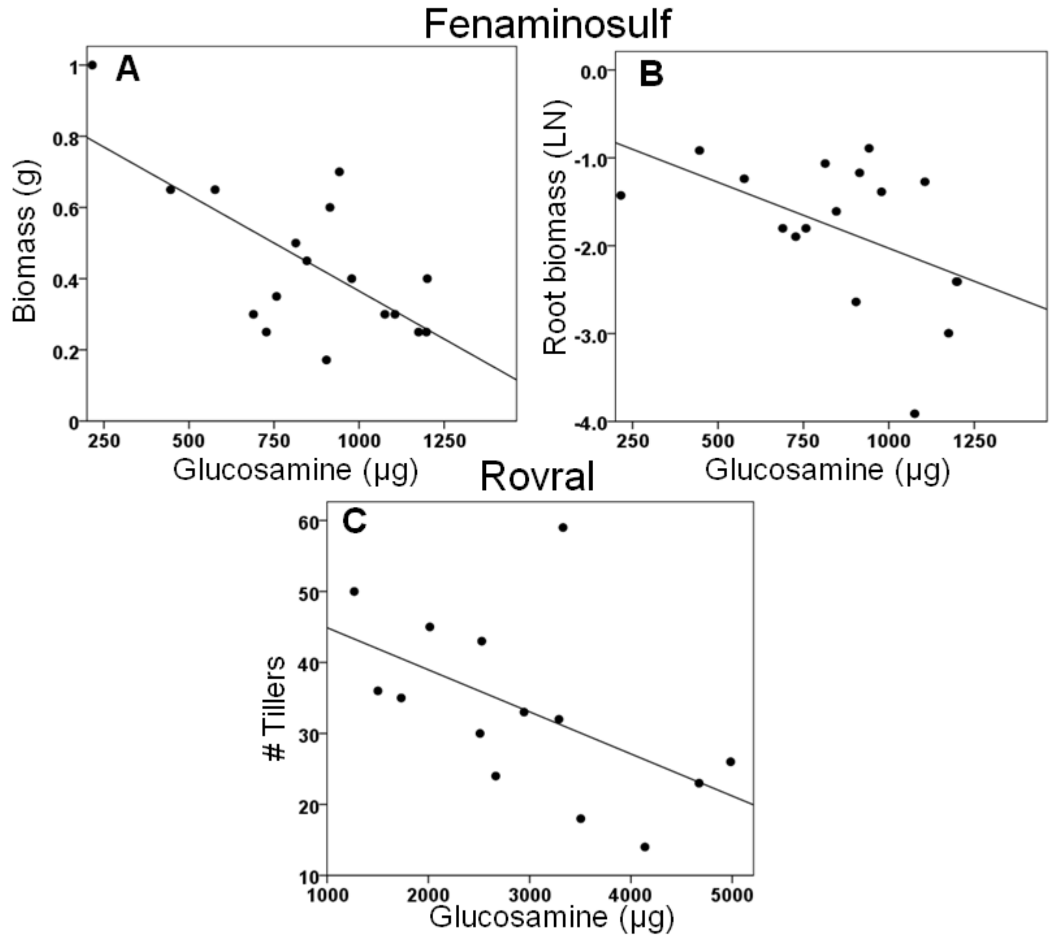

Figure 5.2 *Festuca hallii* (A) biomass (R^2 = 0.46, P = 0.003) and (B) root biomass (R^2 = 0.25, P = 0.042) correlations to glucosamine concentrations in plants treated with Fenaminosulf, and (C) tillers (R^2 = 0.25, P = 0.048) in plants treated with Rovral, showing a decrease in each property as glucosamine concentrations increase.

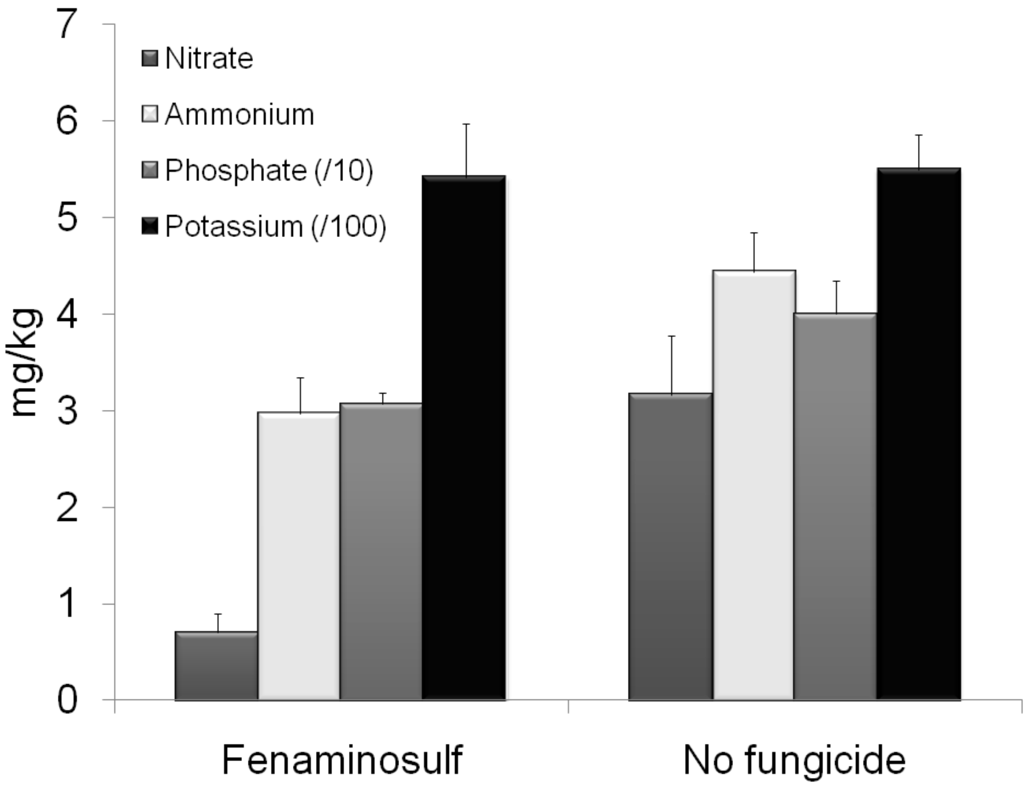

Figure 5.3 Soil properties of fungicide (Fenaminosulf) treated soils compared to non-treated soils showing differences in nitrate (P = 0.001), ammonium (P = 0.011), phosphate (P = 0.011) and no difference in potassium (P = 0.845).

Table 5.1 *Festuca hallii* vegetative and soil substrate properties, showing differences between fungicide, Fenaminosulf and Rovral and non-fungicide treatments (mean ± SD).

	Fenaminosulf	None	P	Rovral	None	P
			Vegetative properties			
Leaf length (cm)	13.8 (2)	10.0 (2)	**0.002**	17.0 (4)	16.9 (2)	0.969
Root length (cm)	18.6 (4)	21.1 (6)	0.331	34.4 (9)	26.4 (4)	**0.035**
Tillers	24.7 (7)	17.0 (5)	**0.007**	27.5 (13)	37.9 (11)	0.130
Biomass (g)	0.5 (0)	0.3 (0)	**0.037**	1.0 (1)	1.3 (1)	0.270
Root biomass (g)	0.2 (0)	0.1 (0)	**0.049**	0.4 (0)	0.5 (0)	0.345
Glucosamine (µg x 10^2)	7.78 (3)	11.8 (6)	0.090	28.8 (13)	29.2 (10)	0.959
			Soil properties			
Aerobic bacteria (x 10^5)	48.3 (25)	16.3 (3)	0.050	74.5 (43)	65.9 (1)	0.861
Anaerobic bacteria (x 10^5)	1.3 (0)	0.9 (0)	0.220	15.5 (2)	7.3 (2)	**<0.001**
Available ammonium (NH_4^+) (mg/kg)	3.0 (0)	4.4 (0)	**0.011**	18.9 (7)	14.9 (5)	0.380
Available nitrate (NO_3^-) (mg/kg)	0.7 (0)	3.2 (1)	**0.001**	17.4 (4)	19.8 (3)	0.390
Available phosphate (PO_4^-) (mg/kg)	30.7 (1)	40.0 (4)	**0.011**	49.3 (3)	48.3 (7)	0.798
Available potassium (K^+) (mg/kg)	541 (56)	549 (36)	0.845	720 (66)	587 (56)	**0.022**
Total carbon C (mg/kg)	5.5 (0)	5.3 (1)	0.729	13.0 (1)	11.7 (1)	0.142
pH	7.5 (0)	7.7 (1)	**0.007**	5.6 (0)	5.5 (0)	0.557
EC (ds/m)	1.1 (0)	0.8 (0)	**0.011**	0.3 (0)	0.4 (0)	0.280

CHAPTER 6. RESTORING GRASSLAND DISTURBANCES WITH NATIVE HAY

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6.1 Introduction

Native grassland is an important resource for range managers, providing self-sustaining, high quality forage (Holechek et al. 2004). In Alberta, much of the once dominant rough fescue grasslands have been lost to cultivation and overgrazing (Adams et al. 2005). Intensive oil and gas development adds pressure to disappearing native grasslands. Few attempts to restore rough fescue plant communities have been successful. Gas well sites and pipelines in central and southern Alberta had fair to poor establishment of rough fescue and other native species from seed mixes (Desserud 2006; Elsinger 2009). Best and Bork (2003) had mixed results transplanting rough fescue seedlings. Once the dominant grass in central Alberta, *Festuca hallii* (Vasey) Piper (plains rough fescue) is particularly difficult to restore once disturbed. This long-lived perennial bunch grass produces seed erratically with several years between seeding events (Johnston and MacDonald 1967; Romo 1996).

Experiments using native hay as a seed source to restore grasslands were successful in Germany (Kiehl et al. 2006) and England (Jones et al. 1995; Edwards et al. 2007), where native hay provided the requisite species to restore long cultivated land to ancient condition. In contrast, Bakker et al. (2003) found no native seedling emergence from native hay application in mixed grass prairie restoration in Saskatchewan. No research was found involving native hay for rough fescue grassland restoration.

The benefits of native hay include no cost for seeds, a natural mix of adapted native species, protective mulch for emerging seedlings and no requirement for special seed processing or seeding. Factors affecting seed viability in native hay include annual seed production variability for species that do not seed annually, timing of use resulting in dominance of species in seed at placement time and methods such as tackifying the hay in place (Romo and Lawrence 1990).

The objective of this research was to assess the potential of native hay as a seed source for restoring rough fescue grassland. Harvesting hay from rough fescue grassland during a year when plains rough fescue produces seed may result in a viable native seed mix. The hypothesis is that native hay will provide rough fescue grassland species for disturbance restoration and, that plains rough fescue will emerge if the hay is harvested in a seed producing year.

6.2 Methods

6.2.1 Study Area

Two natural gas pipeline rights of way (ROW) were studied between 2006 and 2008. One had native hay applied as a seed source, the other was seeded. The study area is located in Alberta, Canada, in uncultivated rangeland in the Central Parkland natural region. Winters are long, cold and dry, while summers are short and moderately warm. Temperatures range from -40 to +35 °C, with growing season (May to October) temperatures averaging 13 °C. Average annual precipitation is approximately 450 mm. Topography is undulating, a complex of small depressions and hills. The soils are loamy Dark Brown Chernozems on medium textured glacial till. Native vegetation on uplands and upper slopes is rough fescue grassland, dominated by *Festuca hallii*, *Hesperostipa curtiseta* (A.S. Hitchc.) Barkworth (western porcupine grass), *Pascopyrum smithii* (Rydb.) A. Löve (western wheat grass) and *Poa* spp. (various bluegrasses).

6.2.2 Experimental design

In August 2005, topsoil was removed from a 15 by 150 m pipeline ROW prior to pipeline installation, spread back following construction within one month and left un-seeded. In central Alberta, plains rough fescue flowered in 2006. Hay cutting occurred on July 16, 2006, prior to mid-summer seed shattering. Three days prior to hay cutting, 2, 4-Dichlorophenoxyacetic acid (2-4-D) was applied to the ROW at a rate of 90 kg/ha to remove weeds. A modified combine, with more durable and sharper than traditional crop blades, was used to cut the hay and immediately spray it on the pipeline ROW. Hay was cut at a height of approximately 30 cm, to obtain rough fescue seed, to avoid forb and potential weed seed and to leave substantial stubble for recovery. Approximately 67 m^3 of native hay was cut, sufficient to cover the ROW to a depth of 2 to 3 cm. The hay was raked to even out large clumps, and then crimped into the soil with shallow disc harrows. The ROW was fenced, in two sections separated by an access road, to prevent cattle grazing, although in 2009, one section was accidentally grazed.

In July 2007, topsoil was removed from a 3 x 150 m pipeline right-of-way (seeded ROW) prior to pipeline installation and spread back following construction. The ROW was lightly seeded with *Elymus* sp. (an annual rye), *Elymus trachycaulus* (Link) Gould ex Shinners ssp. *subsecundus* (Link) A.& D. Löve (slender wheat grass) and *Festuca saximontana* Rydb. (Rocky mountain fescue).

Each year, from 2007 to 2009, two 50 m transects, with five 20 by 50 cm subplots, 10 m apart, were randomly located in each native hay ROW section to assess foliar cover of all species, litter and bare ground. In 2007, vegetation, litter and bare ground in adjacent native grassland were assessed on a 100 m transect with ten 20 x 50 cm subplots, 10 m apart, approximately 15 m from the ROW, in the same area where the hay was cut.

