Influence of management and disturbance history on germinable seed bank composition and legume recruitment in Alberta's Central Parkland and Dry Mixedgrass prairie

By

Lysandra Ann Pyle

A thesis submitted in partial fulfillment of the requirements for the degree of

Doctor of Philosophy

In

Rangeland and Wildlife Resources

Department of Agricultural, Food and Nutritional Science

University of Alberta

Abstract

Seed banks (SB) are a cryptic component of grassland plant community (PC) diversity and are overlooked for their contribution of a significant ecological service in the form of plant propagules that replenish the aboveground plant community with new individuals and thereby aid in recovery following disturbance. SB composition often differs from the aboveground plant community, as it is a legacy of historical disturbance events, ongoing succession, and seasonal shifts in PC composition. Alberta's Central Parkland (CP) fescue grasslands have been subject to significant anthropogenic disturbance through cultivation and changes in fire and grazing regime, with many grasslands now dominated by introduced forages, either intentionally seeded or those encroached under contemporary patterns of grazing. Dry Mixedgrass (DMG) prairie has experienced similar disturbances and is recognized as a region wherein industrial activity (e.g. oil and gas infrastructure) can cause a decline in native grassland and introduced vegetation invasion.

In this study, germinable SBs in the topsoil of grasslands were characterized, including managed Parkland-Boreal pastures of central Alberta and native DMG prairie disturbed by natural gas pipelines. SBs were examined for their similarity to above-ground PC, and their composition linked to ongoing disturbances and/or specific management attributes. This research was conducted with a focus on potential legume recovery in the CP and examined legumes as invasive species along industrial disturbance. Legume emergence was tested further in an additional study looking at the recruitment and survival of native, agronomic, and escaped (potential weedy) agronomic legumes into native grasslands.

In the CP, 102 pastures were sampled, and a previous history of cultivation was found to have a significant influence on both PCs and SBs, including a reduction in native plants, particularly perennial grasses. Unexpectedly, grazing systems (continuous vs. rotational) led to few differences in PCs, SBs, and soils, likely due to similar stocking rates. PCs and SBs each responded to unique historical management factors, with SB composition more responsive to livestock husbandry (i.e., manure spreading, bale grazing, etc.). Similarity in species richness between the SB and PC was related to a few key aspects of management: 1) low RH scores were associated with high similarity and greater SB densities of forbs, 2) previously cultivated and well-established pastures had a higher similarity comprised of mostly introduced forage grasses. Legumes like clovers formed persistent SBs and were resistant to management actions like recent herbicide use.

In DMG prairie, both aboveground PCs and SBs exhibited legacy effects of natural gas pipeline installation, which were further influenced by pipeline diameter and age. Distinct legacy effects were also evident along spatial gradients with increasing distance (to 55 m) from the pipelines. SBs directly on pipeline trenches were associated with higher densities of introduced *Melilotus* spp. and two native grasses typically used to revegetate prairie disturbances; however, these were not representative of native grassland. Wide diameter pipelines were more likely to have greater seed densities of introduced grasses like *Agropyron cristatum* and *Poa pratensis*, which can be invasive in native grasslands. Legacy effects of pipeline disturbance were most pronounced for the cryptic biological soil crust (BSC) community, where the recovery of macro-lichens was nearly absent. BSCs were also linked to shifts in SB composition, where BSC elimination resulted in greater bare soil and higher densities of introduced species in the SB.

Within both native and introduced grasslands of the CP and DMG, legume (six species) recruitment and survival from an artificial SB were monitored over three growing seasons. At all locations litter (ambient or reduced) and defoliation (defoliated or non-defoliated) were manipulated to emulate vegetation structural (i.e., competitive) and microclimate changes that could occur under contrasting management practices (grazing intensities or range health). Litter and defoliation treatments significantly influenced PC structure, with litter removal increasing light availability, and defoliation increasing soil temperature. Different legume species also exhibited unique establishment responses to treatments, likely reflecting contrasting seed ecology. Aspects of germination and recruitment were frequently linked to PC structure, composition, and competition, which were often influenced by the treatments imposed.

Overall, this research greatly expanded our understanding of the influence of disturbance regimes on grassland range health, as well as aboveground vegetation, seed bank and cryptic BSC composition, within both introduced and native grasslands.

Preface

This thesis is an original work by myself, Lysandra Pyle, and I am responsible for directing the collection of data presented and the analysis. This research project received research human ethics approval (for a producer management survey) from the University of Alberta Research Ethics Board, Project Name "Pasture Seed Bank Study", ID: Pro00030842, April 25, 2012.

Chapter 3, in part, of this thesis has been accepted for publication as Pyle, L., Hall, L.M, and Bork, E.W. 2017. Survey of Management Practices and Range Health in Northern Temperate Pastures. Canadian Journal of Plant Science. I was responsible for data collection and analyses, supervisors Edward Bork and Linda Hall contributed to manuscript preparation.

Acknowledgements

My research was made possible by the participation of many individuals. Special thanks go to the many livestock producers and counties, as well as the Blackfoot Grazing Reserve who allowed access to their land for sampling during my North Central Alberta seed bank survey. The time they spent talking to me about their management was significantly appreciated, especially when they were kind enough to refer me to friendly family members and neighbors for potential study sites. Further thanks goes to the Rangeland Research Institute's Roy Berg Kinsella Research Ranch and Mattheis Research Ranch which provided study sites and resources for my pipeline seed bank survey and legume demography study. Guidance from Edward Bork, Barry Irving, and Don Armitage regarding selection of survey locations and advisement on placement of semi-permanent plots was also appreciated.

Funding from NSERC, DuPont, DowAgrosciences, and the Rangeland Research Institute paid for field expenses, conferences, and assistants over the years. Additional funding from scholarship providers like the SM Blair Foundation, Margaret (Peg) Brown, the Government of Alberta, and the Canadian Weed Science Society's PhD (Monsanto) Award made life more comfortable and supplemented research funding. I am also appreciative of the Allicia Hargrave Memorial award from the Inter-Mountain Section of the Society for Range Management.

Field assistants Carly (Hansen) Moore, Laurie Freirichs, Mark Donner, Claire Kisko, and Leah Rodvang endured long hours in field soil coring ($n \approx 9,606$ seed bank cores, not including numerous cores for measuring soil properties), often sacrificing evenings and some weekends to push for as many samples possible during our short survey window. Help from Claire and Leah was greatly appreciated when gluing legume seeds to toothpicks and seeding with said toothpicks ($n \geq 15,360$) for the legume demography study. Erica Schell and Caroline Martin assisted in both the lab and field, making the processing of soils and clipping of biomass (which they identified and clipped by species) quick and efficient. Technical guidance regarding the processing of soils was provided by Daniel Hewins and Mark Lyseng.

Over my many years study at the University of Alberta my ideas and experiences have been influenced by many graduate students and undergraduate students, too many to mention. I am thankful for the numerous opportunities to teach and supervise students which were both challenging and memorable, and helped me become better at communicating my science. This inspired a desire to network and speak publicly about my interest in grasslands, ecology, and research once this platform was gone.

The Rangeland Research Institute also deserves special acknowledgement, as the prairies it preserves across Alberta are vast and beautiful. Field seasons based out of these facilities were amazing, and I spent a lot of time getting to know the flora and landscapes of Matthias Ranch in particular. Leaving beautiful prairies behind in Saskatchewan, this place filled a void that wore me down during my long stay in Edmonton.

Through my studies my family and friends back home have been important for supporting and motivating me to finish. Over the years I have accumulated significant social debt, and I hope to someday repair these neglected relationships and reciprocate support in meaningful ways.

Supplemental employment through the Manitoba Forage and Grassland Association and the Alberta Biodiversity Monitoring Institute provided early career development and an opportunity to apply skills developed through my graduate education.

Finally, I appreciated the patience and flexibility of my primary supervisors Edward Bork and Linda Hall. Somehow, they managed to advise me through writing my ugly thesis and have read it many times. For that, I apologize.

Table of Contents

Abstract ……………………………………………………………………………………..…..………ii

Preface ………………………………………………………………………………………..……….iv

Acknowledgements …………………………………………………………………….….…........v

Table of Contents …………………………………………………...…………..………….....vii

Abbreviations ……………………………………………………………………….....……..xiii

List of Figures …………………………………………………………………...…..…………xiv

List of Tables ………………………………………………..………………………........…xxi

CHAPTER 1. Influence of management on seed bank composition and legume recruitment in Alberta's Aspen Parkland and Mixedgrass Prairie………………………………..…………………1

1.1 Background……………………………………………………………..………1
1.2 Research Objectives……………………………………………………….…2
1.2.1 Study #1: Linking seed bank composition and legume recovery in pastures to management history and site conditions………………………………..3
1.2.2 Study #2: Understand pipeline disturbance impacts on seed bank composition and displacement of Mixedgrass Prairie………………………………..4
1.2.3 Study #3: Recruitment potential of agronomic, escaped-agronomic, and native legumes from an artificial seed bank…………………………………... 6
1.3 Implications……………………………………...……………………….….. 7
1.4 Literature Cited……………………………………………………………… 9

CHAPTER 2. Seed Banks and Legumes in the Northern Great Plains: A Literature Review..…...11

2.1 Introduction to Seed Banks……………………………………………...……11
2.2 Types of Seed Banks……………………………………………………….14
2.2.1 Transient Seed banks..…………………………………………………… 14
2.2.2 Persistent Seed Banks…………………………………………...….... 15
2.3 Seed Bank Formation……………………………………….…………………...16
2.4 Grassland Seed Banks………………………………………………...……17
2.5 Factors Influencing Seed Bank Composition………………..……………………. 18
2.5.1 Anthropogenic Disturbance and Agronomic Impacts……..…………………19
2.5.1.1 Grazing…………………………………………………………... 19
2.5.1.2 Cultivation……………………….…………………………. 22
2.5.1.3 Fertilization…………………………………………………23
2.5.1.4 Herbicide Application…………………………………………24
2.5.1.5 Industrial Disturbance……………….………………………………24
2.5.2 Natural Disturbance………………………………………………………. 25
2.5.3 Abiotic Influences………………………………………………………27
2.6 Aboveground Floristic Composition Relative to the Belowground Seed Bank…... 27
2.7 Recruitment and Community Assembly…………………………………………...29
2.8 Weed Seed Banks and Invasion…………………………………………………34

2.9 Legumes and Legume Seed Banks ... 35
2.9.1 Agronomic Legumes ... 37
2.9.2 Escaped (Invasive) Agronomic Legumes ... 38
2.9.3 Native Legumes ... 40
2.10 Characterization of Seed Banks ... 41
2.10.1 Sampling Design and Subsample Size ... 42
2.10.2 Germinable Seed Bank Assessments (and Their Limitations) ... 43
2.11 Existing Research Gaps in Seed Bank Knowledge ... 44
2.12 Conclusion ... 46
2.13 Literature Cited ... 47

STUDY I

CHAPTER 3. Pasture Management History and Corresponding Range Health in Alberta's Parkland ... 58

3.1 Abstract ... 58
3.2 Introduction ... 59
3.3 Methods ... 61
3.3.1 Study Sites ... 61
3.3.2 Producer Survey ... 63
3.3.3 Rangeland Health ... 64
3.3.4 Stocking Rate ... 65
3.4 Statistical Analysis ... 66
3.5 Results ... 66
3.5.1 Management Activities from Producer Surveys ... 66
3.5.1.1 Land Ownership ... 66
3.5.1.2 Cultivation and Seeding History ... 67
3.5.1.3 Grazing Management ... 69
3.5.1.4 Herbicide Use and Weed Control ... 70
3.5.1.5 Nutrient Addition and Other Management Actions ... 71
3.5.1.6 Fire ... 71
3.5.1.7 Other Disturbances ... 72
3.5.2 Rangeland Health Summary ... 72
3.5.3 Management Regimes ... 73
3.6 Discussion ... 74
3.7 Conclusions and Management Implications ... 84
3.8 Literature Cited ... 85

CHAPTER 4. Using Producer Surveys to Link Pasture Management with Vegetation Composition, Soil Properties and Rangeland Health ... 110

4.1 Abstract ... 110
4.2 Introduction ... 112
4.3 Methods ... 117
4.3.1 Study Site Selection ... 118
4.3.2 Determining Producer Management ... 119
4.3.3 Plant Community and Rangeland Health Assessment ... 119
4.3.4 Soil Sampling and Properties ... 121
4.4 Statistical Analysis ... 122

4.4.1 Plant Community and Soils…………………………………………… 122
4.4.2 Rangeland Health……………………………………………………124
4.5 Results……………………………………………………………………...125
4.5.1 Plant Community…………………………………………………… 125
4.5.2 Soil Properties and Microsite Characteristics……………………… 129
4.5.3 Rangeland Health Response to Management……………………….. 131
4.6 Discussion…………………………………………………………………133
4.6.1 Management and Disturbance Legacies…………………………….. 133
4.6.2 Rangeland Health and Management…………………………………144
4.6.3 Reflection on Methods………………………………………………146
4.7 Conclusions and Management Implications……………………………… 148
4.8 Literature Cited…………………………………………………………… 149

CHAPTER 5. Linking Seed Bank Composition and Legume Recovery Potential in Pastures to Management History and Site Conditions………………………………………………………180

5.1 Abstract…………………………………………………………………… 180
5.2 Introduction……………………………………………………………….180
5.3 Methods…………………………………………………………………… 184
5.3.1 Study Site Selection and Vegetation/Soil Assessment…………………… 184
5.3.2 Determining Producer Management……………………………………185
5.3.3 Plant Community, Rangeland Health, and Soil Properties………………..186
5.3.4 Seed Bank Sampling…………………………………………………187
5.3.5 Characterising the Germinable Seed Bank……………………………..188
5.4 Statistical Analysis…………………………………………………………190
5.4.1 Linking Management to Seed Bank Characteristics………………………190
5.4.2 Seed Bank Relationship to the Plant Community and Environment……...192
5.4.3 Rangeland Health Assessment Criteria………………………………... 192
5.4.4 Stockpiled Manure……………………………………………………...192
5.5 Results……………………………………………………………………...193
5.5.1 Seed Bank Composition…………………………………………….. 193
5.5.2 Seed Bank Responses to Management……………………………… 193
5.5.3 Ownership……………………………………………………………194
5.5.4 Cultivation…………………………………………………………... 195
5.5.5 Grazing History……………………………………………………... 196
5.5.6 Feeding Hay on Pasture……………………………………………... 197
5.5.7 Herbicide…………………………………………………………… 197
5.5.8 Fertilizer and Manure……………………………………………….. 198
5.5.9 Mechanical Pasture Maintenance: Harrowing, Aeration, Swathing/Mowing... …………………………………………………………………… 199
5.5.10 Fire…………………………………………………………………...200
5.5.11 Similarity between Seed Banks and Plant Communities………………… 201
5.5.12 Seed Bank Characteristics, Ground Cover and Soil Properties………… 202
5.6 Discussion………………………………………………………………… 203
5.6.1 Cultivation History………………………………………………….. 204
5.6.2 Grazing Management……………………………………………….. 205
5.6.3 Fertilizer and Manure……………………………………………….. 208
5.6.4 Pasture Maintenance: Harrowing, Aeration, Swathing/Mowing………… 209
5.6.5 Herbicide and Noxious Weeds…………………………………………… 211
5.6.6 Fire………………………………………………………………….. 212
5.6.7 Rangeland Health…………………………………………………… 214