Ten random hay samples were collected for a greenhouse experiment at the time of application to the native hay ROW. They were spread approximately 1 cm thick over 3 cm of potting soil (1:4 vermiculite and peat) in trays (10 x 15 x 5 cm) and watered with tap water when the surface began to dry, approximately every 2 days. Greenhouse conditions were set to simulate the summer growing season, 25 °C with 16 hours of light. Emerging seedlings were enumerated and removed once identified, for a three month period. Percent of each species was calculated based on total number of germinated species.

Vegetation, litter and bare ground were assessed, on the seeded ROW in 2008 and 2009 and in adjacent native grassland in 2007, with a 100 m transect and ten 20 x 50 cm subplots, 10 m apart. To assess the seed bank potential of the ROW, ten soil samples (15 x 15 x 6 cm) were taken from the newly reclaimed ROW, spread approximately 2 cm thick over potting soil in trays (10 x 15 x 5 cm), and, treated similarly to the hay samples previously described.

6.2.3 Statistical Analyses

Data were subjected to one-way analysis of variance (ANOVA) with Tukey's post-hoc test and independent sample T-tests for pair-wise comparisons at 1% level of significance ($P < 0.01$). Recovery results from the native hay ROW were compared over three years (2007 to 2009) and 2009 results from the seeded ROW were compared. Species diversity was calculated with the Shannon-Wiener diversity index. Data analyses employed IBM® SPSS® Statistics (version 18, SPSS, Chicago IL) and Microsoft® Excel© 2007(Redmond, Washington).

6.3 Results

6.3.1 Controls

The adjacent native grassland at both sites was dominated by *F. hallii*, *H. curtiseta*, *Koeleria macrantha* (Ledeb.) J.A. Schultes (June grass), *P. smithii*, *E. trachycaulus*, *Poa* spp., *Carex* spp. (sedge) and an abundance of forbs (Table 6.1 and Table 6.2). Dominant grass species in native hay were *E. trachycaulus*, *F. hallii*, *Poa* spp. and *P. smithii* (Table 6.1). The seed bank from the seeded ROW included plains rough fescue and bluegrasses (Table 6.1).

6.3.2 Native Hay ROW

In the first year (2007), *P. smithii* had the greatest cover on the native hay ROW, followed by *F. hallii*, *H. curtiseta* and *Poa* spp. (Table 6.2). *Festuca hallii* seedlings were approximately 3 cm in height. Forbs included *Achillea millefolium* L. (yarrow), *Artemisia ludoviciana* Nutt. ssp. *Ludoviciana* (prairie sage) and

Artemisia frigida Willd. (pasture sage) (Figure 6.1). In the second year (2008), *Poa* spp. were dominant and *P. smithii* decreased; *Nassella viridula* (Trin.) Barkworth (green needle grass) appeared and *A. frigida* increased (Table 6.2). *Festuca hallii* was still at the seedling stage, approximately 5 to 10 cm in height. By the third year (2009), *Poa* spp. continued to dominate and *N. viridula* increased (Table 6.2). *Festuca hallii* increased in height to over 10 cm.

Weed species, mainly *Descurainia sophia* (L.) Webb (flixweed) and *Hordeum jubatum* L. spp. *jubatum* (foxtail barley) found in year one, almost disappeared by year three (< 0.1% cover). Species found on the ROW which did not germinate in the native hay greenhouse experiment were *H. curtiseta*, *Koeleria macrantha*, *N. viridula*, *A. frigida* and *A. ludoviciana* (Table 6.2). Less than 1% litter cover was found on the ROW in years one and two; however, in year three it averaged over 70%, mainly comprised of *Poa* spp. residue. Less than 1% bare ground was found on the ROW in all three years. The control had higher species diversity (Shannon-Wiener index 2.5), while the native hay ROW decreased in diversity from year one to three (2.1, 1.6 and 1.5, respectively).

6.3.3 Seeded ROW

First year's (2008) growth on the seeded ROW was dominated by seeded *E. trachycaulus* and several weeds, such as *Chenopodium* spp. (lamb's quarters) and *Monolepus nuttalliana* (J. A. Schultes) Greene (narrow leaved goosefoot). Bare ground averaged 30% with no litter. In the second year (2009) non-native species disappeared, *Elymus* spp. decreased, *Poa* spp. increased, and, forbs, especially *A. frigida*, increased. Bare ground decreased and litter, mainly *Elymus* spp. and *E. trachycaulus* residue, increased (Table 6.1). Species diversity in the first year was Shannon-Wiener index 1.7, decreasing to 1.2 in year two. Adjacent grassland diversity was 1.8 (Table 6.1).

6.3.4 Native Hay and Seeded ROW Comparison

Comparing the native hay ROW second year growth to the adjacent grassland showed similarities in *F. hallii* (P = 0.011), *P. smithii* (P = 0.043), *E.*

trachycaulus (P = 0.032) and *Poa* spp. (P = 0.047) cover. The native hay ROW had fewer forbs (P < 0.001) and native species (P < 0.001) and more litter (P = 0.001).

In contrast to the adjacent grassland, the second year the seeded ROW had less *F. hallii* cover (P < 0.001), more *E. trachycaulus* (P = 0.040), fewer native species (P < 0.001), and, greater bare ground (P = 0.024) and litter (P = 0002). Similarities were found in *Poa* spp. (P = 0.056), *P. smithii* (P = 0.668) and total forb (P = 0.423) cover.

Evaluating second year growth of the native hay ROW and the seeded ROW showed similarities in *E. trachycaulus* (P = 0.207), total forbs (P = 0.833), native species (P = 0.198) and litter (P = 0.283). The native hay ROW had greater cover of *P. smithii* (P = 0.018) and *Poa* spp. (P < 0.001). Less bare ground occurred on the native hay ROW although the difference was not significant (P = 0.234). The seeded ROW had no *F. hallii* while the native hay ROW had 12% cover (Figure 6.2).

6.4 Discussion

This experiment supports the hypothesis that native hay cut from rough fescue grassland is a viable seed source for restoring disturbances. All species that emerged on the native hay ROW were found in undisturbed grassland. These results were consistent with those from European and English grassland restoration experiments. These concluded native hay resulted in similar species to the donor site (Edwards et al. 2007), greater species diversity, more native species (Jones et al. 1995) and a return of agricultural lands to ancient grassland conditions (Kiehl et al. 2006).

Of particular note in this experiment was the emergence of *F. hallii* seedlings in the first year, and its continued growth over the following two years. This is a promising result given the failure of rough fescue establishment, even when seeded, on other oil and gas disturbances in the area (Desserud 2006; Elsinger 2009). In contrast, the seeded ROW was dominated by seeded species in the first and second year. Despite the occurrence of *F. hallii* in the seed bank of the seeded ROW, only a small amount of *F. hallii* appeared in the first year, possibly remnant plants from the initial topsoil stripping, and none appeared in the second year.

An advantage of applied hay is increased ground cover, which likely accounted for reduced non-native species on ROW, similar to what occurred in the Jones et al. (1995) experiment. The low amount of bare ground, commencing the first year, is in direct contrast to the seeded ROW and what Elsinger (2009) and Desserud (2006) found on seeded ROWs 30 years after recovery.

Seasonal timing of hay cutting is important in determining which seeds will be available and viable. Since this experiment targeted *F. hallii*, the hay was cut when its seeds were ripe. To obtain a full suite of native grassland species, Edwards et al. (2007) recommended cutting hay several times, such as early, mid and late summer. Kiehl et al. (2006) had success baling hay from a donor site and transporting it; however, further research into the longevity of native hay bales is needed. Being able to store native hay for future use would be important for well site restoration, which may take place several years after construction, or for retaining species, such as rough fescue, which do not produce seeds every year.

6.5 Conclusions and Management Considerations

This experiment showed that native hay has potential to provide a diverse plant cover similar to pre-disturbance grassland conditions with early species establishment. Since only one native hay site was available for study, extrapolation of the results to other sites is not strong. However, results are promising and warrant further study to evaluate timing of hay harvesting, how native hay responds to storage and optimal coverage.

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Table 6.1 Comparison of native hay and seeded ROW, showing years 1 and 2, native grassland control, seed from native hay and seed bank. Different letters indicate significant differences among values.

	Native Hay ROW					Seeded ROW				
	Year 1	Year 2	Control	Hay Seed	*P*	Year 1	Year 2	Control	Seed Bank	*P*
Elymus lanceolatus	0	0	0.3 (1)	1.8 (4)	0.553	0.3 (1)	0.5 (1)	0	0	**<0.001**
Elymus trachycaulus	1.3a (3)	2.0a (6)	5.5a (7)	37b (33)	**<0.001**	37a (19)	6.8b (10)	0c	0c	**<0.001**
Festuca hallii	10 (11)	12 (23)	34 (26)	24 (26)	0.076	0.2a (1)	0a	54b (22)	8.3 (2)a	**<0.001**
Koeleria macrantha	0.5 (2)	0	3.3 (6)	0.8 (2)	0.073	0.3a (1)	0.1a (0)	15b (20)	0a	**0.029**
Hesperostipa curtiseta	5.5a (9)	0b	6.5a (8)	0.5b (1)	**0.003**	2.5 (6)	0	1.5 (5)	0	0.654
Pascopyron smithii	14 (9)a	8.1a (7)	7.6a (8)	0.3b (1)	**<0.001**	0.8 (2)	1.8 (2)	1.3 (3)	0	0.749
Poa spp.	4.5 (6)a	40b (13)	6.3a (9)	4.2a (4)	**<0.001**	1.6 (2)	3.2 (5)	0.2 (1)	8.3 (2)	0.098
Achillea millefolium	2.0 (3)	9.7 (2)	3.9 (4)	4.7 (5)	0.073	0	2.0 (0)	0.3 (1)	0	0.310
Artemisia frigida	1.9 (5)	0.5 (2)	0	0	0.109	2.1 (5)	3.8 (7)	2.1 (6)	17 (3)	0.537
Artemisia ludoviciana	1.8 (5)	3.5 (9)	3.6 (4)	0	0.119	0.6 (2)	1.0 (3)	1.2 (2)	0	0.924
Symphoricarpos occidentalis	0a	0a	6.0b (8)	0a	**<0.001**	2.5 (6)	1.2 (2)	5.1 (9)	0	0.581
Bare ground	0a	10b (13)	0.5a (2)	n/a	**<0.001**	30a (7)	18b (14)	4c (10)	n/a	**<0.001**
Litter	0a	42b (24)	27c (18)	n/a	**<0.001**	0a	55b (21)	25c (14)	n/a	**<0.001**

Table 6.2 Native hay ROW showing years 1 to 3, native grassland control, seed from native hay. Different letters indicate significant differences among values.

	Year 1	Year 2	Year 3	Control	Hay Seed	*P*
Carex spp.	0a	0a	0a	11b (7)	0a	**<0.001**
Elymus lanceolatus	0	0	0	0.3 (1)	1.8 (4)	0.499
Elymus trachycaulus	1.3a (3)	2.0a (6)	0a	5.5a (7)	37b (33)	**<0.001**
Festuca hallii	10ab (11)	12ab (23)	11b (5)	34a (26)	24a (26)	**0.002**
Hesperostipa curtiseta	5.5a (9)	0b	1.4b (5)	6.5a (8)	0.5b (1)	**0.005**
Nasella viridula	11a (11)	5.5a (10)	13a (22)	0b	0b	< **0.001**
Koeleria macrantha	0.5ab (2)	0a	0a	3.3b (6)	0.8ab (2)	**0.031**
Pascopyron smithii	14a (9)	8.1a (7)	1.3b (1)	7.6a (8)	0.3b (1)	**<0.001**
Poa spp.	4.5a (6)	40b (13)	22c (25)	6.3ac (9)	4.2a (4)	**<0.001**
Achillea millefolium	2.0ab (3)	9.7b (2)	0a	3.9b (4)	4.7b (5)	**0.002**
Artemisia frigida	1.9 (5)	0.5(2)	1 (3)	0	0	0.203
Artemisia ludoviciana	1.8 (5)	3.5(9)	1.8 (3)	3.6 (4)	0	0.138
Galium boreale	0a	0a	0a	5.7b (8)	0a	**<0.001**
Descurainia sophia	1.6a (3)	0b	0b	0b	0b	**0.012**
Hordeum jubatum	5(16)	0	0	0	0	**<0.001**
Symphoricarpos occidentalis	0a	0a	0a	6.0b (8)	0a	**<0.001**
Total native species	37ab (20)	24a (27)	36a (30)	99c (9)	73b(66)	< **0.001**
Bare ground	0a	10b (13)	0.1a (0)	0.5a (2)	n/a	**<0.001**
Litter	0a	42bc (24)	65c (32)	27b (18)	n/a	**<0.001**
Species Diversity	1.7	1.6	1.5	2.4	1.4	

Hay applied **One year later**

Figure 6.1 Native hay ROW showing hay application (2006) and vegetation the following year (2007).

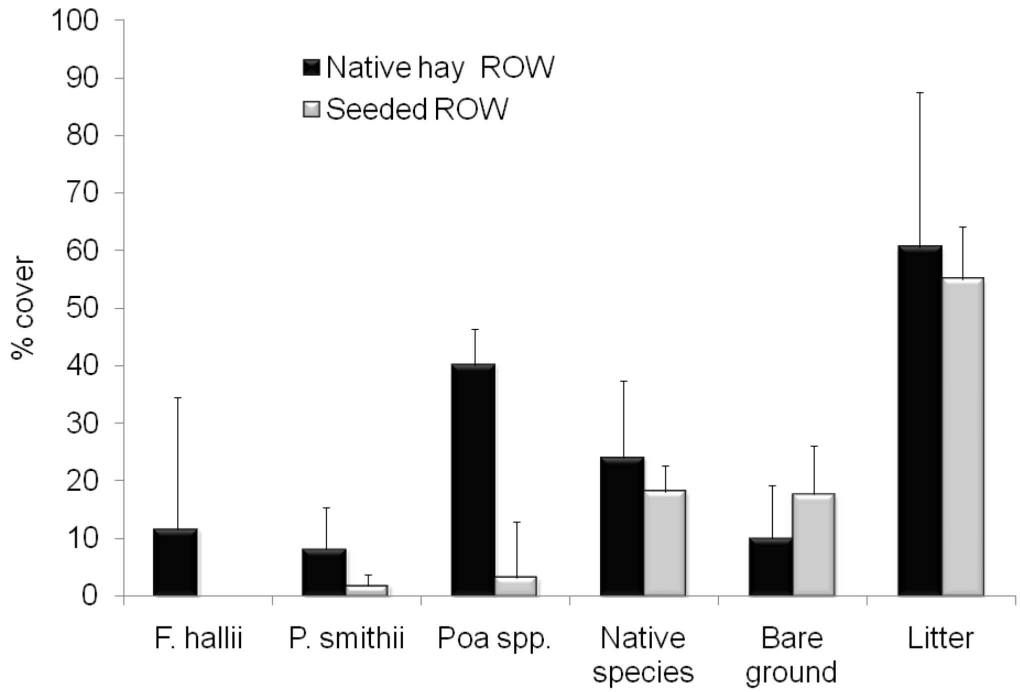

Figure 6.2 Comparison of native hay and Seeded ROWs showing differences in *Festuca hallii* ($P < 0.001$), *Pascopyron smithii* ($P = 0.018$), *Poa* spp. ($P < 0.001$), and similarities in total native species ($P = 0.198$), bare ground ($P = 0.234$) and litter ($P = 0.238$).

CHAPTER 7. SMOOTH BROME RESPONSE TO STRAW-AMENDED SOIL

A version of this chapter was published in the Journal of Ecological Restoration: Desserud, P.A. and Naeth, M.A. 2010. Smooth brome: an unexpected response to straw-amended soil (Alberta). Ecological Restoration 28: 133-135.

7.1 Introduction

Bromus inermis Leyss. ssp. *inermis*, introduced from Europe and Eurasia in the late 1880's (Dwinelle 1884; Elliotte 1948), is one of the most widely planted forage grasses in western North America (Bittman and Simpson 1987; Hardy BBT Limited 1989; Lamond et al. 1992). It is a prolific seed setter, rhizomatous, grows faster than many native grasses and is a long-lived perennial (Romo et al. 1990). As a result, it has successfully invaded prairie ecosystems across the Great Plains and western Canada (Slogan 1997; Sieg et al. 1999; Cully et al. 2003; Elsinger 2009). The original habitat of *B. inermis* in Eurasia is similar to fescue prairie (Looman 1976) making rough fescue grassland particularly susceptible to *B. inermis* invasion (Grilz and Romo 1995). Once established, *B. inermis* spreads rapidly, suppressing the growth and abundance of native flora, reducing wildlife habitat and natural diversity transforming diverse plant communities into virtual monocultures (Grilz et al. 1994; Otfinowski et al. 2007). Numerous control measures including burning, mowing and herbicide application may reduce the abundance of *B. inermis*, but without sustained efforts, it is remarkably persistent (Wilson and Gerry 1995; Willson and Stubbendieck 1996; Brown 1997).

The ability of many native species to out compete introduced species in nitrogen impoverished soil (Morgan 1994; Wilson and Gerry 1995) may provide a potential reclamation path in disturbed grasslands. *Bromus inermis* and other non-natives such as *Poa pratensis* L. require abundant plant-available nitrogen (Zemenchik and Albrecht 2002) and respond well to increased nitrogen. Several researchers have had varying success with improving the competitive ability of native grasses by depleting available nitrogen by applying combinations of sugar, sawdust and straw to soil (Wilson and Gerry 1995; Davis and Wilson 1997; Reever-Morghan and Seastedt 1999). It is possible that *B. inermis* invasions might be controlled with straw amendments to reduce nitrogen when reclaiming disturbances in rough fescue grassland.

7.2 Methods

The research site is the Byemoor site described in Chapter 2, a 72 x 60 m natural gas well site, never drilled, from which topsoil was removed and stored on site, in 2006. Straw was applied to the site as described in Chapter 2. On July 10, 2007, monocultures of *P. pratensis*, *Festuca hallii* (Vasey) Piper and a mix of native grasses, were seeded in strips at right angles to the straw treatments. A monoculture of *B. inermis* was unknowingly seeded on two of the native mix strips. In July 2008, the *B. inermis* was discovered and a reaction to the straw treatment was noticed (Figure 7.1); therefore, vegetation and soil samples were collected. Five *B. inermis* plants from each treatment were extracted, including roots, and adjacent soil samples were taken to 15 cm, the depth of first-year root growth. Roots and leaves were separated and roots thoroughly washed. Roots and leaves were measured for length, dried both at 96 °C for 48 hours and dry biomass weighed. Soil was analyzed for carbon, nitrogen and other properties (Table 7.1). A greenhouse experiment was established with *B. inermis* seeded in 5 pots, duplicating each of the straw treatments, and allowed to grow for 20 weeks at ambient light and temperature. The data were analyzed with one-way ANOVA using IBM® SPSS® Statistics (version 18, SPSS, Chicago IL).

7.3 Results

Bromus inermis growth varied considerably among the three straw treatments. Leaf length and leaf biomass in the field were significantly different with the largest *B. inermis* occurring where there was no straw (Table 7.1; Figure 7.2). Soil characteristics also varied among the treatments, especially ammonium and potassium, where higher levels of both chemicals were found associated with high straw applications (Table 7.1). Similar results were found in the greenhouse experiment, especially between the high straw treatment and the control (Table 7.1; Figure 7.3). *Bromus inermis* leaf length correlated negatively to ammonium (NH_4^+) and potassium (K^+) (Figure 7.4).

7.4 Discussion

Complete decomposition and soil nitrogen immobilization of the straw in the first year was not expected, as it takes up to 2 years to decompose in temperate climates (Parker et al. 1987). In fact, higher levels of ammonium in the straw amended soil in the early stages of decomposition is expected as denitrifying bacteria become more abundant where there is a plentiful fuel source such as straw (Parker et al. 1987). The high levels of potassium are also expected in the first year since potassium leaches readily out of straw, especially under moist conditions (Watts and Sirois 2003). These results elicited the question of whether high levels of ammonium and/or potassium were responsible for the poor response of smooth brome in the high straw treatment.

Little research was found to substantiate findings regarding *B. inermis* response to high levels of ammonium. Most research focuses on the optimum amount of ammonium to promote the growth of *B. inermis* as a forage crop, although Petersen and Moser (1985) noted its growth levelled off or declined with higher levels of nitrogen fertilizer, e.g., between 250 and 350 kg/ha of ammonium nitrate ($NH_4^+NO_3^-$). Similar research was discovered into potassium; for example, recommended potassium levels for *B. inermis* are between 100 and 160 ppm (Lamond et al. 1992). Potassium uptake by *B. inermis* is facilitated by nitrogen (NH_4^+-N) fertilizer (Barta 1975) and combination of nitrogen and potassium result in increased *B. inermis* above- and below-ground yields (Rabotnov 1977). At North Dakota prairie, Blankespoor and May (1996) discovered *B. inermis* growing better at sites with lower potassium (299 ppm) than at sites with higher potassium (399 ppm). Their potassium values are in line with this experiment, i.e., 274 mg/kg where *B. inermis* grew better and 403 mg/kg where it did not. They discounted the significance of the potassium, stating the potassium concentrations were probably not limiting. Leonard (1985) explains that as the potassium concentration in the outside medium increases, the rate of potassium absorption into root cells increases proportionately, but only to a certain point. This is due to “saturation kinetics”, caused by the limited number of K+ binding sites for potassium-carrier enzymes on the plant cell plasma membrane (Leonard 1985).

Another possibility is an allelopathic response of *B. inermis* to the wheat straw, although this is not likely. Hicks et al. (1989) discovered that wheat straw had an allelopathic effect on cotton. Machado (2007) found allelopathic response of *Bromus tectorum* to extracts of broad-leaved plants, and little response to grasses extracts, e.g. barley. Recommendations for growing *B. inermis* as a forage crop include using wheat as a cover crop during initial brome seeding, precluding a negative reaction by *B. inermis* to wheat straw (Lamond et al. 1992).

This experiment was terminated with a glyphosate application after the initial sampling to prevent aggressive *B. inermis* from invading the rest of the research site. Since *B. inermis* was in its first year, had not yet established a strong rhizomatous root network and had not yet produced seed, a single application of glyphosate succeeded in eradicating it.

7.5 Conclusions and Management Considerations

These results indicate a negative relationship between decomposition of wheat straw and early growth of smooth brome. While applying wheat straw to a new grassland disturbance may provide some protection against smooth brome invasion, more research is required to determine mechanisms involved, amount of straw needed and whether the effect continues beyond first year. Research into tolerance limits of smooth brome to potassium and ammonium would abe helpful.

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Table 7.1 Mean (± SD) *Bromus inermis* measurements and soil chemical properties as affected by straw amendments. In both greenhouse and field experiments, leaf length and biomass were highest with no straw, while ammonium and potassium concentrations were lowest. Different letters indicate significant differences among treatments.

	Field				Greenhouse			
	Straw amendment level				Straw amendment level			
	High	Low	None	*P*	High	Low	None	*P*
Leaf length (cm)	26.6a (3)	31.1a (3)	41.9b (4)	< **0.001**	22.0a (2)	34.0b (24)	39.6b (9)	**0.016**
Leaf biomass (g)	0.9a (0)	1.2a (0)	5.7b (2)	< **0.001**	0.3a (0)	0.8b (1)	0.9b (0)	**0.032**
Ammonium (NH_4^+) (mg/kg)	7.6a (1)	5.6ab (1)	3.9b (2)	< **0.001**	4.9a (1)	4.0ab (2)	2.9b (0)	**0.049**
Nitrate (NO_3^-) (mg/kg)	9.3a (4)	8.5a (3)	9.4a (3)	0.914	2.3a (1)	1.4a (0)	1.0a (0)	0.111
Potassium (K^+) (mg/kg)	403a (36)	348ab (56)	274b (19)	**0.001**	436a (26)	398b (16)	364c (13)	**0.001**
Phosphate (PO_4^-) (mg/kg)	17a (3)	16a (3)	13a (4)	0.158	27a (3)	21.8b (3)	16c (2)	< **0.001**
Carbon (C) (%)	3.4a (01)	3.4a (1)	3.1a (1)	0.757	5.1a (0)	5.0a (1)	4.9a (0)	0.797
Nitrogen (N) (%)	0.3a (0)	0.3a (0)	0.3a (0)	0.894	0.3a (0)	0.3a (0)	0.3a (0)	0.375
pH	8.2a (0)	8.4a (0)	8.3a (0)	0.464	7.9a (0)	8.0b (0)	7.7c (0)	**0.001**

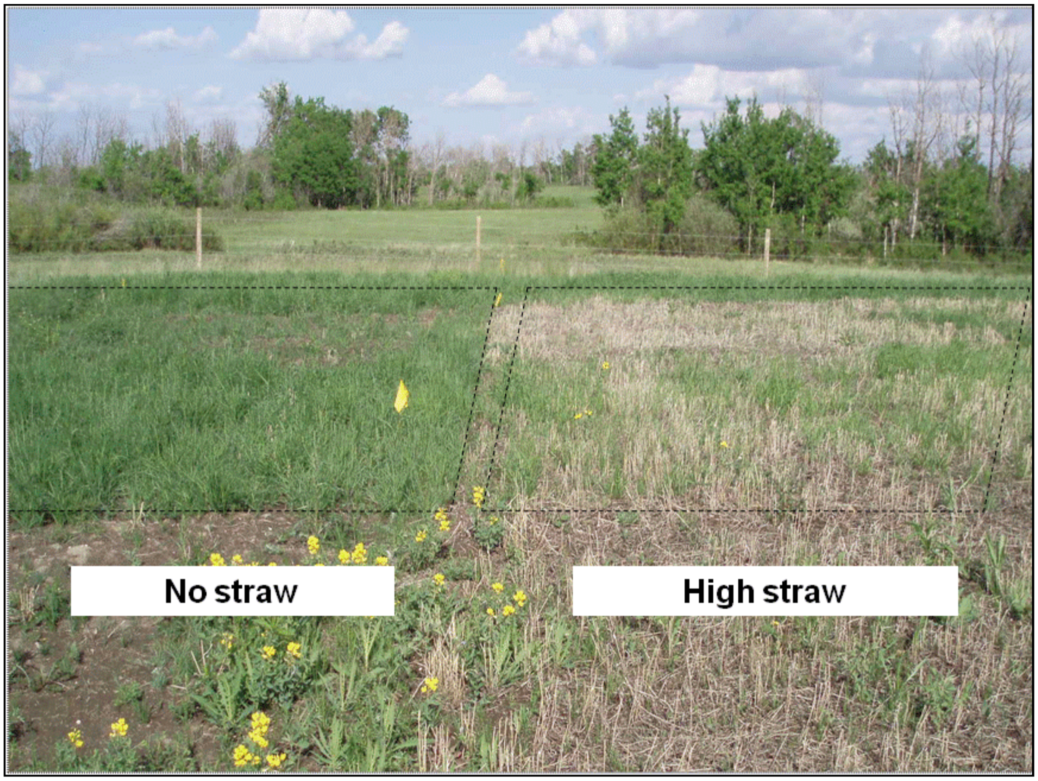

Figure 7.1 *Bromus inermis* plots at the Byemoor site (2008) showing difference between no straw and high straw treatments.