5.6.8 Relationship between Seed Banks, Soils and Ground Cover…………….. 214
5.6.9 Implications for the Management of Legume Seed Banks………………. 216
5.6.10 Similarity………………………………………………………………. 216
5.6.11 Limitations of the Study…………………………………………….. 218
5.7 Conclusions and Management Implications………………………………………219
5.8 Literature Cited……………………………………………………………………221

STUDY II

CHAPTER 6. Understand pipeline disturbance impacts on seed bank composition and displacement of Mixedgrass Prairie……………………………………………………………253

6.1 Abstract……………………………………………………………………………253
6.2 Introduction………………………………………………………………………...253
6.3 Methods…………………………………………………………………………… 260
6.3.1 Study Area and Site Selection…………………………………………. 260
6.3.2 Plant Community and Seed Bank Survey………………………………261
6.3.3 Soil Properties……………………………………………………………..262
6.3.4 Characterizing the Germinable Seed Bank……………………………… 263
6.3.5 Pipeline Characteristics and Reclamation……………………………... 263
6.4 Statistical Analysis…………………………………………………………………264
6.5 Results………………………………………………………………………….. 267
6.5.1 Aboveground Plant Composition……………………………………… 268
6.5.2 Seed Bank Composition……………………………………………….. 272
6.5.3 Biological Soil Crust Composition…………………………………… 277
6.5.4 Soil Properties……………………………………………………………..281
6.6 Discussion……………………………………………………………………………267
6.6.1 Legacy Effects of Pipelines on Soil………………………………………282
6.6.2 Plant Communities…………………………………………………….. 284
6.6.3 Seed Banks………………………………………………………………. 287
6.6.4 Biological Soil Crusts…………………………………………………….292
6.6.5 Dynamics Between Plant Communities, Seed Bank, and Biological Soil Crusts ……………………………………………………………………………… 295
6.6.6 Comments on Reclamation and Revegetation……………………………297
6.6.7 Further Research…………………………………………………………..299
6.7 Conclusions and Management Implications………………………………………302
6.8 Literature Cited……………………………………………………………………303

STUDY III

CHAPTER 7. Recruitment potential of agronomic, escaped-agronomic, and native legumes from an artificial seed bank…………………………………………………………...………………357

7.1 Abstract……………………………………………………………………………357
7.2 Introduction………………………………………………………………………...357
7.3 Methods…………………………………………………………………………… 361
7.3.1 Study Locations…………………………………………………………... 361
7.3.2 Legume Species…………………………………………………………... 362
7.3.3 Germination Trial………………………………………………………… 362

7.3.4 Field Experiment Seeding Legumes into Pasture………………………… 363
7.3.4.1 Treatments………………………………………………….. 363
7.3.4.2 Characterization of Microsite Conditions and Plat Competition…364
7.3.5 Formulas and Definitions……………………………………………… 365
7.4 Statistical Analysis………………………………………………………………366
7.5 Results……………………………………………………………………...368
7.5.1 Germination Trial…………………………………………………… 368
7.5.2 Field Experiment……………………………………………………. 368
7.5.2.1 Microsite and Overstory Plant Community Competition………...368
7.5.2.2 Dry Mixedgrass Native Site (DMG-N)………………………….. 370
7.5.2.3 Dry Mixedgrass Tame Site (DMG-T)…………………………… 372
7.5.2.4 Central Parkland Native Site (CP-N)……………………………. 374
7.5.2.5 Central Parkland Tame Site (CP-T)………………………………375
7.6 Discussion…………………………………………………………………….377
7.6.1 Germination Trial…………………………………………………… 377
7.7 Conclusions and Management Implications………………………………………387
7.8 Literature Cited……………………………………………………………………389

CHAPTER 8. Synthesis and Management Recommendations………………………………………422

8.1 Key Results…………………………………………………..………………422
8.2 Implications…….……………………………………………………………..424
8.3 Future Research………………………………………………………...……… 425
8.4 Conclusions……………………………………………………………………..427
8.5 Literature Cited…………………………………………………………………428

R Packages Used………………………………………………………………………..….……429

Bibliography……………………………………………………………………………….... 430

APPENDICIES…………………………………………………………………………..……...456

Appendix A. Chapter 3………………………………………………………………….. 456
A.1 Producer Management Survey Form 2013…………………………………… 456
A.2 Simplified Tame Pasture Assessment Form Used During the Pasture Field Survey
………………………………………………………………………………….. 458
A.3 Treatment Table for Chapters 3 to 5………………………………………… 461
Appendix B. Chapter 4………………………………………………………………….. 462
B.1 Plant Community…………………………………………………………... 462
B.1.1 Summary Tables for Plant NMDS Ordinations…………………… 463
B.1.2 Indicator Species Analysis………………………………………467
B.2 Rangeland Health…………………………………………………………...473
B.2.1 Summary Tables for Plant NMDS Ordinations…………………… 473
B.3 Defining Plant Communities……………………………………………… 475
B.4 Characteristics of Plant Communities and Soil under Rangeland Health Parameters.
………………………………………………………………………………….. 479
B.4.1 Results and Brief Discussion……………………………………… 479
B.5 Soil Texture………………………………………………………………………499
Appendix C. Chapter 5………………………………………………………………… 501
C.1 Seed Bank Heterogeneity………………………………………………… 501
C.2 Summary Results of the NMDS of Seed Banks for All Pastures…………… 505

C.3 Summary Tables for the NMDS of Seed Banks for Cultivated Pastures…….. 508
C.4 Summary of CCA………………………………………………………….. 510
C.5 Seed Bank of Stockpiled Manure…………………………………………… 515
C.6 Complete ISA Table……………………………………………………….. 517
C.7 Seed Bank Relationship to Rangeland Health………………………………... 521
Appendix D. Chapter 6……………………………………………………………….. 530
Appendix E. Chapter 7……………………………………………………………….552

Abbreviations

AB	Alberta
ANOVA	Analysis of variance
AUM	Animal unit month: the amount of forage required to feed a 453.592 kg animal for one month
C	Carbon
ºC	Celsius
CP	Central Parkland
CSR	CSR theory: competitive (C), stress tolerant (S), and ruderal species (R)
D	Defoliation
+D	Defoliated: Plant biomass defoliated by clipping
-D	Not Defoliated: Plant biomass not removed by clipping
DMG	Dry Mixedgrass
EC	Electrical conductivity (μS/m)
ha	Hectare
HILF	High-Intensity Low-Frequency
IDH	Intermediate disturbance hypothesis
ISA	Indicator species analysis
L	Litter
+L	Not raked: Plant community's soil is covered by ambient litter
-L	Raked: Plant community's ambient litter removed by raking the soil surface
N	Nitrogen
NMDS	Nonlinear multidimensional scaling
OM	Organic matter
perMANOVA	Permutational multivariate analysis of variance
RHA	Rangeland Health Assessment
SE	Standard error
spp	Species
Tg	Teragram (1,000,000,000 g)
Trmt.	Treatment

List of Figures

Figure i. Schematic of multivariate data sets examined in the pasture management and seed bank survey summarized for Study I in Chapters 3 through 5. Directional influences of management and the abiotic environment on rangeland health, plant communities, and seed banks are emphasized with arrows.…….57

Figure 3.1. Lower map indicates the province of Alberta's natural subregions, and sampling locations. Upper map identifies the Edmonton, AB, metropolitan area with cities, towns, and counties outlined. Sampled pastures (black circles) are located within the Parkland, Sturgeon, Strathcona, and Leduc County; with 6 pastures located in the Blackfoot grazing reserve south of Elk Island National Park…………..…90

Figure 3.2. Monthly precipitation (mm) at the University of Alberta's South Campus, Edmonton, Alberta in 2012 and 2013 (Alberta Agriculture and Forestry 2016)………………...…………………………91

Figure 3.3. Distribution of the total size of the allotment from which each pasture sampled, further stratified by county. Pastures within the boundary of the city of Edmonton or the Blackfoot Grazing Reserve were included in the category 'other'. Pastures associated with areas <80 acres in size include acreages and smaller hobby farms…………………………………………..………………...……………92

Figure 3.4. Summary of the number of years pastures in North Central Alberta had been farmed by the current family or land manager, and the number of years since last known cultivation…………………93

Figure 3.5. Summary of known seeding history for pastures following cultivation where producers were able to recall or estimate the seed mixture (N=65/102). 'Grasses', 'legumes', and 'forage mixture' were generic descriptions of species provided by managers. Totals of similar genera were also grouped together (i.e. brome, clover, and fescue)…………………………………………………………………...94

Figure 3.6. Summary of the grazing systems used based on the survey results. Continuous systems included pastures grazed only during the growing season and pastures in which animals were present year-round. Rotational systems included pastures where animals were rotated during the growing season, and pastures grazed only in winter. When no livestock are present, grasslands had been abandoned for multiple years (~10 or more); in at least one case the abandoned pasture was swathed……………………….……95

Figure 3.7. Summary of the identity of herbivores grazed in the pastures surveyed in north central Alberta pastures………………………………………………………………………………….…………96

Figure 3.8. Summary of inferred grazing intensities for pastures as determined from the rangeland health assessment based on the observed utilization levels, soil compaction, productivity, species composition, etc. Abbreviations for intensities are as follows: U = not grazed, L = low, LM = low-moderate, M = moderate, MH = moderate-high, H = high……………………………………………………………..97

Figure 3.9. Occurrence frequency of noxious weeds detected during RHAs (grey) and the proportion of pastures where specific noxious weeds were targeted for removal with herbicide (black) in the last 3 years. Note that field scabious (*Knautia arvensis*), orange hawkweed (*Hieracium aurnatiacum*), and spotted knapweed (*Centaurea maculosa*) are currently classified as prohibited noxious and require control measures by law……………………………………………………………………………..……….98

Figure 3.10. Summary of total scores from the rangeland health assessment. Scores are represented as proportions of maximum and ranked from highest (healthiest) to lowest (unhealthy). Scores a further summarized in Table 3.5……………………………………………………………………………..99

Figure 3.11. Multiple Correspondence Analysis (MCA) ordinations of categorical data representing current pasture management practices (distance=Eigen, dimensions=3). The first 3 axes describe 31.4 % of variation in management. Long-term historical management actions (i.e. fire and cultivation) were excluded, as were non-significant responses (P < 0.01). SR = livestock stocking rate; RHA = range health assessment scores. Variable responses summarized in Tables 3.6 and 3.7……………………………..100

Figure 3.12. Cluster dendrogram of the first 3 MCA axes (distance=Euclidean, clustering method=ward) depicting the hierarchical breakdown of all management factors. Factors in closer proximity within lower levels are more likely to co-occur with one another……………………………………………………101

Figure 4.1. W-shaped transect used for vegetation and soil of vegetation in each pasture. Each segment of the 'W' is 65 m, totaling 260 m. Black squares represent points where foliar cover was measured, every 32.5 m, using a 50 cm x 50 cm quadrat…………………………………………………………..158

Figure 4.2. Median foliar cover (% ± IQR) of various functional plant groups present within parkland pastures of north central Alberta, Canada (Kruskal-Wallis, X^2 = 649.82, df=9, P < 0.001). Component medians with different letters differ at P < 0.05 following Bonferroni correction………………………159

Figure 4.3. NMDS ordination of pasture plant community composition (distance = Bray-Curtis, dimensions = 2, stress = 0.23). Centroids of management factors (bolded) and the vectors for soil properties, RHA scores (total and categorical), plant functional group cover, and other indices (blue text) plotted were all significant at P < 0.05, while listed plant species (grey text) were significant at P < 0.01. Vectors 'Forage Cover', 'Forage Spp. Shift', 'Woody Cover', and 'Wood Density' were derived from RHA scores, the vector indicates sites with high scores…………………………………………………………………160

Figure 6.1. Map of the 18 study site locations (green stars) within the 5,200 ha Mattheis Research Ranch situated 40 km north of Brooks, Alberta on Highway 36, a component of the Rangeland Research Institute affiliated with the University of Alberta…………………………………………………………………161

Figure 4.5. NMDS ordination of rangeland health scores (distance = Euclidean, dimensions = 2, stress = 0.14). A). Symbols of sites show significant management factors (P < 0.05) in the ordination (identified with *envfit::vegan* in R software). Larger symbols indicate higher grazing intensities, smaller symbols indicate a lower grazing intensity. Dark grey symbols indicate sites that were identified as recently burned in the interview, light grey represents no report of fire. A solid circle indicates a pasture where animals were being fed hay, an open symbol indicates no feed, and a cross represents sites surveyed in 2012 where these data were not collected. Centroids of the management factors were also plotted. B) Site scores and symbols are the same as on panel A, with rangeland health categories shown in black text. Note that their position indicates sites that had high scores in each category and should be interpreted as 'healthy' in that category. Significant gradients are plotted in blue, including total RHA score, basal properties of plots (basal bare ground, manure, and vegetation cover), litter depth, plant community (PC) characteristics and seed bank (SB) characteristics for which only noxious weed seed density responded (identified with *envfit::vegan*, P < 0.05). Soil properties were also fit to the ordination and no significant gradients found. Summary of significant management, plant community, seed bank, and environmental variables are in Appendix 4.2………………………………………………………………………………………..162

Figure 5.1. W-shaped transect used for sampling of vegetation and seed bank in each pasture. Each segment of the 'W' is 65 m, totaling 260 m. Black squares represent points where foliar cover was measured, every 32.5 m, using a 50 cm x 50 cm quadrat. Soil cores for seed bank assessment were sampled every 5 m (n = 53)………………………………………………………………………………………………..229

Figure 5.2. Median seed density (seeds/m^2 ± IQR) of various functional plant groups present within northern temperate pastures of north central Alberta, Canada (X^2 = 581.9, df = 9, P < 0.001)…………230

Figure 5.3. Resulting NMDS ordination of seed bank composition (distance = Bray-Curtis, dimensions = 2, stress = 0.31) collected from 102 pastures across north central Alberta during 2012 and 2013. Centroids of all management factors (bolded), plant species (grey text), as well as vectors for soil properties, RHA scores, functional group seed density, and various vegetation indices (blue text) plotted were significant at P < 0.05. Vectors 'Woody Cover' and 'Forage Spp. Shift' were derived from RHA scores. Longer vectors indicate sites with higher scores for the attributes. Significance tests are located in Tables C.2.1 to C.2.3. Significant management factors (Table C.2.1) are decomposed in Fig. 5.4……………………………..231

Figure 5.4. Resulting NMDS ordination of seed bank composition (distance = Bray-Curtis, dimensions = 2, stress = 0.31) using the same scores from Fig 5.3 and demonstrating the relationship between significant management factors (centroids) and their indicator plant species in the seed bank……………………..232

Figure 5.5. Resulting NMDS ordination of seed bank composition from pastures (n=71) where the approximate date of last cultivation was known (distance = Bray-Curtis, dimensions = 2, stress = 0.30). Panels A and B use the same scores, with panel A showing the relationship of site with significant management factors (P < 0.05), and panel B shows the relationship between pasture age and the abundance of various seed bank vegetation groupings……………………………………………………………233

Figure 5.6. Results of a canonical correspondence analysis (CCA) of seed bank composition constrained by overlying plant community cover across 102 pastures of north central Alberta sampled in each of 2012 and 2013 [distance = Euclidean, axes = 27 (only the first 2 displayed in figure)]. The CCA model explained 56.0% of the variance in seed bank composition, with 17/27 axes significant. For simplification only the first two axes explaining nearly half the variance (24.8 %) are displayed. Plant community variables were selected using a step-wise permutational process. Management factors, and seed bank characteristics and species displayed were all significant at P < 0.05, while all plant community vectors included in the CCA model are displayed……………………………………………………………………………..234

Figure 5.7. Relationships between the seed densities (seeds/m^2) of total forbs and total graminoids in pastures and measured rangeland health (RH) scores from field assessments analyzed with Poisson regression. Graminoids: log(y) = 0.0143 + 6.38, P < 0.001; Forbs: log(y) = -0.0103x + 9.13, P < 0.001….235

Figure 5.8. Relationship between similarity (Sørenson Index) of the seed bank and overlying plant community, and associated range health scores determined from the field assessment (R^2 = 0.03; P = 0.043; 95% CI). The relationship between similarity and RH is: y = -0.0014x + 0.4859………………………236

Figure 5.9. Summary table depicting correlations (*r*) in the density of various plant functional groups present in the seed bank of 102 pastures in north central Alberta, and total seed bank density as well as indices of seed bank similarity, diversity, evenness and richness. Only significant correlations are reported, blank cells had no significant relationships……………………………………………………………237

Figure 5.10. Correlations (*r*) of the relationship between seed density and various soil properties and basal cover characteristics in the above-ground vegetation. Only significant correlations are reported (P < 0.05); blank cells had no significant relationships……………………………………………………………238

Figure 5.11. Correlations (*r*) of seed bank density and various above-ground plant community metrics, including various cover groupings and diversity indices. Only significant correlations are reported (P < 0.05), blank cells had no significant relationships……………………………………………………….239

Figure 6.1. Map of the 18 study site locations (green stars) within the 5,200 ha Mattheis Research Ranch situated 40 km north of Brooks, Alberta on Highway 36, a component of the Rangeland Research Institute affiliated with the University of Alberta…………………………………………………..314

Figure 6.2. Monthly observed precipitation (mm) at the University of Alberta Mattheis Research Ranch between January of 2013 and December of 2015, inclusive (Alberta Agriculture and Forestry, 2016). Data were acquired from the Verger Monitoring Station, located NW of the Mattheis Ranch. The 30-year average for the area was obtained from readings taken at T22 – R14 – W4……………………………………315

Figure 6.3. Seed bank sampling intensity along 55 m long transects placed perpendicular to the pipeline trench. Soil seed bank samples were drawn at the following distances: pipeline center, edge of soil trench, and 0.5, 1, 2, 3, 5, 7.5, 10, 15, 20, 25, 35, 45 and 55 m. Plant community foliar cover and basal cover (litter, exposed soil, etc.) were measured at the same points with a 0.25 m^2 quadrat with one exception; no cover was estimated at the pipeline trench edge as it would simultaneously describe cover on the trench and in the adjacent plant community……………………………………………………………………316

Figure 6.4. NMDS ordination biplots of plant community composition (stress = 0.23, dimensions = 2, distance = Bray-Curtis), including A) species with significant correlations to the axes (P < 0.001), and B) overlaid vectors of significant soil conditions, ground cover, and plant community cover attributes (P < 0.05). Larger symbols indicate greater introduced species cover, and darker coloured symbols indicate plant communities (i.e. plots) closer to pipeline disturbance, while lighter colours indicate plant communities further away that were likely undisturbed…………………………………………………………..317

Figure 6.5. NMDS ordination biplot of **p**lant community species biomass assessed at the species level (stress = 0.22, dimensions = 2, distance = Bray-Curtis), including A) species with significant correlations to the axes (P < 0.001), and B) overlaid vectors of significant soil conditions, ground cover and vegetation biomass attributes (i.e. richness, diversity, etc.) (P < 0.05). Larger symbols indicate greater introduced species biomass, and darker coloured symbols indicate plant communities (i.e. plots) closer to pipeline disturbance, while lighter colours indicate plant communities further away that were likely undisturbed……………………………………………………………………………………….318

Figure 6.6. NMDS ordination biplot of seed bank composition (stress = 0.28, dimensions = 2, distance = Bray-Curtis), including A) species in the seed bank with significant correlations to the axes (P < 0.001), and B) overlaid vectors of significant soil conditions, ground cover, and seed bank community attributes (P < 0.05). Larger symbols indicate greater introduced species seed density, and darker coloured symbols indicate seed bank communities (i.e. plots) closer to pipeline disturbance, while lighter colours indicate communities further away that were likely undisturbed…………………………………………………319

Figure 6.7. Poisson regressions of seed densities (seeds/m^2) for various plant lifeforms at increasing distance (m) from pipeline trench…………………………………………………………………..320

Figure 6.8. Poisson regressions of seed densities (seeds/m^2) for plant lifeforms in relation to various pipeline ages (years)……………………………………………………………………………...321

Figure 6.9. Linear regressions for indices of seed bank richness, Shannon's diversity, beta diversity, Pielou's evenness and Sorensen's similarity to the aboveground plant community, along increasing distances from pipeline disturbance, with 95% confidence intervals. Significant relationships have bolded P-values……………………………………………………………………………………………..322

Figure 6.10. Pipeline impacts on biological soil crust cover within the proximity of disturbed trenches. Median crust cover significantly differed at all sampling distances (X^2 = 16.69, P = 0.002), each median is accompanied by the IQR range in boxplots. The smooth blue line represents the non-linear function for soil

crust cover changes over distance (Θ_1 = 10.56 ± 1.19, t = 8.849, P < 0.001; Θ_2 = 2.57 ± 1.41, t = 1.823, P = 0.069; Θ_3 = -0.71 ± 0.34, t = -2.065, P = 0.040; R^2 = 0.084)…………………………………………323

Figure 6.11. NMDS ordination biplot of biological soil crust composition (stress = 0.13, dimensions = 2, distance = Bray-Curtis) including lichens, mosses, and spike-mosses, and metrics associated with the resulting axes. Panel A includes significant biological soil crust species (P < 0.05). Panel B includes biplots of edaphic factors and soil crust indices (P < 0.05). Panel C includes significant (P < 0.05) seed bank composition characteristics (purple) and individual species from the seed bank (grey) as vectors. Larger symbols indicate greater biological soil crust cover, darker coloured symbols indicate plots closer to the pipeline trench, while lighter colours indicate plots further away that were likely undisturbed…………324

Figure 6.12. NMDS ordination biplot of biological soil crust composition and ground cover dynamics (stress = 0.14, dimensions = 2, distance = Bray-Curtis), including the proportion of soil exposure and litter cover, and all metrics significantly related to the resulting axes. Panel A includes significant biological soil crust species (P < 0.05). Panel B includes biplots of edaphic factors and soil crust indices (P < 0.05). Panel C includes significant (P < 0.05) seed bank composition characteristics (purple) and individual species from the seed bank (grey) as vectors. Larger symbols indicate greater biological soil crust cover, darker coloured symbols indicate plots closer to the pipeline trench, while lighter colours indicate plots further away that were likely undisturbed……………………………………………………………………………..325

Figure 6.13. Poisson regressions describing the relationship between the total seed density of native (dark lines) and introduced (light lines) plant species in relation to increasing amounts of either A) surface biological soil crust, B) bare mineral soil, and C) litter cover……………………………………………326

Figure 7.1. Map of study site locations within the Central Parkland and Dry Mixedgrass natural subregions within the province of Alberta, Canada. The Central Parkland locations were located within the Roy Berg Kinsella Research Ranch near Kinsella, Alberta, 140 km SE of Edmonton. Dry Mixedgrass sites were located at the Mattheis Research Ranch near Duchess, 150 km east of Calgary. Both Research Ranches are affiliated with the University of Alberta……………………………………………………………….396

Figure 7.2. Map of the native (N) and tame (T) study site locations (white stars) within the Central Parkland (CP) and Dry Mixedgrass (DMG) at their respective research ranches…………………………………..397

Figure 7.3. Observed and average monthly precipitation (mm) at the University of Alberta's Mattheis Research Ranch in the Dry Mixedgrass (DMG) prairie and Roy Berg Kinsella Research Ranch in the Central Parkland (CP) between 2014 and 2016 (Alberta Agriculture and Forestry, 2017)………………398

Figure 7.4. Summary of mean germination (% ± 1 SE) for each of 6 legume species when seeds were glued to toothpicks or left untreated to germinate on moist filter paper. Seeds were placed in the dark at ambient room temperature. Significant differences in overall germination among species were found (P < 0.001) and are distinguished by capital letters. Gluing seeds had no effect on germination and did not interact with species (P > 0.05)……………………………………………………………………………………399

Figure 7.5. The number of individuals required to survive 1 time-step based on the survival of individual seedlings following the initial growing season for each of 6 legume species seeded into native (N) and tame (T) grasslands within the Dry Mixedgrass (DMG) and Central Parkland (CP). For species with adequate data, medians (±IQR) were compared; when individuals failed to survive from a species no median is displayed and the number of individuals required to survive 1 time-step could be interpreted as infinite……………………………………………………………………………………………400

Figure 7.6. NMDS ordination of plant community composition at the native Dry Mixedgrass (DMG-N) location in 2014 (stress = 0.27, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by

open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were significantly different (P = 0.036). Plant species with significant correlations to the axes (P < 0.05) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots (P < 0.05). Biplots of legume seedling germination, mortality (over winter 2014), height (Ht), and stage (Stg.) were included if significant (P < 0.05)……………………………………………………401

Figure 7.7. NMDS ordination of plant community composition at the tame Dry Mixedgrass (DMG-T) location in 2014 (stress = 0.21, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were not significantly different (P = 0.098) and not displayed. Plant species with significant correlations to the axes (P < 0.05) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots (P < 0.05). Biplots of legume seedling germination, mortality (over winter 2014), height (Ht), and stage (Stg.) were included if significant (P < 0.05)……………………………………402

Figure 7.8. NMDS ordination of plant community composition at the native Central Parkland (DMG-T) location in 2014 (stress = 0.26, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were significantly different (P = 0.003). Plant species with significant correlations to the axes (P < 0.05) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots (P < 0.05). Biplots of legume seedling germination (Germ.), mortality (over winter 2014) (Mort.), height (Ht), and stage (Stg.) were included if significant (P < 0.05)……………………………………403

Figure 7.9. NMDS ordination of plant community composition at the tame Central Parkland (CP-T) location in 2014 (stress = 0.23, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were not significantly different (P = 0.331) and not displayed. Plant species with significant correlations to the axes (P < 0.05) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots (P < 0.05). Biplots of legume seedling germination (Germ.), mortality (over winter 2014) (Mort.), height (Ht), and stage (Stg.) were included if significant (P < 0.05)……………………404

Figure B.3.1. Silhouette widths for plant community groups determine the relatedness of clusters when choosing the partition criteria (Borcard et al. 2011). Optimally we have 27 plant communities, but a meaningful peak of 10 groupings was chosen to simplify descriptions and reduce complexity for pastures in north central Alberta……………………………………………………………………………..475

Figure B.3.2. Summary of final cluster analysis revealing 10 different communities comprised of groupings ranging from 3 to 19 sites for pastures in north central Alberta…………………………………………476

Figure C.1.1. Mean seed abundance (±SE) found within each of 11 pastures as a function of an increasing number of soil cores during subsampling. A maximum number of 53 cores were sampled across all sites……………………………………………………………………………………………………501

Figure C.1.2. Mean seed bank species richness (±SE) found within each of 11 pastures as a function of an increasing number of soil cores. A maximum number of 53 cores were sampled across each pasture…..502

Figure C.5.1. Seed density (seeds/L ± 1 SE) of stock piled manure collected when producers confirmed that they spread manure on their pasture(s). Note that the noxious weeds category included one forb, stork's bill (*Erodium cicutarium*), and one grass, green foxtail (*Setaria viridis*)……………………………515

Figure C.5.2. NMDS ordination of the germinable seeds identified in stockpiled manure (distance = Bray-Curtis, dimensions = 2, stress = 0.04). The majority of manure piles were comprised of cattle manure (closed circles), while one pile consisted of sheep manure (open circle). All species and biplots vectors are displayed……………………………………………………………………………………………516

Figure D.1. Records of the number of successfully drilled wells in Alberta since January 2005 through to December 2016. These data are available to the public through the Alberta Energy Regulator (AER). Provincial drilling activity declined after autumn of 2014……………………………………………530

Figure D.2. NMDS ordination of aboveground plant community composition, colour coded by site (stress = 0.23, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater introduced species cover……………………………………………………………………………………………...531

Figure D.3. NMDS ordination of aboveground plant community biomass clipped by individual species, colour coded by site (stress = 0.22, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater introduced species biomass…………………………………………………………………………531