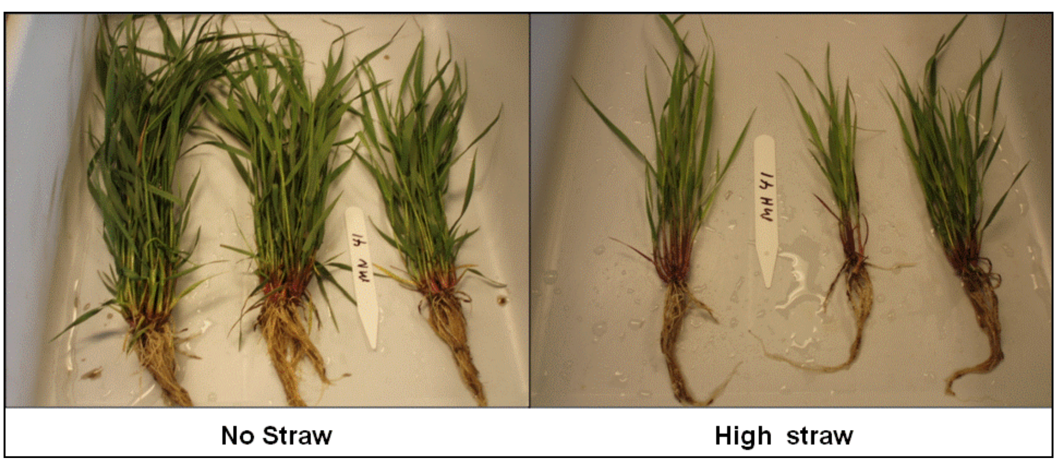

Figure 7.2 *Bromus inermis* samples showing differences in leaf and root length and biomass between no straw and high straw treatments.

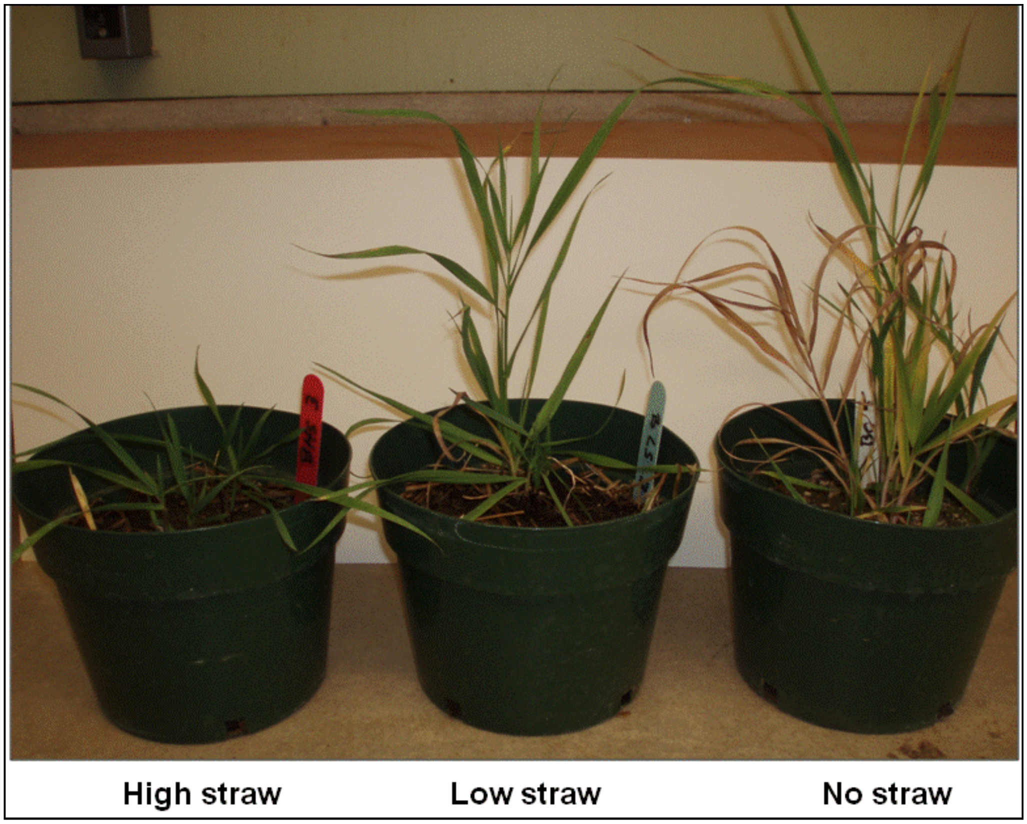

Figure 7.3 *Bromus inermis* greenhouse results, showing differences among high, low and no straw treatments.

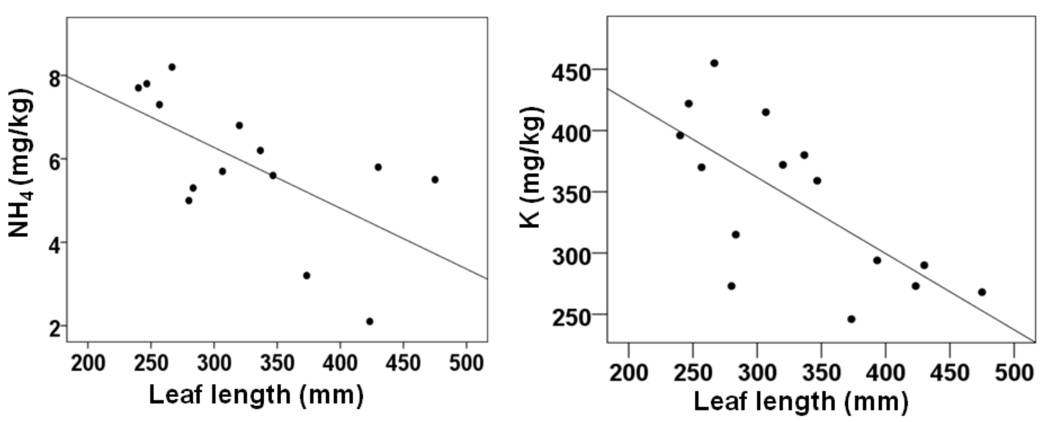

Figure 7.4 Correlation of *Bromus inermis* leaf length to ammonium (NH_4^+; R^2 = 0.40, P = 0.015) and potassium (K; R^2 = 0.48, P = 0.004).

CHAPTER 8. PREDICTING GRASSLAND RECOVERY WITH A STATE AND TRANSITION MODEL

8.1 Introduction

Rough fescue grasslands once extended throughout western and central Canada, and north western and central United States (Pavlick and Looman 1984). *Festuca hallii* (Vasey) Piper (plains rough fescue) was the dominant species in grasslands from central Alberta to Western Ontario, Montana and North Dakota, at elevations below 800 m (Pavlick and Looman 1984). These grasslands have been reduced to remnants, a result of urban and rural development, cultivation, livestock over-grazing, and oil and gas development. One of the largest remaining tracts of *F. hallii*-dominated grassland is located in central Alberta, Canada; 183 km^2 in a provincial protected area known as Rumsey Natural Area and Rumsey Ecological Reserve (Rumsey Block).

Ranching commenced in the Rumsey Block around 1895 and continues to this day. While protected from rural development, oil and gas exploitation is allowed in the Rumsey Block. Oil and gas pipelines and well sites were built with a variety of soil handling techniques including full right-of-way (ROW) stripping, whereby topsoil and subsoil were stripped off a 15 m ROW, stored and replaced following construction; and bucket-width (25 cm) stripping with topsoil salvage with either seeding or natural recovery. Until the 1970s, pipelines or well sites received little reclamation. Early reclamation seed mixes were predominantly introduced species, such as *Bromus inermis* Leyss. ssp. *inermis* (smooth brome) or *Poa pratensis* L. (Kentucky bluegrass). Later seed mixes used native grass cultivars, principally wheat grasses, such as *Elymus trachycaulus* (Link) Gould ex Shinners ssp. *subsecundus* (Link) A.& D. Löve (slender wheat grass), *Pascopyron smithii* (Rydb.) A. Löve (western wheat grass), *Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould ssp. (northern wheat grass). More recent reclamation either involved a varied native grass seed mix or natural recovery.

Festuca hallii, a late-seral and long-lived perennial bunch-grass, is difficult to restore once disturbed, reduced by moderate grazing, and possibly eliminated by heavy grazing (Looman 1969; Sinton 1980; Looman 1983). Wheat grasses occur naturally in rough fescue grassland and their cultivars are used in reclamation seed mixes. Wheat grasses are strong competitors and when seeded may persist and become dominant (Hammermeister 2001; Osterman 2001). Sedges, common in rough fescue grassland, are affected by disturbance, and after disappearing and require many years to recover (Inouye et al. 1987; Wang et al. 2006). Annual forbs appear in early succession following disturbance (Samuel and Hart 1994; Nasen 2009) and increase following heavy grazing (Dormaar and Willms 1990). *Bromus inermis*, an important agronomic forage crop introduced from Europe and Eurasia in the late 1880s, is a long-lived perennial, a prolific seed setter, rhizomatous, and grows faster than many native grasses (Romo et al. 1990; Blankespoor and May 1996). It was once commonly used in reclamation seed mixes, especially in livestock grazing areas, such as the Rumsey Block. Another forage species, introduced from Eurasia in the 1930s, and used in early reclamation seed mixes, is *Agropyron cristatum* (L.) Gaertn. ssp. *pectinatum* (Bieb.) Tzvelev (crested wheat grass), a perennial bunch-grass, with several characteristics attributing to its success, including a large number of tillers and prolific seed and litter production (Henderson 2005). *Poa pratensis* L. (Kentucky bluegrass), commonly included in reclamation seed mixes is considered introduced in Alberta, possibly arriving with European settlement (Tannas 2001). It is now endemic and often establishes from the seed bank when soil is disturbed (Brown 1997; Bizecki Robson et al. 2004; Adams et al. 2005). *Poa pratensis* readily produces seed and is strongly rhizomatous, allowing it to establish and spread rapidly.

Westoby et al. (1989) were the first to propose state and transition models to aid rangeland management. They defined "state" as a persistent vegetation community, that is not simply reversible, and with structural attributes and a characteristic range of variability. Transitions are trajectories of change precipitated by natural or human-made processes resulting in an alternative state (Stringham et al. 2003). Stringham et al. (2003) and Briske et al. (2006; 2008) added thresholds to the model, boundaries between states along irreversible transitions, such that the alternate state must be actively restored before return to a previous state is possible.

Although documented in non-peer reviewed reports, the full impact of oil and gas activity in the Rumsey Block has not been analyzed. A state and transition model was developed, compiling oil and gas disturbances and displaying trends over time and potential consequences of disturbances and grazing on rough fescue grassland in the Rumsey Block. The model may be used as a management tool, allowing land managers to predict the outcome of new disturbances or disturbance renovations in rough fescue grassland. The ability to predict changes from new disturbances will allow land managers to make better decisions when confronted by new development.

8.2 Methods

8.2.1 Study Area

The study area was located in central Alberta, Canada (51° 47′ to 51° 52′ N and 112° 25′ to 112° 42′ E) (Figure 1). Temperatures range from -40 to +35 °C, with growing season (May to October) temperatures averaging 13 °C. Average annual rainfall is approximately 350 mm and snowfall 100 cm. Topography is undulating, with a complex of small depressions (sloughs) and hills, 857 to 884 m above sea level. Upland soils are Black and Dark Brown Chernozems. Native vegetation on uplands and upper slopes is rough fescue grassland, dominated by *F. hallii* and including *Hesperostipa curtiseta* (A.S. Hitchc.) Barkworth (western porcupine grass), *Koeleria macrantha* (Ledeb.) J.A. Schultes (June grass), *Symphoricarpos occidentalis* Hook (western snowberry), *Carex* spp. (sedges), *E. trachycaulus*, *P. smithii*, *E. lanceolatus* and *Poa* spp. (bluegrasses). Non-native species occurring on disturbed sites include *Cirsium arvense* (L.) Scop. (Canada thistle) and *Taraxacum officinale* G.H. Weber ex Wiggers (dandelion). Oil and gas activity has resulted in 126 well sites, with accompanying pipelines (approximately 5 cm diameter); 37 connecting pipelines (between 5 and 15 cm diameter); and a single large diameter (91 cm) pipeline carrying ethane, running from northwest to southeast across the Rumsey Block.

8.2.2 Data Analyses

A meta-analysis of Rumsey Block vegetation data was conducted by compiling historical grassland vegetation assessments by Wroe (1971) (5 reference plant communities) and Eastern Slopes Rangeland Seeds Ltd. (1994; 1995) (29 disturbed sites and 10 reference plant communities), with recent vegetation assessments conducted by Elsinger (2009) (55 disturbed and reference sites), and Desserud and Naeth (2011a) (6 disturbed and reference sites). This resulted in a total of 90 disturbed sites, 76 reference sites, and 189 species. Pipelines and well sites were not distinguished; treating all as disturbances. All sampling was transect-based; therefore, the amount of disturbance described is similar regardless of type of disturbance. If sampling was extrapolated to the entire disturbance, on average, the disturbance of a 1 ha well site would be equivalent to that of a 1 km pipeline 10 m wide.

Alberta Sustainable Development (2009) range health and grazing data were matched with disturbed sites, and if located within 100 m were used as current state grazing regimes. Construction technique and reclamation seeding methods were identified for each disturbed site based on Elsinger (2009) and Desserud and Naeth (2011a) findings. For Eastern Slopes Rangeland Seeds Ltd. (1994; 1995) sites, methods were assigned to their data based on techniques known at the time. Seeding for Eastern Slopes Rangeland Seeds Ltd. (1994; 1995) sites was based on prescribed seeding in the Rumsey Block for the construction year (Elsinger 2009).