Figure D.4. NMDS ordination of seed bank composition, colour coded by site (stress = 0.28, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater introduced species seed density…………532

Figure D.5. NMDS ordination of seed bank composition along pipeline trenches (soil from pipeline center and edge) (stress = 0.24, dimensions = 2, distance = Bray-Curtis). Overlaid vectors represent significant seed bank characteristics and pipeline attributes (diameter and age) (P < 0.05). Note, no pipeline attributes were significant. Larger symbols indicate greater introduced species seed density………………………533

Figure D.6. Correlation coefficients (r) for the relationship between seed bank characteristics and both soil properties and ground cover including lichen species…………………………………………………534

Figure D.7. NMDS ordination of biological soil crust composition, colour coded by site. Panel A is biological soil crust composition (stress = 0.13, dimensions = 2, distance = Bray-Curtis), including lichens, mosses, and spike-mosses. Panel B describes dynamics in ground cover where biological crust composition was analysed with the inclusion of the proportion of bare ground and litter cover (stress = 0.14, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater biological soil crust cover………………535

Figure E.1. Sample of experimental design and plot plan from the native site within the Central Parkland, where treatments of defoliation (D) and litter removal (L) were applied in a two-way factorial. Within treatment plots, each of 6 legume species were randomly seeded in subplots (split-plot)………………552

Figure E.2. Example of defoliation (D) and litter (L) removal treatments from the native Dry Mixedgrass (DMG) prairie site. The treatments were applied as follows: A = control (+L -D) (i.e. no defoliation and no removal of litter); B = defoliation (+D), which occurred every three weeks during the growing season; C = raked to remove standing and fallen litter (-L); D = defoliated and raked to remove litter (-L +D). Treatments were applied to a 1 m x 6 m strip, each subplot for a species was 1 m x 1 m, and the seeds were seeded within a 50 cm x 50 cm area in the subplot's center……………………………………………………..553

List of Tables

Table 3.1. One-way ANOVA tests for relationships between stocking rate and density with management factors……………………………………………………………………………………………..……..102

Table 3.2. LS Mean (±SE) stocking rate and density in response to pasture management………………103

Table 3.3. Summary of herbicide products chosen including their active ingredient, herbicide group (mode of action), and systemic/residual properties…………………………………………………………...104

Table 3.4. Summary of industrial disturbances reported by landowners in 2013………………………105

Table 3.5. Summary of mean scores from the rangeland health assessments conducted on 102 pastures distributed across north-central Alberta during 2012-2013. Full details on the range health assessment used can be found in Adams et al. (2009). Also shown are the proportion of tame (n=90) and modified-tame (n=12) pastures falling in the maximum and minimum categories within a criterion. Tame pastures had a known history of cultivation and seeding, while modified-tame were not seeded and therefore comprised of a mix of native and naturalized tame species………………………………………………………..106

Table 3.6. Summary of management factors associated (i.e. correlated) with the MCA axes (Fig. 3.11). Only significant factors (P < 0.05) are listed…………………………………………………………..107

Table 3.7. Significant management factors (P < 0.05) for MCA axes (Fig. 3.11)………………………108

Table 3.8. Significant biplot vectors (P < 0.05) describing pasture age, health, and stocking rate under current management and disturbance history (Fig 3.11)…………………………………………………109

Table 4.1. Dominant plant species ranked by mean foliar cover (±SD) found across all pastures (n=102) from central Alberta during 2012 and 2013……………………………………………………………163

Table 4.2. Summary of PerMANOVA assessment of vegetation composition responses to various producer management factors taking place in northern temperate pastures sampled in 2012 and 2013. Analysis was conducted using a Bray-Curtis distance metric, and 999 permutations. Significance was set at P < 0.05, with those values meeting this level shown in bold…………………………………………………………164

Table 4.3. Significance of PerMANOVA contrasts evaluating cultivation effects on plant community composition in 102 pastures surveyed across north central Alberta during 2012 and 2013. Analysis was conducted using a Bray-Curtis distance metric, and 999 permutations. Significance was set at P < 0.05, with those values meeting this level shown in bold…………………………………………………………164

Table 4.4. Summary of indicator species analysis for plant community species association with various management factors documented on 102 pastures of north central Alberta during 2012 and 2013. Analysis was run with 999 permutations, and only species significant at P < 0.01 are shown……………………165

Table 4.5. Indicator species analysis of plant community functional group association with various pasture management factors evaluated across 102 pastures in north central Alberta during 2012 and 2013. Analysis was run with 999 permutations, and results with P < 0.1 are shown, significant results (P < 0.05) are bolded……………………………………………………………………………………………166

Table 4.6. Significant one-way ANOVA and Kruskal-Wallis tests (univariate) for effects of management factors on the abundance of various primary vegetation cover groupings documented across 102 pastures of north central Alberta during 2012 and 2013………………………………………………………..167

Table 4.7. Summary LS means (±SE) for all significant management effects on the cover of primary vegetation groups. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups……………………………………………………………168

Table 4.8. Significant ANOVA and Kruskal-Wallis tests of management factors on the cover of specific plant functional groups, as sampled across 102 pastures across north central Alberta during 2012 and 2013...169

Table 4.9. Summary of LS mean (±SE) cover values of various plant functional groups with significant treatment responses to various management factors. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups……………………….170

Table 4.10. Significant ANOVA and Kruskal-Wallis tests of management factors on plant richness, diversity, and evenness within 102 parkland pastures sampled across north central Alberta during 2012 and 2013………………………………………………………………………………………………171

Table 4.11. Summary LS mean (±SE) values of plant richness, diversity, and evenness, for pastures sampled in relation to the management factors. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups………………………………172

Table 4.12. Significant ANOVA and Kruskal-Wallis tests effects of management factors on various soil properties found across 102 pastures surveyed across north central, Alberta during 2012 and 2013……173

Table 4.13. Effect of significant management factors on the LS means (±SE) of various soil properties as sampled across 102 pastures of north central Alberta sampled during 2012 and 2013. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups……………………………………………………………………………………….....174

Table 4.14. Summary of significant ANOVA and Kruskal-Wallis tests of management factors on various ground cover characteristics found in 102 pastures of north central Alberta during 2012 and 2013……175

Table 4.15. Summary LS mean (±SE) responses of various ground cover characteristics in relation to different management factors. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups……………………………………………..176

Table 4.16. Significant ANOVA effects on total RHA score found for 102 pastures of north central Alberta during 2012 and 2013……………………………………………………………………………………177

Table 4.17. Summary of LS means (±SE) for the total RHA scores for various management factors (P < 0.05)………………………………………………………………………………………...……177

Table 4.18. Results of the PerMANOVA analysis assessing the impact of management factors on rangeland health scores. Analysis was conducted using both a Euclidean and Bray-Curtis distance metric, and 999 permutations. Significance was set at P < 0.05, with those values meeting this level shown in bold……178

Table 4.19. Indicator analysis of inversed RHA scores to detect which management actions are associated with deteriorating RHA scores. Analysis was run with 999 permutations, and results with P < 0.1 are shown, significant results (P < 0.05) are bolded……………………………………………………………….179

Table 5.1. Summary of the 10 most dominant plant species abundance (and ranks) found in the aboveground community (based on % cover ± 1SD) and the seed bank (based on seed density; seeds/m^2 ± 1SD) of pastures sampled across north central Alberta during 2012 and 2013…………………………..240

Table 5.2. Results of the perMANOVA tests evaluating seed bank composition responses to individual pasture management factors based on the assessment of 102 sample sites examined across north central Alberta during 2012 and 2013……………………………………………………………………………241

Table 5.3. Results of the perMANOVA contrasts assessing management factor impacts on pasture seed bank composition…………………………………………………………………………………...242

Table 5.4. Summary of the indicator species analysis relating seed bank species composition to each of the management factors. Only those species with significance at P < 0.01 are shown. For a more complete list of species, see Table C.6.1……………………………………………………………………………243

Table 5.5. Results of the indicator species analysis identifying those vegetation functional groups found in the seed bank that were associated with each pasture management factor assessed in the survey, as well as rangeland health measured in the field (P < 0.10)……………………………………………………..244

Table 5.6. Results of the ANOVA tests results reporting effects of various pasture management factors on the seed density (seeds/m^2) of primary plant groupings, including total SB density, based on data collected across 102 pastures in north central Alberta during 2012 and 2013…………………………………...245

Table 5.7. Comparison of mean (±SE) seed bank density (seeds/m^2) of primary plant groupings in relation to different management factors and range health criteria. Data are based on 102 pasture sampled across north central Alberta during 2012 and 2013………………………………………………………………..246

Table 5.8. Results of the ANOVA test results reporting effects of various pasture management factors on the seed density (seeds/m^2) of specified functional plant groupings, based on data collected across 102 pastures in north central Alberta during 2012 and 2013………………………………………………247

Table 5.9. Comparison of mean (±SE) seed bank density (seeds/m^2) of specified functional plant groupings in relation to different management factors and range health criteria. Data are based on 102 pastures sampled across north central Alberta during 2012 and 2013……………………………….…………248

Table 5.10. Results of the ANOVA tests reporting effects of various pasture management factors on seed bank similarity (to aboveground vegetation), richness, diversity, and evenness, based on data collected across 102 pastures in north central Alberta during 2012 and 2013……………………………………249

Table 5.11. Comparison of mean (±SE) seed bank similarity, richness and diversity in relation to different management factors and range health criteria. Data are based on 102 pastures sampled across north central Alberta during 2012 and 2013…………………………………………………………………………250

Table 5.12. Results of the perMANOVA tests evaluating seed bank composition responses to rangeland health categories based on the assessment of 102 sample sites examined across north central Alberta during 2012 and 2013.……………………………………………………………….…………………………..251

Table 5.13. Results of the perMANOVA contrasts of range health categories relationships with seed bank composition. ………..……………………………………………………………………………....252

Table 6.1. Relative rankings of the top 15 species based on mean foliar cover for vegetation and mean seed density for the seed bank (± SE) stratified by samples from the native plant community (25 m to 55 m) and pipeline trench (center). Significant differences in foliar cover and seed density between the trench and native grassland for the top 15 species were tested using generalized linear mixed models (binomial distribution for cover and Poisson for seed density)………………………………………………………..327

Table 6.2. Plant community and seed bank composition responses to various characteristics associated with pipelines installed within Mixedgrass prairie, as determined through perMANOVA (distance = Bray-Curtis, permutations = 999). Distance from pipeline, age of disturbance, and diameter of pipeline were blocked by site in the analysis. Bolded tests indicate those with P < 0.05……………………………………...328

Table 6.3. Pairwise comparisons of either plant community or seed bank composition at different sampling distances away from the pipeline using perMANOVA (distance = Bray-Curtis, permutations = 999) blocked by 18 study sites. Contrasts were focused on the center of the pipeline and furthest sampling distance (55 m), to determine the extent of variance either away from, or towards, the pipeline trench, respectively…329

Table 6.4. Aboveground plant community and seed bank compositional responses to soil properties in Mixedgrass prairie, as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m) for this analysis……………………………………………………………………330

Table 6.5. Aboveground plant community and seed bank compositional responses to ground cover and litter mass in Mixedgrass prairie, as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m)……………………………………………………………………...331

Table 6.6. Results of the indicator species analysis of aboveground vegetation cover assessed in relation to varying distances from pipeline disturbance…………………………………………………………..…..332

Table 6.7. Results of the indicator species analysis relating aboveground plant community composition with varying pipeline diameters……………………………………………………………………...333

Table 6.8. Results of the indicator species analysis of aboveground plant community composition with varying age classes of pipelines……………………………………………………………………..334

Table 6.9. Results of the indicator species analysis of aboveground plant community composition and various interactions of pipeline distance, age, and diameter……………………………………………..335

Table 6.10. ANOVA summary statistics for plant community responses to pipeline disturbance and characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect……………………………………………………………………………….337

Table 6.11. LS means (±SE) for plant community responses to pipeline disturbance and characteristics……………………………………………………………………………………….338

Table 6.12. ANOVA summary statistics for plant community dry biomass responses to pipeline disturbance characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect……………………………………………………………………………339

Table 6.13. LS means (±SE) dry biomass (kg/ha) responses to pipeline characteristics………………….340

Table 6.14. Results of the indicator species analysis of seed bank composition in relation to varying distances from pipeline disturbance………………………………………………………………………341

Table 6.15. Results of the indicator species analysis relating seed bank composition with varying pipeline diameters………………………………………………………………………………………………342

Table 6.16. Results of the indicator species analysis of seed bank composition with varying age classes of pipeline…………………………………………………………………………………...…………343

Table 6.17. Results of the indicator species analysis of seed bank composition and various interactions of pipeline distance, age, and diameter…………………………………………………………………...344

Table 6.18. LS means (±SE) for seed density (seeds/m^2) of plant groups of major life histories and origins. Significant differences in seed density among pipeline diameter classes were determined using ANOVA and Kruskal-Wallis tests. Lower case letters distinguish significant difference ($P < 0.05$)………………345

Table 6.19. ANOVA summary statistics for seed density (seeds/m^2) responses to pipeline disturbance characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect……………………………………………………………………………346

Table 6.20. LS mean (±SE) for seed density (seeds/m^2) and diversity responses to pipeline disturbance and characteristics……………………………………………………………………………………347

Table 6.21. ANOVA summary statistics of diversity indices describing seed bank responses to pipeline disturbance and characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect…………………………………………………………348

Table 6.22. Biological soil crust compositional responses to various pipeline disturbances in the Mixedgrass prairie, as determined through perMANOVA (distance = Bray-Curtis, permutations = 999). Distance from pipeline, age of disturbance, and diameters of pipeline were analysed in a perMANOVA blocked by site……………………………………………………………………………………349

Table 6.23. Pairwise comparisons of biological soil crust composition at different sampling distances away from pipelines using perMANOVA (distance = Bray-Curtis, permutations = 999), blocked by site (n=18)……………………………………………………………………………………………….350

Table 6.24. Biological soil crust indicators associated with different pipeline disturbance treatments in the Mixedgrass prairie along a subset of sampling distances observed (pipeline center trench, 1 m, 5 m, 20 m, 55 m), pipeline age classes, diameters, and interactions thereof. Only those indicators with $P < 0.10$ are shown……………………………………………………………………………………………….351

Table 6.25. Biological soil crust compositional responses (including lichen) to soil properties in Mixedgrass prairie as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m) from which soil was sampled. Lichen composition was also analysed separately………………..352

Table 6.26. Biological soil crust compositional responses to ground cover and litter mass in Mixedgrass prairie, as well as their interactions, as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m). Detailed ground cover was ocularly assessed and fallen litter was weighed……………………………………………………………………………………………..353

Table 6.27. Coefficients for mixed effects models of soil properties and their relationship to distance from the center trench, pipeline diameter and age, as well as soil texture……………………………………354

Table 6.28. ANOVA tests for legacy effects on soil properties at increasing distance from pipeline trenches where distance and ecosite texture are fixed factors……………………………………………………..355

Table 6.29. LS means (± SE) of soil properties along 18 pipelines at 5 sampling distances and among differing soil textures……………………………………………………………………………..356

Table 7.1. Summary statistics for the germination of 6 legumes species when seeds were glued to toothpicks or left untreated to germinate on moist filter paper. Seeds were in the dark at ambient room temperature. Seeds that did not germinate were classified as 'hard' or 'degraded' and were grouped for analysis…….405

Table 7.2. Summary of mean germination (% ± 1 SE) for each of 6 legume species when seeds were glued to toothpicks or left untreated to germinate on moist filter paper. Seeds were in the dark at ambient room temperature. Seeds that did not germinate were classified as 'hard' or 'degraded' and were grouped for analysis…………………………………………………………………………………………...405

Table 7.3. Summary statistics for the effect of defoliation (D) and litter removal (L) on microsite variables measured such as litter depth (cm), live vegetation height (cm), bare soil exposure (%), soil surface temperature (℃), photon flux density (PFD), and soil moisture (%), within native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions……………….406

Table 7.4. Differences in the mean (± SE) microsite characteristics in relation to defoliation (D) and litter removal (L) on microsite variables measured such as litter depth (cm), vegetation height (cm), bare soil (%), soil surface temperature (℃), photon flux density (PFD), and soil moisture (%) in native (N) and tame (T) grasslands of the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions…………..407

Table 7.5. Summary statistics for perMANOVA tests of plant community composition shifts under litter manipulation (L) and defoliation (D) treatments across all Dry Mixedgrass (DMG) and Central Parkland (CP) sites in 2014……………………………………………………………………………………408

Table 7.6. Summary statistics for plant community characteristics altered via defoliation (D) and litter removal (L) treatments at all Dry Mixedgrass (DMG) and Central Parkland (CP) sites during the summer of 2014……………………………………………………………………………………………409

Table 7.7. Summary mean (± SE) aboveground plant community characteristics altered via defoliation (D) and litter removal (L) treatments across all Dry Mixedgrass (DMG) and Central Parkland (CP) sites during summer 2014………………………………………………………………………………..……410

Table 7.8. Summary statistics for germination (Germ.) and recruitment (Rec.) of legume seedlings over 3 successive years for each of 6 legume species planted in plots with defoliation (D) and litter removal (L) treatments at native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions……………………………………………………………………………..411

Table 7.9. Summary mean (± 1 SE) germination (Germ.) and recruitment (Rec.) as a percent (%) of legume seedlings placed into the initial seed bank over 3 successive years for each of 6 legume species subset in plots that had been treated with defoliation (D), litter removal (L), or combinations thereof, during the initial year at native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions……………………………………………………………………………………..412

Table 7.10. Summary statistics for the assessment of mortality rate of 6 species (S) of legume seedlings between the first and second year for plots that were exposed to varying litter removal (L) and defoliation (D) treatments. Legumes were seeded into native (N) and tame (T) grassland in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions……………………………………………………………414

Table 7.11. Differences in the mortality rate (%) of legume seedlings between the first and second year, in combination with all combinations of litter manipulation (L) and defoliation (D) treatments. Legumes were seeded into native (N) and tame (T) grassland in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions……………………………………………………………………………..415

Table 7.12. Summary statistics for the height and growth stage of 6 legume species subset within plots that had been treated with defoliation (D) and litter removal (L) treatments during the initial year within native (N) and tame (T) grasslands of the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions……………………………………………………………………………………………..416

Table 7.13. Mean (± 1 SE) height (cm) and growth stage of 6 legume species subset in plots that had been treated with defoliation (D) and litter removal (L) during the initial year of establishment at native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions. See Appendix E (Table E.2) for definition of growth stages………………………………………………417

Table 7.14. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Dry Mixedgrass native site and their relationship to plant community characteristics……418

Table 7.15. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Dry Mixedgrass tame site and their relationship to plant community characteristics………419

Table 7.16. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Central Parkland's native fescue prairie site and their relationship to plant community characteristics……………………………………………………………………………………………420

Table 7.17. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Central Parkland's tame site and their relationship to plant community characteristics……421

Table A.1. Summary of management factors, plant community types, and rangeland health questions used to analyze plant community, seed bank, and biophysical responses……………………………………..461

Table B.1. Summary of plant community and soil attributes observed, their abbreviations, units, and transformation for univariate tests………………………………………………………………………..462

Table B.1.1.1. Summary of significant management centroids arising from the NMDS ordination of plant community composition (P < 0.1) (Figure 4.3)………………………………………………………..463

Table B.1.1.2. Significant biplot vectors for NMDS ordination of plant community composition (Figure 4.3)………………………………………………………………………………………………………464

Table B.1.1.3. Individual plant species' relationship to NMDS for plant community composition (P < 0.1) (Figure 4.3)……………………………………………………………………………………………465

Table B.1.2.1. Indicator analysis linking plant community species association with various management factors (P < 0.05)……………………………………………………………………………………467

Table B.1.2.2. Indicator plant species analysis assessing significant Rangeland Health Assessment (RHA) categories describing shifts in plant community composition (P < 0.05)………………………………470

Table B.2.1.1. Summary of significant management factor centroids arising from the NMDS ordination of rangeland health assessment scores (Figure 4.5)…………………………………………………………473

Table B.2.1.2. Summary of biplot scores from the NMDS ordination of various pasture soil and vegetation properties, including scores (Figure 4.5)…………………………………………………………….....474

Table B.3.1 Indicator species analysis and dominant species cover for each of 10 plant communities identified through a cluster analysis…………………………………………………………………...477

Table B.3.2. perMANOVA of plant community composition among natural subregions, plant community types, and defined communities through clustering……………………………………………………478

Table B.3.3. Plant community indicator species for the natural regions in central Alberta………………478

Table B.4.1. PerMANOVA of plant community composition responses to individual rangeland health metric categories in north central Alberta……………………………………………………………..483

Table B.4.2. PerMANOVA contrasts of management factors and rangeland health assessment categories affecting plant community composition in north central Alberta…………………………..…………..484

Table B.4.3. Indicator species analysis of plant community functional group association with rangeland health metrics. Results with P < 0.1 are shown, significant results (P < 0.05) are bolded………………485

Table B.4.4. Significant effects of rangeland health metrics on the abundance of various primary vegetation cover groupings……………………………………………………………………………………..486

Table B.4.5. Summary LS means (±SE) for all significant management effects on the cover of primary vegetation groups. Within a column and management factor, means with different letters differ, P < 0.05 after Bonferroni correction………………………………………………………………………………487

Table B.4.6. Significant effects of rangeland health metrics on the cover of specific plant functional groups in north central Alberta……………………………………………………………………………..488

Table B.4.7. Summary of LS mean (±SE) cover values of various plant functional groups with significant responses to various management factors……………………………………………………………..489

Table B.4.8. Significant effects of rangeland health on plant richness, diversity, and evenness within parkland pastures of north central Alberta…………………………………………………………………490

Table B.4.9. Summary LS mean (±SE) values of plant richness, diversity, and evenness, for pastures sampled in relation to the management factors……………………………………………………………..491

Table B.4.10. Significant effects of rangeland health metrics on various soil properties found across parkland pastures in north central Alberta…………………………………………………………………492

Table B.4.11. Effect of significant rangeland health metric on the LS means (±SE) of various soil properties as sampled across parkland pastures of north central Alberta………………………………………..…….493

Table B.4.12. Summary of significant effects of rangeland health metric on various ground cover characteristics in parkland pastures of north central Alberta……………………………………………..494

Table B.4.13. Effect of significant rangeland health metric on the LS means (±SE) on ground cover variables sampled across parkland pastures of north central Alberta……………………………………495

Table B.4.14. Significant ANOVA effects on total rangeland health assesment score of north central Alberta pastures……………………………………………………………………………………..…496

Table B.4.15. Summary of LS means (±SE) for the total rangeland health assesment scores for various management factors (P < 0.05) in north central Alberta pastures ……………………………….………497

Table B.4.16. Results of a multiple stepwise regression of total rangeland health scores against various plant community cover characteristics for north central Alberta pastures………………………………498

Table B.4.17. Coefficients for a multiple linear regression of total rangeland health score explained by plant community cover characteristics and soil properties for north central Alberta pastures…………………498

Table B.5.1. Results of the ANOVA analysis assessing the impact of management factors on observed levels of soil texture across 102 pastures in north central Alberta………………………………………499

Table B.5.2. Summary of LS mean (±SE) soil texture values in relative to various management factors for north central Alberta Pastures……………………………………………………………………………500

Table C.1.1. Results of the perMANOVA examining differences in seed bank composition among topographical features…………………………………………………………………………………503

Table C.1.2. Results of the perMANOVA contrasts examining seed bank composition among topographic positions………………………………………………………………………………………………503

Table C.1.3. Indicator species associated with the seed bank among various topographic positions….504

Table C.2.1. Summary of significant management centroids from the NMDS of seed bank composition in Figure 5.3 (P < 0.10)…………………………………………………………………………...505

Table C.2.2. Biplot vectors associated with the final NMDS ordination of seedbank composition. Data were collected from 102 pastures across north central Alberta during 2012 and 2013 (Fig 5.3)………..506

Table C.2.3. Relationship of seed bank species' abundance to the NMDS (Fig. 5.3) axes (P < 0.10)….507

Table C.3.1. Summary of significant management centroids for the NMDS ordination (Fig. 5.5) of seed bank composition and pasture age (P < 0.10)……………………………………………………………508

Table C.3.2. Significant biplot vectors for various pasture characteristics based on the NMDS (Fig. 5.5) of seedbank composition for cultivated pastures…………………………………………………………509

Table C.4.1. Axes included in reduced CCA model (Fig. 5.6) of seed bank constrained by plant community………………………………………………………………………………………………510

Table C.4.2. Plant community variables permutationally selected for the CCA (Fig. 5.6) model (seed bank composition constrained by plant community cover)…………………………………………………511

Table C.4.3. Species vectors for CCA (Fig. 5.6) of seedbank constrained by plant community. Seed bank vectors included at P < 0.1, while all plant community variables used in constrained ordination were included……………………………………………………………………………………………..512

Table C.4.4. Summary of significant management centroids for CCA (Fig. 5.6) of seed bank constrained by plant community (P < 0.10)…………………………………………………………………………513

Table C.4.5. Significant biplot vectors for CCA (Fig. 5.6) of seedbank composition…………………514

Table C.6.1. Indicator species analysis of seed bank's species association with management factors (P < 0.05) in north central Alberta's pastures…………………………………………………………………517

Table C.6.2. Indicator species analysis for the rangeland health assessment (RHA) categories describing shifts in seed bank composition (P < 0.05) in north central Alberta's pastures…………………………..519

Table C.7.1. Results of the perMANOVA tests evaluating seed bank composition responses to rangeland health assessment factors based on the assessment of 102 sample sites examined across north central Alberta during 2012 and 2013…………………………………………………………………………………524

Table C.7.2. Results of the perMANOVA contrasts assessing the influence of rangeland health scores on pasture seed bank composition……………………………………………………………………...524

Table C.7.3. Indicator species analysis of the seed bank's functional plant group in response to various management factors and rangeland health (P < 0.10)…………………………………………………525

Table C.7.4. Significant ANOVA effects of rangeland health criteria on the total seed density (seeds/m^2) of various major plant groups……………………………………………………………………………526

Table C.7.5. Mean (±SE) responses in total seed density (seeds/m^2) of the major plant groupings in relation to various rangeland health criteria………………………………………………………………………526

Table C.7.6. Significant relationships between seed density (seeds/m^2) of specified plant functional groups and rangeland health scores…………………………………………………………………………527

Table C.7.7. Mean (±SE) seed density (seeds/m^2) of various plant functional groupings in relation to rangeland health scores……………………………………………………………………………..528

Table C.7.8. Significant effects of various rangeland health criteria on similarity, as well as seed bank richness, diversity and evenness…………………………………………………………………………529

Table C.7.9. Effect of management on the mean (±SE) richness, similarity, diversity, and evenness of the seed bank……………………………………………………………………………………………529

Table D.1. Descriptions of pipelines surveyed at the Mattheis Research Ranch between 2013 and 2015..536

Table D.2. Biplot vector scores obtained from the NMDS ordination of plant community composition, for various pipeline, plant community, ground cover, and soil properties. Factors significant at P < 0.05 were included in the ordination……………………………………………………………………………..537

Table D.3. Biplot vector scores of significant plant species associated with the NMDS ordination of plant community composition. Only species significant at P < 0.001 were plotted……………………………538

Table D.4. Biplot vector scores obtained from the NMDS ordination of individual plant species biomass (clipped by species), in relation to pipeline, plant community, ground cover, and soil properties, across 18 pipeline study sites. Factors significant at P < 0.05 were included in the ordination……………………540

Table D.5. Biplot vector scores associated with the NMDS ordination of individual plant biomass (clipped by species) along 18 pipeline study sites. Only species significant at P < 0.05 were plotted……………541

Table D.6. Biplot vector scores associated with the NMDS ordination of seed bank composition, in relation to pipeline, seed bank, plant community, ground cover, and soil properties, for 18 pipeline study sites. Factors significant at P < 0.05 were included in the ordination…………………………………………543

Table D.7. Biplot vector scores associated with the NMDS ordination of seed bank composition, from 18 pipeline study sites. Only species significant at P < 0.05 were plotted…………………………………..544

Table D.8. Summary of all biological soil crust species (lichens, mosses, and others) recorded during the 2015 survey, ranked by average relative cover (%)………………………………………………………546

Table D.9. Linear regressions describing the chrono-sequence of biological crust recovery along pipelines stratified by sampling distance……………………………………………………………………...547

Table D.10. Biplot vector scores associated with the NMDS ordination of biological soil crust composition, including pipeline, biological soil crust, plant community, seed bank, ground cover, and soil properties, across 18 pipeline study sites. Factors significant at P < 0.05 were included in the ordination………….548

Table D.11. Biplot vector scores for significant species associated with the NMDS ordination of biological soil crust composition, based on 18 pipeline study sites. Only species significant at P < 0.05 were plotted……………………………………………………………………………………………549

Table D.12. Biplot vector scores associated with the NMDS ordination of biological soil crust composition, including the proportion of soil exposure and litter cover. Factors significant at P < 0.05 were included in the ordination……………………………………………………………………………………..550

Table D.13. Biplot vector scores for significant species related to the axes of the NMDS ordination for biological soil crust composition, including the proportion of soil exposure and litter cover along 18 pipeline study sites. Only species significant at P < 0.05 were plotted……………………………………………551

Table E.1. Summary of the ecosite and soil characteristics for each of the Dry Mixedgrass (DMG) prairie and Central Parkland (CP) native (N) and tame (T) grasslands used to study legume seedling demographics………………………………………………………………………………………554

Table E.2. Legume stages used to describe plant growth and development. Based on stages described for alfalfa by Fick and Mueller (1989)………………………………………………………………………555

Chapter 1

Influence of management on seed bank composition and legume recruitment in Alberta's Aspen Parkland and Mixedgrass Prairie

1.1 Background

Native grasslands are a threatened ecosystem (Samson et al. 2004) providing a suite of ecological goods and services such as wildlife habitat, biodiversity, carbon storage, nutrient cycling, forage, and pollination which also benefit society's needs for food, clean water, and air (Havstad et al. 2007). Canadian prairies are poorly protected (Gauthier and Wiken 2003), highly fragmented, and highly modified (i.e. invasion from exotic cool-season grasses or historically disturbed by attempted cultivation of forest clearing) making conservation difficult. This is evident in the north-central Alberta's Central Parkland which was historically a mosaic of fescue grassland and aspen forest, which have been highly modified by nearly ~150 years of European settlement and disruption of natural disturbances. Settlement and disturbance is similar in south eastern Alberta's Dry Mixedgrass prairies, however industrial disturbances are prevalent in the region and can have negative impacts on native grassland remnants.