A state and transition model was developed showing transitions between original grassland states and disturbed states. The model is based on knowledge of oil and gas construction practices and references to grazing impacts. Potential returns to reference states were based on findings in the literature regarding grazing impacts and vegetation species succession and competition.

8.3 Statistical Analyses

Two-way cluster analysis, using Ward's method, and the resulting dendogram was used to classify disturbed sites (Kent and Coker 1992). Reference sites were grouped according to their corresponding disturbed sites. Disturbed and reference sites were ordinated with non-metric multidimensional scaling (NMS) to display the separation between years and reference sites (McCune and Grace 2002). Key species and significant factors, such as bare ground, were overlain on NMS diagrams to display trends and to describe vegetation progression over 11 years.

Indicator species analysis (ISA) validated the dominant species of the plant communities. Indicator values (IV) are based on the relative abundance and relative frequency of each species in a group. IV values range from 0 to 100, where 100 indicates a species is exclusively found in a particular group (Dufrene and Legendre 1997). Nonparametric multiple response permutation procedure (MRPP), operating on Sorenson (Bray-Curtis) distance measures, was used to evaluate significant differences between disturbed and reference states (Zimmerman et al. 1985). MRPP generates a chance-corrected within-group agreement value (*A*), which evaluates the difference between species composition of grouped plots and the lower the *A* value, the more similar the groups. A negative MRPP value indicates heterogeneity between groups (McCune and Grace 2002). Classification and ordination used PC-ORD (version 5.31, MjM Software, Gleneden Beach OR). Differences between disturbed and reference states for specific species was determined by t-tests with IBM® SPSS® Statistics (version 18, SPSS, Chicago IL).

8.3.1 Comparison of 11 Years of Recovery

Classification of seven disturbed well sites, assessed in 1995 and again in 2006, including their reference sites, resulted in a well-distributed dendogram with 2.1% chaining and four vegetation communities (Table 8.1). Chaining is the addition of single items to existing groups. The lower the chaining, e.g. < 25%, the better defined are the groups (McCune and Grace 2002).

Community A – Rough Fescue / Western Porcupine Grass

Community A is dominated by *F. hallii* and *H. curtiseta*, and includes all references sites and one disturbed site from 2006 (Figure 2). The amount of *F. hallii*, mosses and lichens, low bare ground, and the lack of forbs such as *A. frigida* indicate little or no grazing (Willms et al. 1990).

Community B – Wheat Grasses

Community B is composed of only disturbed sites and is dominated by *T. officinale* and wheat grasses (*P. smithii*, *E. lanceolatus*, and *E. trachycaulus*) (Table 8.1). It includes another introduced grass, *Festuca ovina* L., (sheep fescue), commonly used in reclamation seed mixes prior to 1990, and found to persist by Desserud et al. (2010). Bare ground and *T. officinale* indicate continuing disturbance; nevertheless, this community has a greater abundance of native species than the other two disturbed communities. One of the sites classed as B in 1996, became A in 2006, still trending towards introduced grasses, but with more abundant native species (Figure 2). One site classed as B in 1995 remained B in 2006, probably due to the persistence of *F. ovina* and the competitive wheat grasses.

Community C – Smooth Brome

Community C is composed of only disturbed sites and is dominated by *B. inermis* (Table 8.1), a species known to dominate once established (Romo et al. 1990; Grilz et al. 1994). It includes *A. cristatum*, a species difficult to eradicate (Wilson and Pärtel 2003; Henderson 2005) and *P. pratensis*, also known to persist (Naeth et al. 1997; Desserud et al. 2010). Two sites classed as C in 1995 remained so in 2006, likely a result of the endurance of the introduced grasses, while one became community B, a result of increased *T. officinale*.

Community D – Kentucky Bluegrass

Community C is composed of only disturbed sites and is dominated by *P. pratensis* (Table 8.1), a result of initial seeding and possible over-grazing Bush (Schwan et al. 1949; Blood 1966; Bush 1998). This community contains abundant

Festuca rubra L. (creeping red fescue), known to increase by re-seeding (Suzuki et al. 1999; Van der Graaf et al. 2005), and benefit from grazing (Van der Graaf et al. 2005). *Elymus repens* (L.) Gould. (quack grass) also occurs in this community. It is considered a competitive weed throughout North America (Claus and Behrens 1976), with toxic properties for alfalfa (Toai and Linscott 1979), possibly explaining the low forb cover in this community. One site remained classed as D between 1995 and 2006, while one increased wheat grass cover and became B, possibly due to moderate grazing, which could have resulted in increased *P. smithii* (Dormaar and Smoliak 1985; Willms et al. 1986).

8.4 State and Transition Model

Cluster analysis of all disturbed sites resulted in a well-distributed dendogram (6.1% chaining). Eight disturbance states were identified at a cut-off of 50% of information remaining (Figure 3). Disturbance states were classified by dominant species, identified by IV and percent cover values (Table 8.2). Reference sites were initially grouped according to their corresponding disturbed site classification; however, due to similarities between them, reference sites were re-grouped into three original plant community states (Table 8.3). Comparison of disturbance and reference states for *F. hallii*, wheat grasses (*E. trachycaulus*, *E. lanceolatus*, *P. smithii*), and *P. pratensis* indicated significant differences ($P < 0.001$) and confirmed classification of plant community states (Figure 4).

8.4.1 Transition States

Hypothetical transition trends were caused by construction and reclamation methods, from severely disturbed states to minimal changes from original undisturbed states. Cattle-grazing was a potential contributing feature to differences in reference states and changes in disturbances over time.

T1. Light grazing, no future disturbances; possibly leading to a return to undisturbed states.

T2. Moderate grazing resulting in litter breakup, possible rough fescue reduction, and unwanted species removal; may assist in a return to undisturbed states.

T3. Heavy grazing resulting in elimination of *F. hallii* and other species intolerant of heavy grazing, ground exposure and soil degradation, and introduction of unwanted species such as *B. inermis*, *P. pratensis* and *A. frigida*; leading to or continuing a disturbance state.

T4. Construction with minimum disturbance such as narrow trenching or small well site size, natural recovery allowing incursion of native species through seed rain from adjacent grassland, or from undisturbed seed bank; leading to a return to undisturbed states.

T5. A moderate disturbance such as narrow trenching with topsoil removal or native species seeding; may allow a return to undisturbed states depending on type of seeding.

T6. Complete topsoil stripping, complete removal of original species, and seeding introduced species; resulting in slow or no return to a pre-disturbance state.

T7. Complete removal of introduced species and re-seeding with native species or native hay; requiring a lengthy time to succeed.

8.4.2 Reference States

Vegetation states 1 through 3 are reference sites, each of which is a variation of rough fescue grassland. All include a variety of shrubs and forbs, and the two species typical of rough fescue grassland in central Alberta, *F. hallii* and *H. curtiseta.*

Reference State 1 – Rough Fescue

State 1 is typical of undisturbed rough fescue grassland, with *F. hallii* dominating almost to the exclusion of other species (Moss and Campbell 1947; Looman 1969). Its species composition indicates little or no grazing, depicted by the amount of *F. hallii* and the lack of forbs such as *A. frigida* (Willms et al. 1990). The presence of *P. pratensis* and *P. smithii* indicate some grazing (Slogan 1997), possibly in the past, although it is unlikely these species will overtake *F. hallii* in this state.

Reference State 2 –Rough Fescue / Western Porcupine Grass

Moderate grazing (T2) may have produced changes in State 1 to State 2, which similar to Willms et al. (1986) and Pantel (2006) observations, has an increase in *H. curtiseta.* According to Slogan (1997), light or no grazing (T1) would result in an eventual return to State 1.

Reference State 3 – Western Porcupine Grass

Further grazing may have produced State 3, with a reduction in *F. hallii* (Looman 1969; Sinton 1980; Looman 1983) and an increase in *E. lanceolatus*, *Carex* spp., and *P. smithii* (Moss 1955; Pantel et al. 2011). Light or no grazing (T1) should result in an eventual return to State 2.

8.4.3 Disturbance States

Disturbance states ranged from introduced species dominance to diverse species resembling undisturbed grassland (Table 8.2; Table 8.4).

Disturbed State 4 – Smooth Brome

State 4 is one of the most disturbed states and the most dissimilar to its reference sites. Despite including older sites, it is dominated by *B. inermis*, a result of seeding (Table 8.4) and similar to the examination of well sites after 11 years, which showed both sites continued to be dominated by *B. inermis*. As Brown (1997) discovered, only a combination of cutting, burning, and herbicide application had any effect on *B. inermis*, and once reduced, it could be replaced by *P. pratensis*. The second most dominant species, *A. cristatum*, is also difficult to eradicate (Wilson and Pärtel 2003; Henderson 2005). Even if reduced, its seed bank persists for some time (Ambrose and Wilson 2003; Henderson 2005). *Festuca ovina* and *P. pratensis*, also found in this state, persist once established (Desserud et al. 2010). *P. pratensis* readily colonized disturbed sites (Bizecki Robson et al. 2004) and dominated pipeline disturbed sites (Naeth et al. 1997; Desserud et al. 2010). This state has crossed a recovery threshold and will require complete eradication of established species (T7), with a combination of cutting and herbicide application over a several year period, followed by re-seeding with native species (Figure 5).

Disturbed State 5 – Kentucky Bluegrass

The Kentucky bluegrass state is highly disturbed, dissimilar to its reference sites, caused by full-strip construction. Its sites are among the youngest, resulting in a low amount of litter and high amount of bare ground (Table 8.4). Dominance by *P. pratensis* and inclusion of *E. lanceolatus* and *E. trachycaulus* was the result of seeding. *Poa pratensis* readily produces seed and is strongly rhizomatous, allowing it to establish and spread rapidly (Tannas 2001). Once established it is difficult to remove, requiring several years of herbicide treatment, cutting and cultivation (Taylor et al. 1969). The 11 year examination (section 8.4.1), confirmed this with one site dominated by *P. pratensis. Poa pratensis* cover could have increased due to grazing (Schwan et al. 1949; Blood 1966; Bush 1998). The presence of *A. frigida* is another indication of grazing pressure (Dormaar and Willms 1990; Slogan 1997). Other introduced species are *E. repens*, *B. inermis*, and *T. officinale* (Table 8.2). This state has crossed a recovery threshold and will require complete eradication of established species with herbicide, cutting, cultivation, and re-seeding with native species to return to its undisturbed state (Figure 8.5).

Disturbed State 6 – Kentucky Bluegrass-Shrubs

State 6 is dominated by *P. pratensis*; however, unlike state #5, the majority of its other species are native. Species which would have been seeded are *P. smithii*, *B. inermis*, *F. rubra*, and *Melilotus officinalis* (L.) Lam. (Table 8.2). This state includes some of the oldest sites; consequently, species such as *S. occidentalis* and *H. curtiseta* had time to become established. The lack of *F. hallii* indicates this state would unlikely return to its original state (State 1) without species eradication and re-seeding. Nevertheless, with moderate grazing, *H. curtiseta* and *S. occidentalis* could increase (Moss 1955) and move this state towards reference State 2. Other indications of possible recovery are presence of moss and lichens, relatively low bare ground, and high amounts of litter (Table 8.4), consistent with Evans and Belnap (1999) observations. In the study of well sites after 11 years, one site originally classified as *P. pratensis* moved to wheat grass dominance, which could occur with this state if *P. smithii* were to increase. Heavy grazing would increase *P. pratensis* (Bush 1998) and possibly move this state to an even more disturbed state such as State 5 (Figure 8.5).

Disturbed State 7 – Northern Wheat Grass

Elymus lanceolatus dominates State 7, a result of seeding and probably a native cultivar. Hammermeister (2003) noted wheat grass cultivars are more aggressive than native species. The presence of *A. frigida* indicates grazing pressure (Dormaar and Willms 1990), and grazing could also contribute to increased wheat grasses (Moss and Campbell 1947). Younger sites make up this state, which has the greatest amount of bare ground among the disturbed sites. This state may not return to its original reference state without herbicide and re-seeding. Heavy grazing could result in a shift to *P. pratensis* dominance (Blood 1966) and move towards state 5, even further from recovery (Figure 8.5).

Disturbed State 8 – Western Wheat Grass

State 8 is similar to State 7, having been created by seeding with native wheat grass cultivars. It differs having been constructed with minor disturbances and is dominated by *P. smithii* rather than *E. lanceolatus* with less *P. pratensis* and more native species. *Pascopyrum smithii* is a strong competitor, appearing early in succession and remaining (Samuel and Hart 1994; Bush 1998). With moderate grazing, the existing native species might increase, moving this state back to its original reference state (3). On the other hand, heavy grazing could promote *P. pratensis* growth (Blood 1966) and transform it to State 5, the highly disturbed *P. pratensis* state.

Disturbed State 9 – Sedge / Wheat Grasses

A combination of *Carex duriuscula* C.A. Mey., *P. smithii*, and *E. lanceolatus* dominate State 9. Grazing pressure is indicated by abundant sedge dominance and *A. frigida* (Dormaar and Willms 1990; Slogan 1997). This state is comprised of older sites, resulting in less bare ground and more moss and lichens than other disturbed states. Unlike the previous disturbed states, it resembles its reference state; therefore, with light to moderate grazing could return to its original state (3). With increased grazing *P. smithii* could proliferate (Dormaar and Smoliak 1985; Willms et al. 1986) and this state could progress towards State 8 (Figure 5).