Seed banks are an important component of grassland plant communities, contributing sexually produced angiosperm propagules as seedlings to the aboveground floral community following disturbance or stochastically into unoccupied niches. Species in the seed bank vary in persistence and recruitment strategies (Thompson et al. 1993), where certain disturbances like grazing can encourage certain suites of species to emerge (Kinucan and Smeins 1992; Willms and Quinton 1995). Despite the important contributions of seed banks to revegetation, maintaining biodiversity, and introducing novel species (Eschtruth and Battles 2009), the composition of seed banks has been flagrantly understudied in the Northern Great Plains, especially in Western Canadian rangelands (Clements et al. 2007; Harker et al. 2000; Johnston et al. 1969; Otfinowski et al. 2008; Ren and Bai 2016a; Ren and Bai 2016b; Ren and Bai 2007; Romo and Gross 2011; White et al. 2012; Willms and Quinton 1995).

The aboveground plant community is well understood, contributing to our understanding of ecosystem function, ecosystem classification, and forage production; while seed bank formation and composition under unique disturbance histories (Sanderson et al. 2007) and invasion (Gioria et al. 2014) is also poorly understood. Canadian rangelands were historically maintained by disturbances like grazing and fire; vegetation responses to disturbances are well understood, but seed bank responses have been overlooked (Sanderson et al. 2007). Seed bank responses are often inferred based on changes in vegetation, but rarely measured (Cox et al. 2008; Gioria et al. 2014).

Legumes are valued forage in both native and seeded grasslands, as they fix nitrogen and improve forage quantity and quality (Ledgard and Steele 1992). However, specific legumes can be sensitive to grazing and broadleaf herbicides; the potential of these species to voluntarily re-establish from the seed bank following removal is poorly understood. In addition, introduced forage legumes can exhibit invasibility through voluntary establishment and intentional introductions (Turkington et al. 1978). This research will attempt to characterize seed bank composition under divergent pasture management regimes, oil and gas disturbance, and then link germinable seed bank composition to recruitment in grasslands with a focus on legumes and other forages.

1.2 Research Objectives

The overall goal of this research is to increase our understanding of the germinable persistent soil seed bank in Alberta's rangelands (i.e. native grasslands and introduced pasture), as well as evaluate the potential for legume recovery therein. This research consists of three complimentary studies focusing on seed banks within perennial grasslands in the context of their corresponding plant communities. Two studies survey and quantify the germinable seed bank in pasture and native grassland affected by diverse disturbance histories, while a third study examines *in-situ* recruitment of legume seedlings from an artificial seed bank under simulated disturbance. All studies attempt to address general seed bank knowledge deficiencies in western Canadian perennial grasslands, while linking recruitment with management implications. More specifically, this research will: 1) quantify the abundance and composition of legumes, forages and various weeds in pastures, both in the existing pasture plant community and associated seed bank; 2) interpret seed bank composition relative to eco-site conditions and divergent management history, including disturbances such as grazing and oil/gas infrastructure; and 3) experimentally investigate the demographic processes and mechanisms regulating legume re-establishment for both native and introduced grasslands in Alberta.

1.2.1 Study #1: Linking seed bank composition and legume recovery in pastures to management history and site conditions

Legumes are an important component of pastures due to their ability to fix nitrogen (N) and reduce input costs, as well as increase forage productivity and quality, particularly crude protein (Ledgard and Steele 1992). As a result, land use management practices that reduce legume abundance are likely to reduce overall production efficiency. Where broadleaf weeds are common in northern temperate pastures, land owners are often mandated to control weeds through regulations such as the *Weed Control Act* in Alberta (Province of Alberta 2010). Herbicides can be an effective tool for reducing weeds (Grekul and Bork 2007), restoring forage production (Bork et al. 2007), and meeting local municipal guidelines for weed control. However, one undesirable side effect of herbicides is that those with the greatest efficacy on perennial weeds are also highly deleterious to legumes, eliminating them from the forage sward (Grekul and Bork 2007; Bork et al. 2007). Moreover, volunteer legume re-establishment from the existing seed bank or deliberate reintroduction by pasture over-seeding, may be negatively impacted by the soil residual properties of these herbicides. The potential for natural legume re-establishment from the seed bank is the focus of this research; seed banks of forages, forbs, and weeds will also be characterized.

In order to better understand the potential for natural legume recovery in northern temperate pastures, I designed a study to examine the seed bank composition of a large sample of pastures across central Alberta and assess the role of environment and management history (grazing and other disturbances) in altering this composition, including the associated potential for legume recovery from the soil seed bank. Producer surveys were designed to quantify current management (e.g. fertilization, bale grazing, timing of grazing, etc.) and historical disturbances (cultivation and fire), which were suspected to influence plant communities, soils, and seed bank via the reproductive potential of plants aboveground (e.g. grazing) or direct seed input (e.g. manure). Additionally, a rangeland health assessment was used to interpret the health of pastures under current management. This study addresses several deficiencies in seed bank research by examining multiple management factors within managed pastures at numerous (n=102) study site locations, which contrasts with most previous research that tends to examine select disturbance factors at few or single locations (Clements et al. 2007; Harker et al. 2000; Johnston et al. 1969; Otfinowski et al. 2008; Ren and Bai 2016a; Ren and Bai 2016b; Ren and Bai 2007; Romo and Gross 2011; White et al. 2012; Willms and Quinton 1995). These results are presented in Chapters 3 through 5.

Specific Objectives

1) Summarize producer surveys and relate management to rangeland health.
2) Characterize the diversity and abundance of species within the germinable seed bank of pastures in north central Alberta.
3) Relate seed bank composition to producer management and rangeland health.
4) Identify the recruitment potential of legumes, other forages, and weeds, from the germinable seed bank.
5) Examine plant communities and soils for responses to management factors and relate them to seed bank responses.

1.2.2 Study #2: Understand pipeline disturbance impacts on seed bank composition and displacement of Mixedgrass Prairie

Industrial disturbances such as pipelines, roads, and well sites can function as corridors for seed dispersal and provide an opportunity for invasive species to establish. In the case of linear disturbances like pipelines, species with invasive properties like crested wheatgrass (*Agropyron cristatum*) may be planted to revegetate the disturbed area or opportunistically establish. It is suspected that over-time introduced or ruderal species capable of forming a persistent seed bank could saturate the soil near the disturbance and may eventually begin to establish, “creeping” outward into the adjacent native grassland community. Similarly, legumes such as sweet clover (*Melilotus* spp.) can also exhibit invasive properties in resource limited environments like the Mixedgrass prairie and often exploit disturbed areas such as roadsides and pipelines (Wolf et al. 2008). Sweet clover is deleterious to native grasslands creating a microsite that is open and nitrogen-enriched, which in-turn facilitates the invasion of other exotic species (Van Riper and Larson 2009).

In theory, establishment of invasive plants and saturation of the seed bank will be a function of distance from disturbance, disturbance intensity, and time since establishment. To test whether distance from disturbance increases over time for both plant establishment and seed bank saturation, the seed bank was sampled along pipeline disturbances, with high sampling effort immediately adjacent to the disturbance. The question of potential legume establishment will also be addressed in this study, examining agronomic legumes like *Astragalus cicer* and *Melilotus* spp. which are common in the region and can exhibit invasive properties. We expect to see an increase in their abundance adjacent to pipelines, and perhaps dominance by agronomic legumes and grasses. Overall, studies examining the effects of industrial disturbance, reclamation, and restoration on seed banks are limited (Petherbridge 2000) and their composition is often speculative. Soil surface disturbances (i.e. bare ground, litter) and biological soil crust communities were also examined for their relationship with pipeline disturbance and germinable soil seed banks. Crusts are sensitive to disturbance and slow to recover (Belnap and Eldridge 2001; BLM 2001; Cole 1990), additionally they also serve as a barrier to seed rain and have been demonstrated to influence the seed bank in other environments (Li et al. 2005). This research will inform the influence of disturbance legacies on cryptic communities such as the seed bank and biological crust of Dry Mixedgrass prairie.

Specific Objectives

1) Quantify the diversity and abundance of species present in the seed bank of mixed prairie at various distances from pipeline corridors.
2) Relate differences in seed bank composition and associated vegetation to reclamation practices and pipeline characteristics.
3) Examine the relationship between seed bank composition and density with soil surface biological crusts.

1.2.3 Study #3: Recruitment potential of agronomic, escaped-agronomic, and native legumes from an artificial seed bank

The first two studies identified the diversity and abundance of species in the seed bank, but no direct connection was made between the seed bank and the process of plant recruitment into the above-ground plant community. We monitored the fate of individual legume seeds inserted into established perennial pasture, including emergence, growth and survival. Microsite was manipulated (litter abundance and defoliation of overlying vegetation) to simulate varying conditions created through grazing management. Simulated grazing was expected to reduce light interception by competitive vegetation, potentially aiding in the germination and establishment of legumes. Soil surface litter provides many functions such as soil moisture retention (Adams et al. 2005), and its abundance can be influenced by grazing history where thinner litter layers are associated with heavier forage utilization. Litter cover could influence the germination and recruitment of legumes due to its effects the microenvironment, where abundant litter holds moisture and intercepts light while sparse litter could raise soil temperature and increase light availabilty for seedlings. This would provide a connection between the presence of persistent legume seed banks and the probability of recruitment into the community. Studies 1 and 2 observed legume seed banks in the Central Parkland (along with bordering boreal forest) and Dry Mixedgrass natural subregions, thus this study will test legume species of concern in both ecosystems, including both desirable forage legumes native and tame, as well as legumes known to exhibit invasive properties in native grasslands. Recruitment responses of these species could be linked to observations in the two previous surveys mentioned. These results are reported in Chapter 7.

Specific Objectives

1) Monitor recruitment of six legumes over the growing season and track demographic transitions among life stages.
2) Identify management practices and microsites favorable for each legume species.
3) Identify potential influences of plant communities and microenvironment on legume species and recruitment processes.

1.3 Implications

This research will improve understanding of seed bank responses and legume recruitment potential under divergent management regimes. The effect of various aspects of producer management (i.e. herbicide application, fertilizer, grazing systems, etc.) on seed banks in Western Canadian rangelands is poorly understood and has not been examined in a large multivariate study. Generally speaking, seed bank research in Canadian grasslands is deficient, and replication across ecosystems is often non-existent, with many ecosystems under represented across the Canadian Prairie Provinces. This research will examine the seed banks of managed pastures influenced by many disturbance factors at numerous locations, while many studies examine few factors at a few or single locations (Clements et al. 2007; Harker et al. 2000; Johnston et al. 1969; Otfinowski et al. 2008; Ren and Bai 2016a; Ren and Bai 2016b; Ren and Bai 2007; Romo and Gross 2011; White et al. 2012; Willms and Quinton 1995). In addition, effects of reclamation and industrial disturbance on seed banks have also been overlooked (Petherbridge 2000); this research will provide new insights into community dynamics in invaded and intensely disturbed grasslands. Our pipeline study in native Dry Mixedgrass prairie will also quantify the disturbance legacies on biological crusts, which are understudied in prairie particularly in relation to industrial disturbance (Bowker 2007) and attempt to link their composition and structure to the germinable soil seed bank. Further, legume recruitment in established grasslands can be linked with germinable legume seed banks characterized during the pasture and pipeline survey. Overall, a general increase in knowledge of grassland seed bank responses to management and disturbance in both the Central Parkland and Dry Mixedgrass prairie is expected.

1.4 Literature Cited

Adams, B. W., Ehlert, G., Stone, C., Lawrence, D., Alexander, M., Willoughby, M., Hincz, C., Moisey, D., Burkinshaw, A., Carlson, J. and France, K. 2005. Rangeland health assessment for grassland, forest and tame pasture. Public Lands and Forests Division, Alberta Sustainable Resource Development.

Belnap, J. and Eldridge, D. 2001. Disturbance and recovery of biological soil crusts. In Biological soil crusts: structure, function, and management. Springer Berlin Heidelberg. Pp. 363-383.

Bork, E. W., Grekul, C. W., and De Bruijn, S. L. 2007. Extended pasture forage sward responses to Canada thistle (*Cirsium arvense*) control using herbicides and fertilization. Crop Protection **26**(10):1546-1555.

Bowker, M.A. 2007. Biological soil crust rehabilitation in theory and practice: an underexploited opportunity. Restoration Ecology **15**(1):13-23.

[BLM] Bureau of Land Management. 2001. Biological soil crusts: Ecology and Management, Technical Reference 1730-2. Denver, CO, USA.

Clements, D.R., Krannitz, P.G., and Gillespie, S.M. 2007. Seed bank responses to grazing history by invasive and native plant species in a semi-desert shrub-steppe environment. Northwest Science **81**(1):37-49.

Cole, D.N. 1990. Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park. The Great Basin Naturalist **50**(4):321-325.

Cox, R. D. and Allen, E. B. 2008. Composition of Soil Seed Banks in Southern California Coastal Sage Scrrub and Adjacent Exotic Grassland. Plant Ecology **198**(1):37-46.

Eschtruth, A.K. and Battles, J.J. 2009. Assessing the Relative Importance of Disturbance, Herbivory, Diversity, and Propagule Pressure in Exotic Plant Invasion. Ecological Monographs **79**:265-280.

Gauthier, D.A. and Wiken, E. B. 2003. Monitoring the conservation of grassland habitats, prairie ecozone, Canada. Environmental Monitoring and Assessment **88**(1):343-364.

Gioria, M., Jarosik, V., and Pysek, P. 2014. Impact of invasions by alien plants on soil seed bank communities: emerging patterns. Evolution and Systematics **16**:132-142.

Grekul, C. W. and Bork, E. W. 2007. Fertilization augments Canada thistle (*Cirsium arvense* L. Scop.) control in temperate pastures with herbicides. Crop Protection **26**(4):668-676.

Harker, K. N., Baron, V. S., Chanasyk, D.S., Naeth, M.A., and Stevenson, F.C. 2000. Grazing intensity effects on weed populations in annual and perennial pasture systems. Weed Science **48**(2):231-238.

Havstad, K.M., Peters, D.P., Skaggs, R., Brown, J., Bestelmeyer, B., Fredrickson, E., Herrick, J. and Wright, J. 2007. Ecological services to and from rangelands of the United States. Ecological Economics **64**(2):261-268.

Henderson, D.C. and Naeth, M.A. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions **7**:639-650.

Johnston, A., Smoliak, S. and P.W. Stringer. 1969. Viable seed populations in Alberta prairie topsoils. Canadian Journal of Plant Science **49**(1):75-82.

Ledgard, S.F. and Steele, K.W. 1992. Biological nitrogen fixation in mixed legume/grass pastures. Plant and Soil **141**:137-153.

Li, X. Jia, X., Long, L., and Zerbe, S. 2005. Effects of biological soil crusts on seed bank, germination and establishment of two annual plant species in the Tengger Desert (N China). Plant and Soil **277**(1):375-385.

Otfinowski, R., Kenkel, N.C., and Van Acker, R.C. 2008. Reconciling seed dispersal and seed bank observations to predict smooth brome (*Bromus inermis*) invasions of northern prairie. Invasive Plant Science and Management **1**:279-286.

Petherbridge, W.L. 2000. Sod salvage and minimal disturbance pipeline reclamation techniques: implications for native prairie restoration. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Province of Alberta. 2010. Weed Control Act. Her Majesty the Queen in the Right of Alberta, Edmonton.

Sanderson, M. A., S. C. Goslee, K. D. Klement, and Soder, K.J. 2007. Soil seed bank composition in pastures of diverse mixtures of temperate forages. Agronomy Journal **99**:1514-1520.

Turkington, R. A., Cavers, P. B., and Rempel, E. 1978. The biology of Canadian weeds: 29. *Melilotus alba* Desr. And *M. officinalis* (L.) Lam. Canadian Journal of Plant Science **58**(2):523-537.

Van Riper, L.C. and Larson, D.L. 2009. Role of invasive *Melilotus officinalis* in two native plant communities. Plant Ecology **200**:129-139.

White, S.R., Bork, E.W., Karst, J., and Cahill, J. 2012. Similarity between grassland vegetation and seed bank shifts with altered precipitation and clipping, but not warming. Community Ecology **13**(2):129-136.

Willms, W.D., and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Wolf, J. J., Beatty, S.W., Carey, and G. 2008. Invasion by sweet clover (*Melilotus*) in montane grasslands, Rocky Mountain National Park. Annals of the Association of American Geographers **93**(3): 531-543.

Chapter 2

Seed Banks and Legumes in the Northern Great Plains: A Literature Review

2.1 Introduction to Seed Banks

A seed bank contains sexually produced angiosperm propagules (seeds) stored in the topsoil, which contribute new individuals and genes to plant communities overtime. Seed banks are dynamic communities that fluctuate seasonally and temporally (Coffin and Laurenroth 1989) depending on the current and recent properties of the overlying community but are also shaped by their disturbance history and stochastic interactions in the environment. Composition can influence current grassland community dynamics, but more importantly affect community successional trajectories (Clements et al. 2007; Renne and Tracy 2007). Aboveground, plants compete for resources to reproduce, however, only a fraction of their seed will be successfully incorporated into the soil seed bank and even fewer seeds will eventually be recruited. Belowground, generations of seed can remain dormant until physiological (e.g. imbibition, temperature, light) or physical (e.g. degradation of seed coat) requirements for germination are met. Dormancy is affected by seed morphology; where seed size, shape and hardness determine which species will achieve burial, germinate, degrade, or be preyed upon (Thompson et al. 1993; Thompson et al. 1997). Thus, seed banks can differ substantially from the floral composition aboveground by containing historical or novel species awaiting disturbance, or by varying in proportional abundance relative to the existing vegetation (White et al. 2012).

Seed banks are an important part of natural and agro-ecosystems. Expression of seed banks is particularly evident following severe disturbances (i.e. fire, erosion, flooding, tilling, etc.) that disrupt existing vegetation and necessitate revegetation from seed, or where the vegetation is largely annual and thereby tied to seed bank availability. In either situation, ruderal species that easily break dormancy and grow quickly will initially dominate the community. Seed banks also contribute to succession, as overtime ruderals will be replaced with more competitive perennials. In addition to their role in regulating community assembly, seed banks also maintain biodiversity and contribute individuals to microsites that may emerge at smaller scales opened by herbivores and associated plant mortality (Renne and Tracy 2007; Sanderson et al. 2007). Formation of persistent seed banks (lasting multiple growing seasons) of desirable forages and native species is preferred as these seed banks are thought to buffer the community from degradation and maintain late seral or rare species (Thompson et al. 1993).

Despite their importance, seed banks have been understudied in Western Canadian rangelands. Johnston et al. (1969) pioneered seed bank research in Alberta observing seed banks in southern Mixedgrass prairie and cultivated fields. Willms and Quinton (1995) sampled the seed bank of a foothills rough fescue prairie exposed to different long-term grazing intensities west of Stavely, Alberta. Harker et al. (2000) observed weed seedling recruitment among annual and perennial forage grasses grazed at various intensities over a four-year period at Lacombe, Alberta, providing insight into the germinable seed banks in tame (seeded) grasslands and annual forage crops. White et al. (2012) measured seed bank responses to warming, defoliation, and reduced precipitation in a plains rough fescue grassland near Kinsella, Alberta. Using a series of studies, Ren and Bai (2016a; 2016b; and 2017) examined the influence of prescribed fire and smoke on the germinable seed bank and germination cues in plains rough fescue grasslands at the Kernen Prairie near Saskatoon, Saskatchewan. Romo and Gross (2011) examined the effects of burn season and pre-burn history on the composition of fescue grassland seed banks at the Kernen Prarie as well. Clements et al. (2007) observed seed bank responses to different grazing histories in Washington and British Columbia's Semi-Desert Shrub Steppe. These studies represent our current knowledge of seed banks in the Canadian plains, with grasslands in the provinces of British Columbia, Manitoba, and Saskatchewan distinctly under-represented. Replication within Alberta's diverse grassland community types is also lacking. Although the geographical scope of these studies is limited, they are beneficial to our understanding of seed banks because they attempt to answer questions involving management factors and environmental change. For example, results of these studies have provided valuable knowledge on the impacts of grazing on seed bank ecology and potential rangeland vegetation dynamics (Johnston et al. 1969; Willms and Quinton 1995). A handful of seed bank studies by neighboring states (primarily in North Dakota) have been published, with a focus on prairie wetlands in the context of cultivated agro-ecosystems (Gleason et al. 2003; Poiani and Johnson 1988; Wienhold and van der Valk 1989) and tame forages (Carr et al. 2005).

Seed bank studies in agro-ecosystems are more prevalent, as the economic loss due to weed competition with crops has motivated significant seed bank research in this area (Ball 1992; Buhler et al. 1997; Johnston 1969; Harker et al. 2000; Mayor and Dessaint 1998). In cropping systems, the seed bank is a major source of infestations of annual taxa, with a few dominant species comprising 70 to 90% of the seed bank (Buhler et al. 1997). Seed bank formation in cultivated systems favours transient species that pulse seasonally (Buhler et al. 1997), as frequent perturbation prevents persistent seed banks from forming (Thompson et al. 1997). Controlling the weed seed bank in these management systems focuses on eliminating seed production and exhausting the seed bank (Buhler et al. 1997). Concerns in cultivated fields include weed resistance to herbicides and adaption to the cropping system (Buhler et al. 1997). Annuals that occur in cultivated fields can also emerge from grassland seed banks, and 10 to 20% of cropland seed banks may include native plant species endemic to the region (Buhler et al. 1997).

Many studies observe aboveground vegetation changes in response to treatments, but the effect of those treatments on seed banks is seldom included (Sanderson et al. 2007). Further, changes in seed bank composition may be inferred based on vegetation changes (Cox et al. 2008; Gioria et al. 2014), though this assumes a high degree of responsiveness in the seed bank relative to aboveground vegetation change. A flaw in many seed bank studies is the emphasis placed on similarity indices, which although informative, the general consensus is that seed banks and floral communities are dissimilar unless xeric (Hopfensperger 2007). With limited direct observation of seed bank communities, any research into their composition and formation under divergent management regimes is justified. Little is known about how management actions and disturbances in the environment affect the seed bank. Maintaining desirable seed bank composition should be a management objective in grasslands as seed banks are indicative of past disturbances and can be predictive of future plant communities (Clements et al. 2007). With the rise of weeds and invasive species in our ecosystems globally, we should focus on minimizing the development of exotic persistent seed banks (Eschtruth and Battles 2009).

In this review, the types of seed banks discussed in the literature are described, influences on grassland seed bank composition are discussed in relation to Alberta's grasslands, and concepts of recruitment from seed banks and community assembly are discussed. In addition, methods of comparing seed banks to the aboveground community, characterizing and quantifying seed banks, and review plant groups and species of concern regarding my research questions (i.e. legumes, forages, exotic invasive species, and native species) are discussed.

2.2 Types of Seed Banks

There are two main types of seed banks described in the ecological literature – transient and persistent (Thompson and Grime 1979). Transient and persistent seed banks differ functionally, contributing to the above ground plant community in unique ways.

2.2.1 Transient Seed Banks

Transient seed banks are typically comprised of species with short dormancy periods and low persistence in the soil; hence, these seeds are confined to the top of the soil profile for a short period after seed entry (i.e. seed rain) into the soil (Thompson et al. 1997). In northern temperate grasslands, germination of transient species often occurs in the fall, especially when secondary ripening is not required (i.e. in the case of winter annuals) but can also occur in the spring. The role of seed size and shape in persistence has been studied extensively; seeds with transient seed banks tend to be relatively larger, flattened, or elongated when compared to the small compact seeds that comprise persistent seed banks (Thompson et al. 1993). Transient species are often susceptible to degradation or predation in their environment (Sanderson et al. 2014). Degradation is caused by soil microorganisms, but mechanical weathering (i.e. freeze-frost cycles, fluctuations in soil moisture, etc.) can also break down the thinner seed coats of these species. Plants with larger seeds tend to have short-lived seed banks, primarily because their seeds are more susceptible to granivores and they are more difficult to incorporate into the soil (Thompson et al. 1993), thereby increasing their exposure to predators at the soil surface. A handful of species can bypass these obstacles through self-burial, which can be assisted by unique seed morphologies (i.e. *Hesperostipa*) (Molano-Flores 2012). Populations of transient species pulse seasonally, thus sampling transient seed banks is best done in the fall after the current year's growth has dispersed its seed. Dandelion (*Taraxacum officinale*), along with numerous other species in the Asteraceae family, exemplify transient seed banks; all these species have light-weight seeds with thin seed coats that often break down quickly in the soil if they do not land in a suitable microsite (Tracy and Sanderson 2000).

Transient species commonly occupy an important ecological niche as colonizers and pioneer species. These species are typically described as weedy in habit but encompasses both native and exotic species. Alternatively, Kinucan and Smeins (1992) proposed that late seral grass species may be transient in nature as their seeds can be detected in seed rain but are rarely found in the soil seed bank. For these species, seeds are suspected to germinate immediately following dispersal, provided there is sufficient moisture, or are otherwise lost to predation and pathogens (Kinucan and Smeins 1992).

2.2.2 Persistent Seed Banks

Persistent seed banks are comprised of species that remain viable over multiple growing seasons. Seeds from these species tend to have indurate (hard) and thick seed-coats, longer dormancy, and relatively small size (<3 mg) (Eriksson and Eriksson 1997; Sanderson et al. 2014; Thompson et al. 1993). These characteristics enable seeds to survive long enough to become incorporated deeper into the soil profile, where the probability of remaining viable for longer is greater (Thompson et al. 1993; Thompson et al. 1997). Methods of penetrating the soil include entering cracks in the soil surface, ingestion by earthworms, and self-burial mechanisms (Thompson et al. 1993). The most persistent species are often very abundant in the seed bank but can be rare or absent in the existing plant community (Kinucan and Smeins 1992). These species will also be found deeper in the soil profile (Willms and Quinton 1995). In northern temperate grasslands, plants from families with small hard seeds exhibit high abundance and diversity in persistent seed banks. Sampling soil in late winter and early spring is the ideal time to capture the abundance and diversity of persistent seed banks. Formation of a persistent seed bank can provide a plant community with increased resilience and resistance when facing contemporary disturbances and allows stochastic recruitment of later seral species that are rare or in decline (Thompson et al. 1983). However, persistence is lower in soils experiencing repeated and intense disturbance (Thompson et al. 1997).

Thompson and Grime (1979) further described two types of persistent seed banks—short-term (1-5 years) and long-term (>5 years) persistent seed banks. Short-term persistent species will still pulse seasonally, resembling the function of more transient species. In their original description a longer-term transient seed bank was also identified of 1 to 2 years was identified, but it may be more useful to clump intermediate types as they are difficult to distinguish (Thompson and Grime 1979).