Disturbed State 10 –Rough Fescue - Shrubs

State 10 is similar to its reference state having no introduced species. Despite the majority of its sites having been fully stripped, and half seeded with introduced species, over time and possibly with minimal grazing, this state has moved towards its reference state (1). The abundance of *S. occidentalis* is likely a result of fire suppression or f grazing, as it is unpalatable (Smith et al. 1968) and thus will increase in grazed prairie (Moss 1955; Slogan 1997). Minimal grazing should increase *F. hallii* cover (Willms et al. 1988; Slogan 1997). Moderate grazing could result in an increase in *H. curtiseta* (Moss 1955) and move this state towards reference State 2.

Disturbed State 11 – Western Porcupine Grass / Rough Fescue

State 11 is the most similar to its reference state (3), a result of minimum disturbance and natural recovery. This state includes the youngest sites, having the benefit of newer low-impact construction techniques. All species are native; however, the presence of *A. frigida* may indicate grazing pressure. With continued moderate grazing, *H. curtiseta* should increase (Moss 1955), resulting in a trajectory towards reference state 3. Less grazing should promote *F. hallii* (Willms et al. 1990), resulting in a move towards reference State 2.

8.5 Summary

These results show distinct recovery trends caused by construction and reclamation methods, from severely disturbed states to minimal changes from original undisturbed states. Cattle grazing was thought to be a contributing feature to differences in reference states and changes in disturbances over time. While all disturbances were not identified, these results present a reasonable picture of the state of the Rumsey Block, its disturbed and undisturbed grassland, as illustrated in the state and transition model (Figure 8.5).

Forty-five percent of the disturbed sites (states 8 through 11), could return to a state similar to reference states with judicious grazing management. Full restoration to prior states is not possible as this would require establishment of the entire diversity of native species and full complement of late seral soils (Allen 1993). Instead, these states may be reclaimed, a process that requires lower diversity of original species but still a high level of ecosystem function, self-organizing and stable, and capable of existing without human intervention (Bradshaw 1983; Allen 1993). The time required to achieve reclamation is impossible to determine; however, possibly relative time frames for each disturbed state. States 10 and 11, already containing many of the species found in their reference states, may become similar more rapidly than states 8 and 9. While containing many native species, States 8 and 9 lack *F. hallii*; and therefore, will take longer and may never achieve a state similar to their reference states.

The remainder of the States (4 through 7) likely require human intervention including removal of existing species and further reclamation. All of these sites had strongly competitive introduced species or native cultivars seeded. Removal with herbicides, cutting, and cultivation may require several rounds over several years. Once eradicated, native species must be seeded; however, *F. hallii* and *H. curtiseta* are difficult to obtain and may be expensive.

8.6 Conclusions and Management Considerations

The states with the best recovery were constructed with minimal disturbance. Based on these results, minimum disturbance and natural recovery is recommended in grasslands. For small diameter pipelines, narrow width trenching is preferred. For wider pipelines and well sites, which require a larger disturbance, sod salvage, (Petherbridge 2000), may succeed. If seeding is required, an alternative is using native hay, cut from nearby undisturbed grassland, a process successfully employed by Desserud and Naeth (2011b).

This state and transition model presents actual results of construction and seeding methods; however, predicted outcomes as a result of reclamation or grazing practices are only possibilities. As with any model, results may be probable but can never be exact, and each disturbed state could move along trajectories unlike what was presented. Nevertheless, this model may allow land managers to estimate the outcome of new disturbances or disturbance renovations in rough fescue grassland, and assist them in making better decisions when confronted by new developments.

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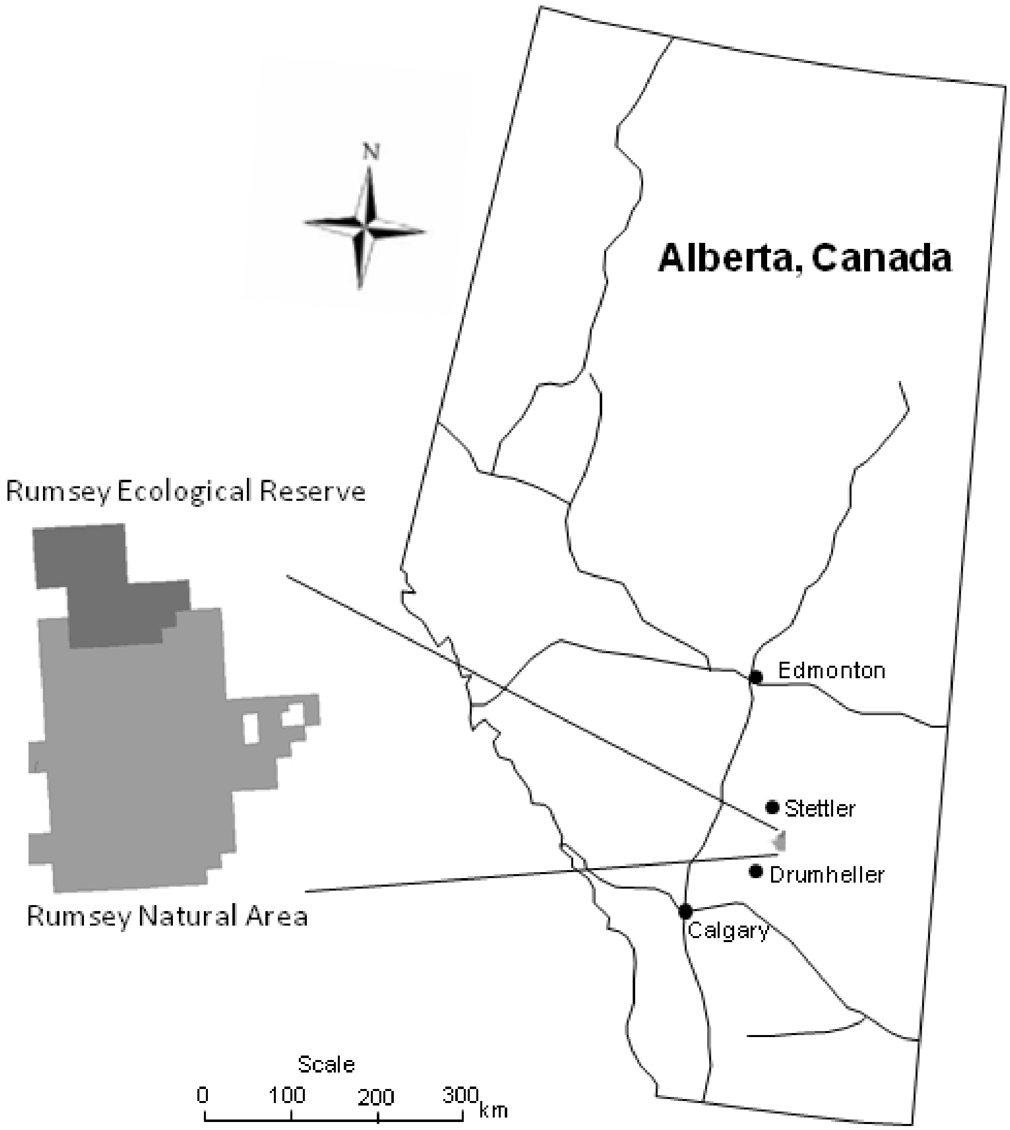

Figure 8.1 Study area in central Alberta, Canada.

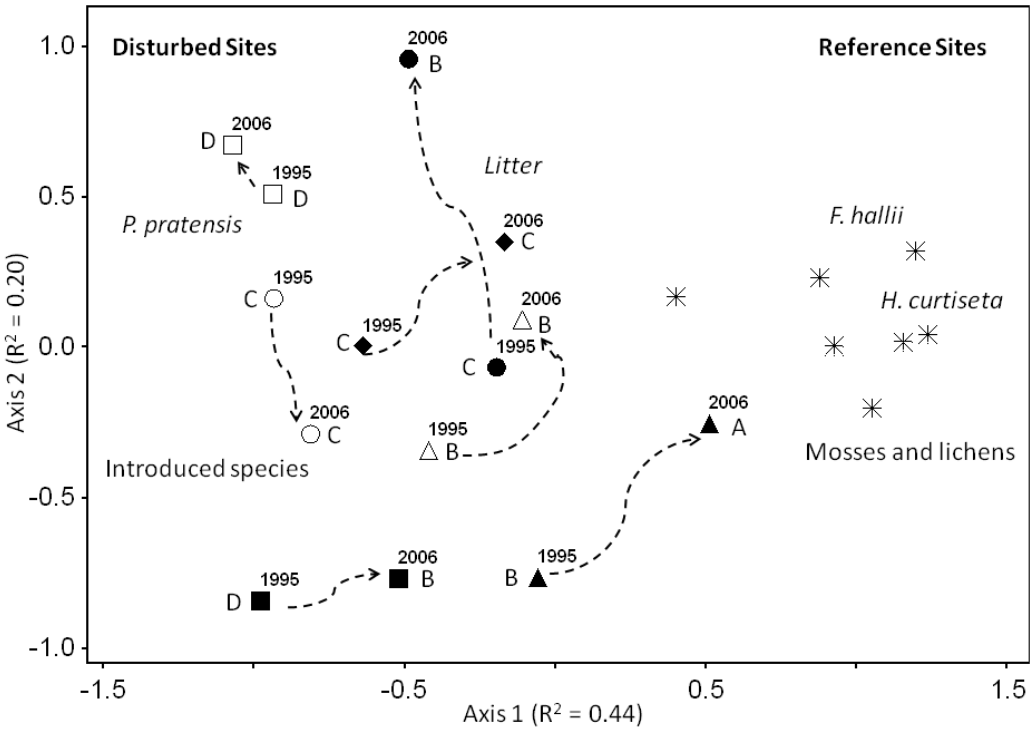

Figure 8.2 NMS ordination of well sites studied in 1995 and 2006 and their reference sites (starred) showing axes 1 and 2, the most significant variation explanation. Species and litter were overlain to show trends along the axes. Each disturbed site has a unique symbol.

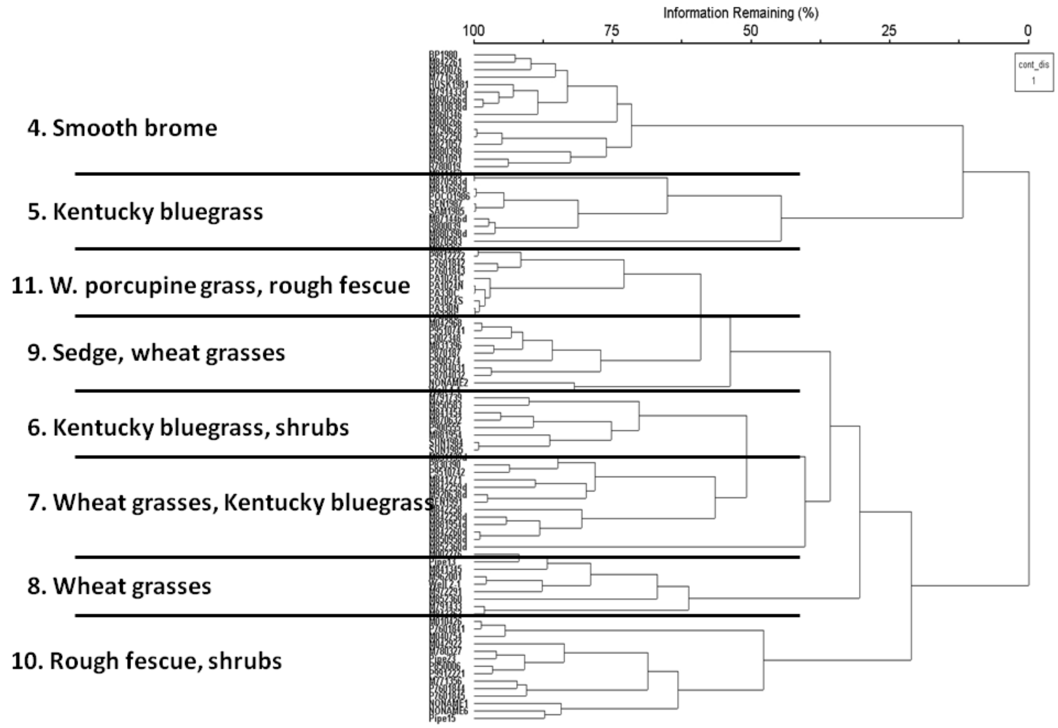

Figure 8.3 Cluster analysis dendogram, with 6.1% chaining, showing plant community groupings at a cut-off of 40 to 60% remaining information.

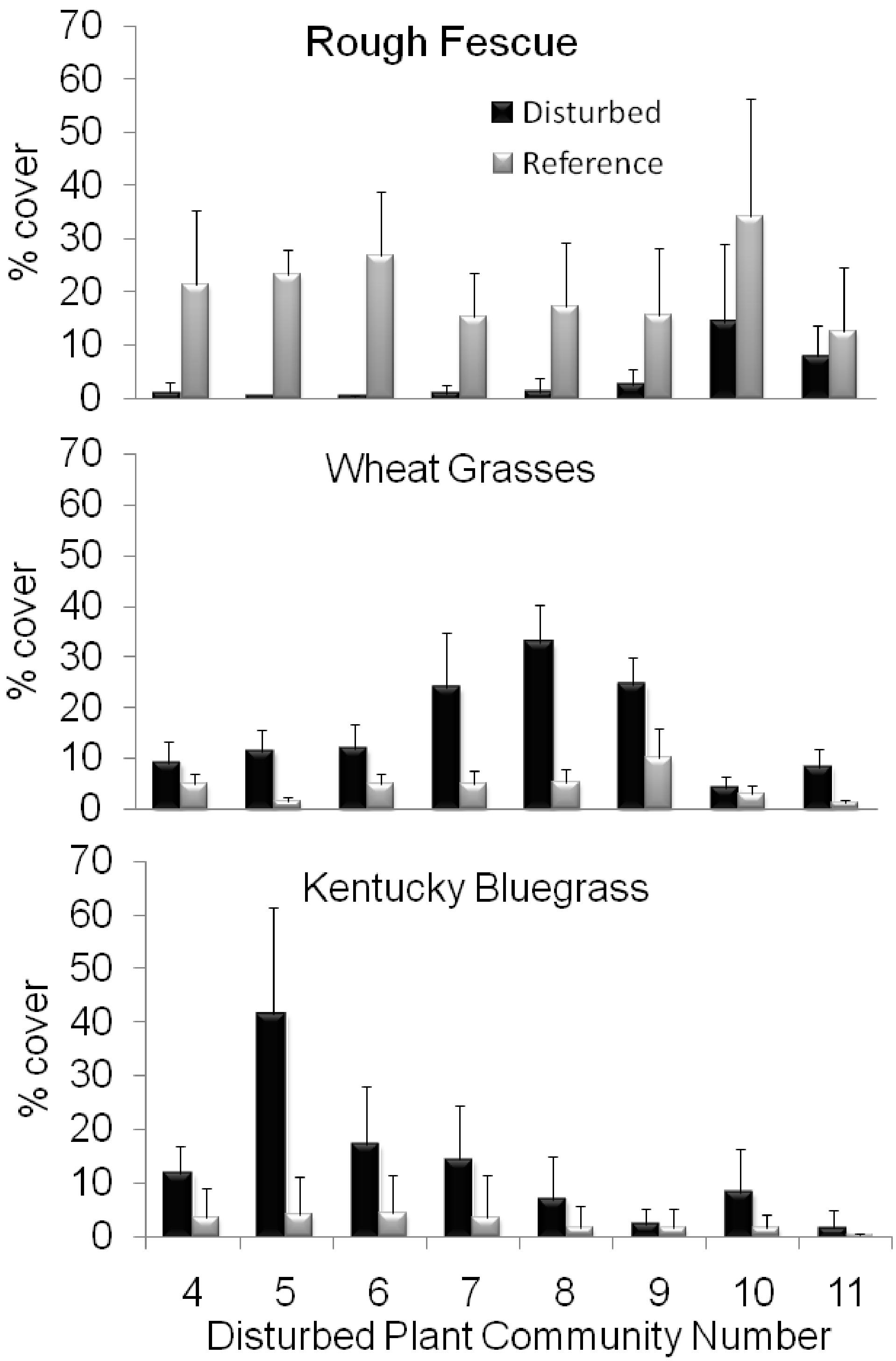

Figure 8.4 Comparison of ROW and control cover of rough fescue (*Festuca hallii*), Kentucky bluegrass (*Poa pratensis*), and wheat grasses (*Elymus trachycaulus*, *Elumus lanceolatus*, *Pascopyron smithii*). Differences between all Disturbed and Reference sites are significant ($P < 0.001$). Error bars are standard deviation.

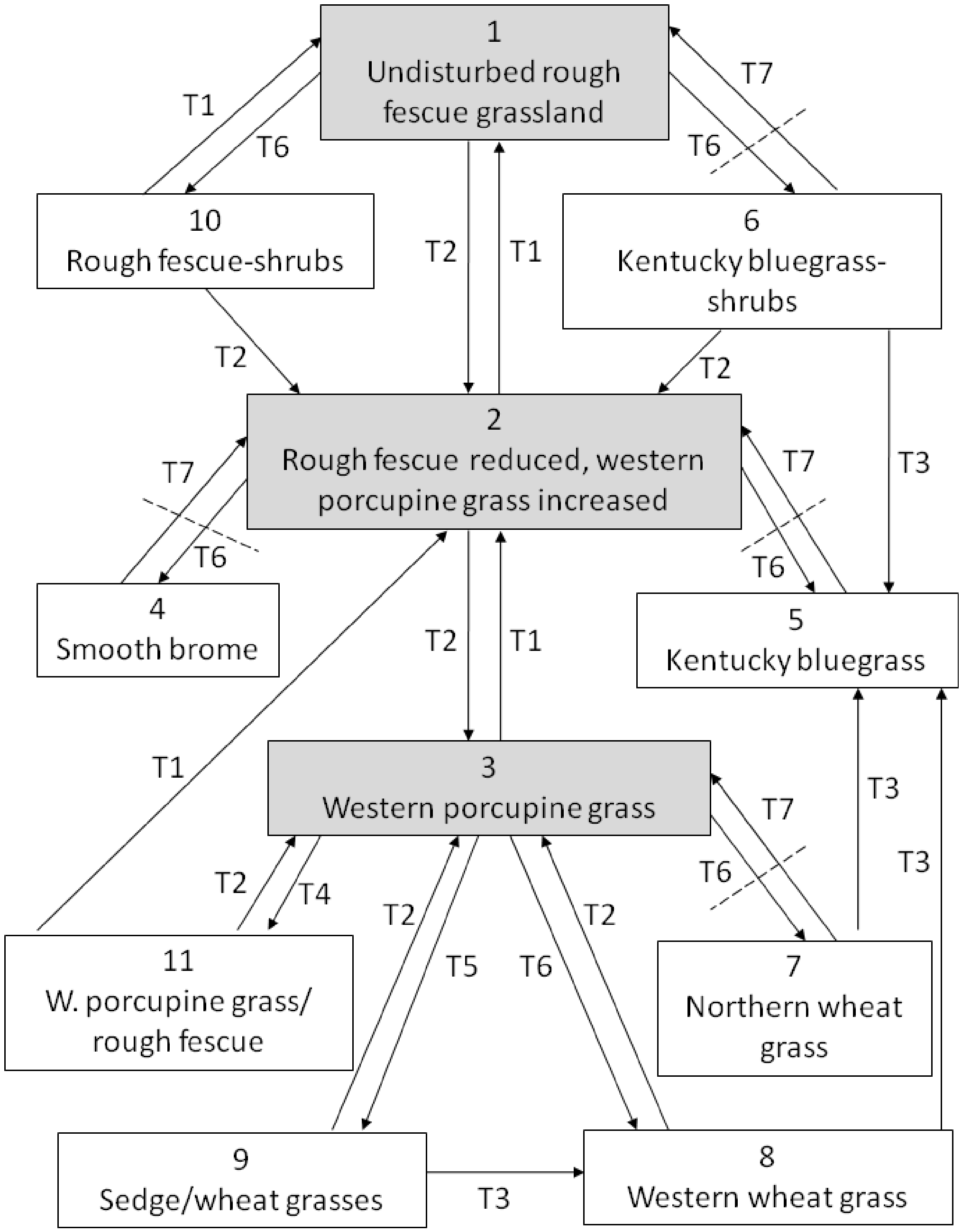

Figure 8.5 State and transition model of undisturbed reference states (shaded) and corresponding disturbed states. Transitions between states are indicated by arrows. Dashed lines represent thresholds for which substantial intervention is required for return to the original state. See text for details of states, transitions and thresholds.

Table 8.1 Plant communities on well sites assessed in 1995 and 2006 and their controls. Letters correspond to the NMS diagram.

A Rough fescue/Western porcupine grass (n = 15)			B Wheat grasses (n = 5)		
	Cover	IV		Cover	IV
Festuca hallii	15.6	99	*Taraxacum officinale*	19.1	66
Hesperostipa curtiseta	17.5	94	*Pascopyrum smithii*	10.5	52
Koeleria macrantha	6.8	84	*Nassella viridula*	7.8	50
Bouteloua gracilis	1.6	75	*Vicia americana*	1.6	38
Rosa arkansana	2.6	69	*Elymus lanceolatus*	5.8	28
Symphoricarpos occidentalis	5.4	56	*Festuca ovina*	9.1	23
Elymus lanceolatus	2.8	19	*Elymus trachycaulus*	4.5	20
Introduced grasses	4.4		Introduced grasses	23.6	
Forbs	30.7		Forbs	24.2	
Moss and lichens	11.1		Moss and lichens	0.4	
Bare ground	5.0		Bare ground	30.8	
Litter	20.6		Litter	17.1	

C Smooth brome (n = 5)			D Kentucky bluegrass (n = 3)		
Bromus inermis	29.2	99	*Poa pratensis*	37.5	60
Agropyron cristatum	11.5	80	*Festuca rubra*	24.6	65
Taraxacum officinale	8.7	30	*Poa compressa*	2.4	29
Artemisia frigida	5.1	28	*Elymus repens*	20.7	26
Poa pratensis	14.2	23	*Elymus trachycaulus*	2.2	17
Elymus trachycaulus	2.1	14	*Elymus lanceolatus*	3.1	14
Elymus lanceolatus	3.2	13	*Pascopyrum smithii*	2.0	12
Introduced grasses	60.5		Introduced grasses	83.0	
Forbs	8.8		Forbs	2.9	
Moss and lichens	0.4		Moss and lichens	0.0	
Bare ground	29.5		Bare ground	28.3	
Litter	20.2		Litter	13.7	

Table 8.2 Disturbed state plant communities showing top seven species based on cover and indicator species analysis values (IV).

4 Smooth brome (n=15)	Cover	IV	5 Kentucky bluegrass (n=11)	Cover	IV
Bromus inermis	28.0	70	*Poa pratensis*	41.5	40
Agropyron cristatum	5.9	48	*Taraxacum officinale*	7.4	19
Elymus repens	5.7	19	*Elymus lanceolatus*	8.6	15
Poa pratensis	12.0	11	*Elymus trachycaulus*	1.5	8
Taraxacum officinale	4.4	9	*Artemisia frigida*	4.3	8
Festuca ovina	2.3	7	*Elymus repens*	5.6	4
Pascopyrum smithii	5.1	6	*Bromus inermis*	2.9	4

6 Kentucky bluegrass-shrubs (n=9)			7 Northern wheat grass (n=13)		
Poa pratensis	17.1	15	*Elymus lanceolatus*	15.2	25
Symphoricarpos occidentalis	6.8	13	*Elymus trachycaulus*	4.4	24
Hesperostipa curtiseta	4.5	12	*Artemisia frigida*	9.7	18
Pascopyrum smithii	6.4	8	*Poa pratensis*	14.3	13
Bromus inermis	3.4	7	*Pascopyrum smithii*	4.3	5
Festuca rubra	1.9	7	*Artemisia ludoviciana*	1.3	4
Melilotus officinalis	4.4	7	*Bouteloua gracilis*	0.5	4

8 Western wheat grass (n=9)			9 Sedge/wheat grasses (n=9)		
Pascopyrum smithii	25.5	42	*Carex duriuscula*	14.6	68
Nassella viridula	9.9	28	*Artemisia frigida*	16.2	32
Poa compressa	2.2	15	*Koeleria macrantha*	6.5	31
Festuca ovina	6.7	13	*Hesperostipa curtiseta*	7.7	23
Koeleria macrantha	2.8	12	*Pascopyrum smithii*	13.5	22
Elymus trachycaulus	2.4	8	*Elymus lanceolatus*	10.7	20
Poa pratensis	6.9	6	*Bouteloua gracilis*	14.6	11

10 Shrubs-rough fescue (n=14)			11 Western porcupine grass/rough fescue (n=10)		
Symphoricarpos occidentalis	28.3	67	*Hesperostipa curtiseta*	9.9	30
Festuca hallii	14.3	44	*Festuca hallii*	7.9	25
Nassella viridula	7.3	25	*Vicia americana*	2.2	22
Rosa arkansana	2.3	18	*Rosa arkansana*	2.8	21
Hesperostipa curtiseta	7.7	18	*Helictotrichon hookeri*	1.0	21
Artemisia ludoviciana	3.8	17	*Koeleria macrantha*	3.5	15
Carex inops	1.1	14	*Artemisia frigida*	6.8	11

Table 8.3 Reference state plant communities showing top ten species based on cover and indicator species analysis values (IV).

1 Rough Fescue			2 Rough fescue/Western porcupine grass			3 Western porcupine grass		
(n=26)	Cover	IV	(n=13)	Cover	IV	(n=37)	Cover	IV
Festuca hallii	32	47	*Festuca hallii*	22.0	32	*Hesperostipa curtiseta*	15	35
Symphoricarpos occidentalis	6.6	28	*Hesperostipa curtiseta*	15	37	*Festuca hallii*	14	21
Hesperostipa curtiseta	11.0	25	*Achillea millefolium*	1.7	29	*Elymus lanceolatus*	1.8	29
Solidago missouriensis	1.0	22	*Carex duriuscula*	5.1	29	*Carex duriuscula*	6.3	28
Rosa arkansana	2.4	21	*Anemone patens*	1.3	29	*Bouteloua gracilis*	1.8	26
Carex duriuscula	5.0	20	*Koeleria macrantha*	5.6	27	*Erigeron caespitosus*	2.1	25
Koeleria macrantha	4.2	19	*Symphoricarpos occidentalis*	5.9	26	*Pascopyrum smithii*	3.3	24
Poa compressa	0.7	19	*Bouteloua gracilis*	1.6	25	*Achillea millefolium*	1.5	21
Poa pratensis	2.3	19	*Solidago missouriensis*	1.1	23	*Rosa arkansana*	1.4	14
Pascopyrum smithii	2.3	18	*Pascopyrum smithii*	2.4	16	*Carex inops*	1.5	14

Table 8.4 Disturbed states ground cover, construction and seeding attributes, and MRPP analysis results. Means with standard deviation in parentheses. Build and seeding values are explained in the text.

	4 Smooth brome (n=15)	5 Kentucky bluegrass (n=11)	6 Kentucky bluegrass/ Shrubs (n = 9)	7 Northern wheat grass (n = 13)
Moss/Lichen	0.9 (1)	0.0	2.0 (3)	0.5 (1)
Bare ground	21.2 (20)	28.4 (21)	10.8 (8)	39.6 (28)
Litter	20.9 (21)	3.7(12)	20.3 (23)	12.1 (25)
Age	21 (5)	11 (4)	16 (6)	12 (6)
Build	Full strip	Full strip	Full strip	Full strip
Seeding	Introduced	Introduced	Introduced	Rumsey Mix
MRPP	0.21, $P < 0.001$	-0.01. $P = 0.471$	0.13, $P<0.001$	0.13, $P < 0.001$

	8 Western wheat grass (n = 9)	9 Sedge/ Wheat grasses (n = 9)	10 Rough fescue/ Shrubs (n = 14)	11 Western porcupine grass/ Rough fescue 10
Moss/Lichen	1.1 (2)	4.3 (3)	1.9 (3)	1.1 (2)
Bare ground	15.3 (16)	9.2 (7)	4.0 (4)	36.9 (28)
Litter	31.5 (19)	16.9 (6)	32.9 (28)	21.6 (18)
Age	17 (7)	17 (11)	20 (13)	9 (11)
Build	Narrow strip	Full strip	Narrow strip	Minimum
Seeding	Rumsey Mix	Native Mix	Rumsey mix	Natural recovery
MRPP	0.16, $P < 0.001$	0.07, $P < 0.001$	0.06, $P < 0.001$	0.02, $P = 0.072$

CHAPTER 9. SYNTHESIS AND MANAGEMENT RECOMMENDATIONS

9.1 Can rough fescue be restored?

Festuca hallii (plains rough fescue) responded positively to several reclamation factors in this research project. Minimal disturbance appears to be one of the most important. Throughout the Rumsey Natural Area, the best recovered oil and gas disturbances were narrow or constructed with minimal soil disturbance (Chapter 8). In the assessment of natural recovery, the lowest disturbances resulted in the best rough fescue recovery (Chapter 4). The deep-rooting characteristics of *F. hallii* would make it a good candidate for retaining intact sod and preserving whole plants, resulting in successful recovery, as reported by Petherbridge (2000). Its negative relation to the high pH (over 8) and EC (above 0.2), of admixed soils on one well site (Chapter 2) also underlines the importance of reducing disturbance.

This research sheds doubt the effects of topsoil storage on *F. hallii* establishment. One factor, the reduced viability of arbuscular mycorrhizae fungi (AMF) after several years of topsoil storage (Rives et al. 1980; Gould and Liberta 1981; Liberta 1981), may not affect *F. hallii.* Although *F. hallii* is readily colonized by AMF (Molina et al. 1978), contrary to expectations, juvenile *F. hallii* growth increased with less AMF colonization, leading to the conclusion topsoil storage may not be detrimental to *F. hallii* establishment (Chapter 5). My research led to the possible conclusion that AMF is not obligative for *F. hallii*, and the relationship may not be mutualistic for juvenile plants.

Competition is another important factor affecting *F. hallii* establishment. Requiring several years to become established, *F. hallii* is a slow growing species, making it susceptible to many factors during its juvenile state. Climate, especially conditions in prior years, is important, as *F. hallii* appeared to respond well to good soil water conditions the following year (Chapter 2). The proximity of fast growing, large species, such as *Elymus trachycaulus* (slender wheat grass) may impede its early growth, as was shown in its response to a monoculture versus a native seed mix. *Festuca hallii* may be particularly susceptible to proximity of *Poa pratensis* (Kentucky bluegrass) (Chapter 3). Straw, as mulch, benefited rough fescue, possibly by reducing establishment of non-native species. Native hay, while providing a seed source, also provided mulch that may have impeded establishment of non-native species (Chapter 6). In addition, native hay produced grassland species normally occurring with *F. hallii*, and which may co-exist without competition. Natural recovery may be another mechanism to reduce competition. When combined with minimal disturbance and intact sod segments, natural recovery would allow established *F. hallii* plants to compete with species that might emerge from the seed bank or seed rain (Chapter 4).

Straw amendment appeared to benefit *F. hallii*, where definite increases in biomass, leaf length and tiller counts were found in the highest straw amendments (Chapter 2). My initial hypothesis that straw would benefit *F. hallii* by reducing nitrogen was not proven. Instead, a combination of nutrients released by straw, such as ammonium and potassium, and reduced competition from non-native species as a result of straw mulching appeared the most beneficial for *F. hallii* (Chapter 2). *Festuca hallii* use of ammonium was also observed in the AMF experiment (Chapter 5).

The reaction of *F. hallii* to soil water underlined its growth habits and ecosystem status. The best established *F. hallii* plants were found at the driest site, Drumheller, similar to findings by Stout et al. (1981) for *F. campestris*. *Festuca hallii* also responded positively to prior-year soil water, a reaction consistent with bunch grass development (Holechek et al. 2004).

Festuca hallii established well from seeding, which involved viable, wild-harvested seed (Chapter 2). Harvesting seed during a flowering year produced viable seed (Chapter 2), as did cutting native hay (Chapter 6). Successful establishment may be related to the fact that seed was obtained from the ecoregion where reclamation took place, and thus would be the optimal ecoptype and contain the genetic base of viable individuals (Lippitt et al. 1994). Natural selection may result in plants performing better in one location than another, with slight genetic variations caused by climate, soils, insects and microorganisms (Turesson 1922; Norcini et al. 2001). Seeding rate of *F. hallii* and relative seeding of other species was important. Low seeding rates, such as less than 15 kg/ha, resulted in bare patches allowing infill from adjacent grassland. A seed mix, including few or no wheat grasses, and instead a mix of other native grasses common in the area, was most successful (Chapter 3). Wheat grasses appeared least affected by seed mix rates, as their high germination, establishment and large growth form, effectively prevented *F. hallii* establishment. Although *E. trachycaulus*, an early seral species, will die out within 5 to 10 years (Desserud 2006), its initial dominance is probably sufficient to suppress slow growing species such as *F. hallii*.

9.2 *Poa pratensis* and *Bromus inermis* Reactions to Straw

Straw had little effect on *P. pratensis* although it appeared to respond positively to potassium (K^+) released from straw (Chapter 2). *Poa pratensis* may benefit from soil disturbance as it responded well to higher pH and EC levels (Chapter 2). *Poa pratensis* also responded to higher nitrate levels released by soil tilling, a known response (Christians et al. 1979; Bowman et al. 1989). These interactions may explain why *P. pratensis* appears so readily on soil disturbances.
Poa pratensis soil water responses were consistent with its growth habits, generally preferring mesic habitat and annual growth occurring in early summer (Sinton et al. 1996). Straw had a strong negative effect on *Bromus inermis* (smooth brome), possibly a result of ammonium and potassium released by straw (Chapter 7).

9.3 What is still unknown?

My research exposed possible reactions of *F. hallii* to soil properties and more detailed research could reveal specific interactions. The mechanisms of ammonium and potassium nutrients for *F. hallii*, and *F. hallii* reaction to higher pH and EC, could help explain poor recovery on disturbed sites where soil admixing may have altered natural values. Growth chamber experiments with controlled nutrient absorption and pH/EC variations followed by tissue analyses would help validate my field observations.

The ease with which *P. pratensis* establishes on disturbed soils could be confirmed with research into pH and EC levels. Much research has already been done on optimum pH levels for *P. pratensis*; however little is available on pH above 7.

Further research into native hay sources is required including timing of cutting, perhaps several times over the summer, and the viability and longevity of native hay bales. Being able to store native hay for future use would be important for well-site restoration, which may take place several years after construction, or for retaining species, such as *F. hallii*, that do not produce seeds every year.

The role of arbuscular mycorrhizae fungi with *F. hallii* is still unclear. Further research using sterile soils or hydroponics with AMF inoculants could provide more exact results. Field experiments tracking *F. hallii* growth over several years, with and without AMF could provide insights into AMF effects and over time; however, the difficulty of isolating environmental factors in the field would reduce the success of such experiments.

The role of competition in *F. hallii* establishment could be further analyzed. Growth chamber experiments could assess *F. hallii* reactions to species such as wheat grasses, with controlled nutrients and tissue absorption analyses.

Study into the effects of wheat straw, potassium and ammonium on existing *B. inermis* stands and in new native prairie restoration sites could determine if these could aid in removing or controlling brome.

9.4 Management Recommendations

1. Straw is an inexpensive and readily available amendment for newly reclaimed disturbances. Incorporated straw increases soil nutrients, such as ammonium, and provides surface mulching that suppresses some non-native species. Straw

must be weed-free, and should be obtained from a reputable supplier.

2. In areas where smooth brome is an issue, straw amendments may assist in preventing its encroachment into disturbed sites.
3. The importance of avoiding soil admixing was illustrated by the different responses of rough fescue and Kentucky bluegrass to pH and electrical conductivity levels. Admixed soil, which may be alkaline, appears to favour Kentucky bluegrass and hinder rough fescue.
4. Nitrates released following soil disturbance, for example spreading and feathering topsoil, may promote establishment of Kentucky bluegrass from seed bank propagules, which supports the need for minimum disturbance.
5. Native hay may provide a viable seed source for reclaimed grassland, for disturbances in close proximity to intact native grassland. Seasonal timing of hay cutting may be important in determining which seeds will be available and viable. To obtain a full suite of native grassland species, cutting hay several times, such as early, middle and late summer is recommended.

9.5 Literature Cited

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APPENDIX A RUMSEY BLOCK PLANT COMMUNITIES PRIOR TO 1982 (Wroe 1971; Fehr 1982)

Plant Community	Hydrologic Function	Terrain	Soils
Agrostis scabra-Achillea millefolium-Antennaria parvifolia	Subxeric to submesic	Depressions on moraine plateau crests	Dark Brown Chernozem
Carex atheroides	Subhydric and hydric	Depressions	Solonetzic or Dark Brown Chernozem
Eleocharis palustris – Typha latifolia		Wetter marshes	Sandy loam, heavy clay
Koeleria macrantha – Pascopyrum smithii		Alkaline depressions	Solonetzic
Festuca hallii	Submesic to mesic	Level terrain, north-facing slopes	Black or Dark Brown Chernozem
Poa palustris – Cirsium arvense		Wet meadows	
Puccinellia nuttalliana	Subhydric	Saline meadows / saline sloughs	Sandy loam, heavy clay
Stipa curtiseta - Festuca hallii	Xeric to submesic	South-facing slopes	Dark Brown Chernozem
Stipa curtiseta - Artemisia frigida	Xeric	Crests and upper slopes	Orthic, Calcacerous and Regosolic Dark Brown Chernozem
Populus tremuloides	Submesic to subhydric	Depressions, north-facing and occasionally south-facing slopes	Black Chernozem, Humic Gleysol
Elaeagnus commutata- Symphoricarpos occidentalis	Mesic to submesic	Middle and lower levels of north or south-facing slopes	

Appendix A continued

Plant Community	Hydrologic Function	Terrain	Soils
Salix petiolaris	Mesic to hydric	Around ponds and wetlands	Rego and Orthic Humic Gleysol
Symphoricarpos occidentalis	Submesic	North/east and south/west slopes and on the edges of *Populus tremuloides* and *Salix* spp. communities	Orthic and Rego Dark Brown Chernozem

APPENDIX B HISTORY OF LAND USE IN THE RUMSEY BLOCK

The following dates were compiled from several reports and papers (Wroe 1971; Bradley and Bradley 1977; Alberta Agriculture Food and Rural Development: Public Land Services 1993; Eastern Slopes Rangelands Seeds Ltd. 1994; Alberta Environmental Protection and Alberta Agriculture Foods and Rural Development 1998).

1985 Imperial Ranch commences ranching and cattle grazing

1907 Dominion of Canada Legal Land Survey of the area completed.

1911 Grazing rights purchased by Burns ranching interest from the Imperial Ranching Co.

1917 Grazing lease acquired by Tom Usher and Jim Walters.

1920 (or 1937) Lease divided and Usher's portion encompassed the current Ecological Reserve

1947 Ducks Unlimited recommends draining potholes into each other to create waterfowl habitat. Ditches, still visible, were dug in the 1950s and 1960s, but were unsuccessful.

1948 Land inspectors report the soil is arable but topography and climate preclude economic farming.

1971 S. Hatfield, Lands Division inspector, recommends preservation of an expanse of grassland in the area.

1975 Usher Ranching Ltd. lease established in the area of the Ecological Reserve.

1990 Rumsey Ecological Reserve designated by Order-In-Council 511/90. ERCB (now EUB) issues IL 90-21 with guidelines for application reviews for all oil and gas development in the Rumsey Block, pending development of the RID.

1993 Rumsey Parkland South Regionally Integrated Decision (RID) published.

1994 Ranching leases in the Rumsey Block include Usher (3,337 ha Ecological Reserve, 3,237 ha Parkland South), Stewart (5,989 ha Parkland South), Stewart Atlin (842 ha Parkland South), Jakes Butte (1619 ha Parkland South), Rowley (3,237 ha Parkland South).

1996 Rumsey Natural Area (Parkland South) designated by Order-In-Council 390/96.

1998 Rumsey Ecological Reserve Management Plan Published.

Stocking rate in the Rumsey Block was originally set to 3.2 AUM/ha in 1920, decreased to 2.0 AUM/ha in 1962, increased to 2.8 AUM/ha in 196), and settled at 2.6 AUM/ha in 1994. Grazing occurred between June and October on most leases over this time, with some rotational and continuous year long grazing.