2.3 Seed Bank Formation

Like the aboveground community, a number of environmental factors can shape seed bank composition. Formation can be regulated by propagule inputs from aboveground and by factors that regulate seed dormancy.

Seeds are primarily sourced from seed rain; this includes diaspores (seeds or fruits) shed from the parent and disseminated across the ground. The capacity of a plant's seed to move is determined by modifications to fruits such as a pappus (Asteraceae and *Epilobium*), hooks/burs (*Glycyrrhiza*), wings, dehiscence or more active dispersal mechanisms like explosive dehiscence (i.e. *Viola, Geranium*, *Impatiens*) (Moss 2010). Navigation is absent; direction is determined by dispersal vectors like wind, gravity, or animals (Damschen et al. 2008). Ultimately, seeds may only travel a few meters, especially in species which invest in large and heavy seeds (Dornier et al. 2011; Honnay et al. 2005).

The effect of time on seed banks has been explored in two main ways, including composition change with temporal distance from a disturbance event and seasonality (Coffin and Laurenroth 1989; Willms and Quinton 1995). Seed banks are dynamic, with seed density and diversity fluctuating as seeds are introduced and incorporated into the seed bank from seed rain throughout the growing season and eliminated due to pathogens and granivory. Resident time of species is then determined by their seed's biology. However, persistent annuals tend to be abundant throughout the year (Coffin and Laurenroth 1989).

There are two theories explaining seed bank formation (Helsen et al. 2015): 1) species richness gradually increases with progressive species introductions at each successional stage (Davies and Waite, 1998), with the final seed bank representing an ecological legacy of its past communities; or 2) progression in aboveground plant community assembly is paired with belowground seed bank community disassembly, where species are lost deterministically from the seed bank based on seed characteristics and ongoing elimination of species from the aboveground floral community (Zavaleta et al. 2009). In grasslands, both ideas could play a role in plant community dynamics as divergent management histories have been linked to changes in seed bank composition and time elapsed since disturbance has been linked to seed bank composition nested in the diversity of their more disturbed predecessor (Helsen et al., 2015). Essentially, seed banks are responsive to management or environmental changes (i.e. introduction of invasive species), but if conditions for establishment of a species stored in the seed bank in not met it is expected that it could be eliminated overtime. Hence, we expect that seed banks will reflect the current plant community and retain residual species (potential indicator species) incorporated into the seed bank following major disturbance events like cultivation or fire.

2.4 Grassland Seed Banks

Grasslands are disclimax communities maintained through disturbance (i.e. grazing, fire, or drought) (Molles and Cahill 2008). In the Northern Great Plains, native grasslands are comprised of perennial grasses, forbs of various life strategies, and shrubs; contemporary grasslands are fragmented and are often found in xeric, rocky, saline/alkaline, or sandy regions. Dominant perennial grasses contribute little to the seed bank, as they primarily invest in vegetative growth (Coffin and Laurenroth 1989; Ma et al. 2010; Sanderson et al. 2014); a handful of perennial forbs and shrubs use a similar strategy. Many other native forbs however, tend to be more ephemeral, requiring regular recruitment from the seed bank (Clements et al. 2007). The functional importance of fire and grazing in these systems is discussed in section 2.5.

Introduced grasslands are functionally different from native grasslands, being dominated by cool-season forage grasses and introduced legumes. In North America the seed banks of introduced grasslands have been found to consist of weedy annuals (40%), perennial grasses (11%), perennial forbs (23%), and legumes (19%) (Tracy and Sanderson 2000). Abundant forages found in the seed bank include white clover (*Trifolium repens*), Kentucky bluegrass (*Poa pratensis*), and dandelion (*Taraxacum officinale*) (Sanderson et al. 2007; Tracy and Sanderson 2000). Due to the low abundance of desirable species Tracy and Sanderson (2000) concluded introduced grasslands do not have a large reservoir of seeds representing desirable forages, and as such, managers seeking to diversify their pastures will likely have to reseed.

2.5 Factors Influencing Seed Bank Composition

Seasonal inputs to the composition of the seed bank are influenced by the aboveground vegetation, ongoing disturbance, and growing conditions. Inputs can be affected by the management of the community aboveground if flowering and seed set potential are altered through disturbances such as herbivory/mowing, frost, fire, herbicide, drought, etc. At a landscape level, topographic variation effects soil moisture, texture, organic matter and nutrients, all of which can lead to heterogeneity in plant community expression, and consequently the composition of annual seed rain. Thus, seed bank composition can be expected to vary with shifts in the expression of plant communities, although migration of seed rain among communities may also occur.

2.5.1 Anthropogenic Disturbance and Agronomic Impacts

Anthropogenic disturbances encompass the variety of direct and indirect manners in which humans influence the environment. This definition is broad, including responsible range management practices such as grazing, fertilization and weed control to more intrusive disturbances like roads and oil and gas development. Disturbances like grazing and fire can fall into the realm of natural and non-natural depending on their source; where domestic livestock and prescribed or accidental fire are anthropogenic. In contrast, grazing by wildlife and environmental fire ignitions (lightning) would be the natural equivalent of these disturbances.

2.5.1.1 Grazing

Grazing practices are known to influence the composition of seedbanks through the timing and intensity of grazing (Kinucan and Smeins 1992). Grazing functions as a disturbance via the removal of plant biomass and associated flowering parts, directly reducing seed inputs (Sanderson et al. 2007). However, herbivores may also serve an important role in burying and compacting soil around seed (Williams 1984), potentially improving the success of seed emergence and survival. Historically herbivores and grasslands have closely co-evolved, and grazing activity has a number of positive effects on grassland ecosystems (Milchunas et al. 1988). A number of native grasses and forbs rely on herbivores for seed dispersal (in fur or manure) and thereby facilitate the exchange of genetic material among populations. Seed bank species richness is often highest in grazed communities (Jacquemyn et al. 2011; Zhan et al. 2007); an effect reflected in aboveground communities as well, particularly those that evolved with herbivory (Milchunas et al. 1988). Increasing intensities of herbivore activity can also alter the micro-environment at the soil surface by reducing litter (Willms and Quinton 1995) and microphytic crust (Clements et al. 2007) abundance, as well as increasing bare soil for seed reception, all of which are believed to influence dormancy and germination.

Grazing intensity refers to the amount of vegetation removed and is reflected by livestock stocking rates; however, stocking rate impacts can be further modified if paired with long durations of defoliation or by frequently removing vegetation within a growing season. Species sensitive to grazing may be reduced by high stocking rates (Willms et al. 1985), changing the aboveground species composition and ultimately altering the associated seed bank (Willms and Quinton 1995).

High stocking rates can alter the composition and structure of plant communities (Smoliak 1974; Willms et al. 1985), due to disturbance from livestock and defoliation which alters propagule inputs (Kinucan and Smeins 1992). High grazing intensity often leads to recruitment of weedy species from the seed bank (Wellstein, et al. 2007), which in turn, increases the abundance of weedy species in the seed bank (Kinucan and Smeins 1992; Renne and Tracy 2007). Willms and Quinton (1995) found that more seed accumulated on the soil surface of heavily grazed sites, but these same areas had relatively fewer seeds in the soil than ungrazed sites. Low seed density from heavy grazing has been explained by increased bare ground. Bare ground leaves seeds vulnerable to predation and degradation from microbes but favors the establishment of ruderal species that require minimal competition to establish. Tracy and Sanderson (2000) found that dominant vegetation was under represented in the germinable seed bank, and legumes like white clover (*Trifolium repens*) were more abundant than perennial grasses. Similarly, under high stocking rates Kentucky bluegrass (*Poa pratensis*) becomes the dominant perennial grass species in the seed bank (Sanderson et al. 2007; Tracy and Sanderson 2000; Willms and Quinton 1995), which likely reflects the grazing tolerant nature of this grass and the increase this species experiences under prolonged heavy grazing (Willms et al. 1985). Relatively low abundance of late-seral perennial grasses in seed banks can be explained by their reproductive strategy and the grazing strategy used by the producer. Perennial grasses invest more resources into vegetative than reproductive growth (Ma et al. 2010; Sanderson et al. 2014), thereby limiting seed inputs. Moreover, if grazing coincides with the sexually reproductive period of a perennial grass, reproductive effort may be further negated through defoliation

(Tracy and Sanderson 2000). Long-term heavy grazing results in seed banks with limited restoration potential and low persistence of palatable forages (Zahn et al. 2007).

In contrast, Willms and Quinton (1995) found that perennial forbs were most abundant in pastures grazed at lower stocking rates. Tracey and Sanderson (2000) proposed that cattle grazing may increase white clover abundance in the seed bank indirectly through the defoliation of taller grasses, which increases light availability for more prostrate plants. Aside from species like white clover, grazing has not been shown to increase desirable forage species in the seed bank (Tracy and Sanderson 2000).

When grazing pressure is removed over multiple growing seasons, many plant communities eventually decrease in diversity (Milchunas et al. 1988). Consequently, the corresponding seed bank decreases in both species richness and diversity (Eriksson and Eriksson 1997; Jacquemyn et al. 2011). If the goal of grazing pressure removal is to improve range condition, removal can lead to higher proportions of late seral perennial monocot taxa (Kinucan and Smeins 1992). Tracey and Sanderson (2000) also found that the removal of grazing increased the abundance of grass seeds within the soil for seeded (tame) pastures.

A less direct effect of grazing on the seed banks of grassland systems is the reduction in litter quantity, which increases seed losses to germination and predation (Willms and Quinton 1995). Litter regulates soil temperature, water evaporation from soil, and light availability—factors that together influence the quality of micro environment where seeds germinate (Facelli and Pickett 1991). Higher relative abundance of germinable seed on lightly grazed sites suggests abundant litter could assist seed preservation and maintain dormancy (Williams 1983; Willms and Quinton 1995). In contrast, in an ungrazed system where vegetation is dense, litter accumulation is greater, and bare ground limited, seeds may be prevented from reaching favorable germination sites in mineral soil (Williams 1984). Overall, germination and establishment from the seed bank tend to be lower when litter depth increases (Xiong et al. 1999), which could be attributed to the degradation of seeds captured in litter by pathogens (Xiong et al. 1999). Hence, both a high intensity grazing and the absence of grazing may negatively affect the accumulation of seed in the soil seed bank and/or plant recruitment, in turn suggesting effective management of seed banks requires light grazing to maintain species richness.

Seed banks can also be modified through grazing systems, which control the intensity and duration of grazing temporally and topographically. There is great variation between grazing systems and the effects of their management, thus we will only discus a few broad categories of grazing systems, including continuous and rotational systems. Continuous grazing is characterized by season-long grazing, where livestock graze repeatedly on preferred patches of vegetation, often in large pastures. Selection of vegetation will vary with plant phenology and time of year. With continuous systems, certain plant species will be highly selected, causing their populations to decrease or lead to degradation of certain areas abundant in preferred and easily accessible forage. When season-long grazing is employed it is expected that seed production of taller forage plants will decrease and disturbance in high-use patches will contribute to a seed bank dominated by grazing tolerant and ruderal species (Kinucan and Smeins 1992). In addition, this can lead to a decrease in seed bank diversity (Tracy and Sanderson 2000). In contrast, rotational systems utilize strategic fencing to target or defer grazing in certain areas based on the production potential and conservation value of certain paddocks. Ideally, rotational systems can prevent localized overuse problems that arise in continuous systems by allowing producers to defer use of sensitive ecosystems like wetlands and stream banks, as well as defer grazing until seed set and thereby facilitate seed cast and plant renewal, among other benefits.

2.5.1.2 Cultivation

Tame grasslands are often seeded; previous use can vary from annual crop production, hay fields, to grazed pasture. If the land has been previously broken by plow, expression in the plant community and seed bank will reflect this disturbance (Sanderson et al. 2007). In some regions, periodic ploughing can be employed to rejuvenate pastures (Levassor et al. 1990); with intermediate disturbance (every 2 to 4 years) promoting higher seed density and diversity. Early successional stages following ploughing see the abundance of generalist species increase, which tend to remain present throughout later successional stages (Levassor et al. 1990). Within seeded pastures, up to 79% of the germinable seed bank can be comprised of annual non-leguminous forbs (Sanderson et al. 2007). With the exceptions of Kentucky bluegrass (*Poa pratensis*) and white clover (*Trifolium repens*) (Sanderson et al. 2007), seeded species tend to contribute little to the seed bank of introduced pastures. In some cases, it may be desirable to convert previously cultivated land into perennial grassland. When this conversion is made, the seed bank lacks desirable perennial grasses (Zhan et al. 2007) and seeding is required.

2.5.1.3 Fertilization

Fertilizer can be applied to grasslands in two forms: chemical fertilizer dissolved in water and sprayed or manure. Long-term effects of fertilizer application on pasture seed banks have been understudied. In theory, seed production should increase with fertilization and the removal of nutrient limitations, in turn, increasing seed inputs to the seed bank provided vegetation is allowed to progress through seed cast. However, green house and field experiments by Williams (1984) both showed that the effect of fertilizer on seed production is dependent on plant species, with some showing potential for reduced seed production. Williams (1984) concluded that the seed of perennial grasses becomes more abundant in the seed bank when intensively managed and well-fertilized.

Unlike fertilizer, manure application has the potential to directly modify the seed bank through the addition of seeds that passed intact through an herbivore's digestive system. Seeds that can survive digestion by cattle include ruderal species like common lamb's quarters (*Chenopodium album*), yellow foxtail (*Setaria glauca*), common chickweed (*Stellaria media*) (Pleasant and Schlather 1994) and legumes (Gardener et al. 1993), at the potential rate of 75,100 seeds/kg of manure (Pleasant and Schlather 1994). Thus, a higher density of annuals occurs in pasture seed banks historically treated with manure (López-Mariño et al. 2000). Moreover, several studies have explored the use of livestock as deliberate agents for the introduction of desirable forages into pasture swards (Edwards and Younger 2006; Neto et al. 1987). When considering cow pats for vectors of seed dispersal and introduction, Malo and Suárez (1995) found that recruitment on dung pats was primarily from endozoochorous seeds (seed that passed through an animal) and resulted in small scale community heterogeneity at the site of disintegrated dung pats.

2.5.1.4 Herbicide application

Studies on the effect of herbicide application on seed banks are deficient outside of cultivated (i.e. annually cropped) agro-ecosystems, however inferences can be made from the latter weed management studies. Ball (1992) found that weeds are more abundant if they are less susceptible to the herbicide(s) chosen to treat specific crops, in-turn affecting seed bank composition. Whether similar results would occur in pastures remains to be tested. Overall seed bank density can decline with persistent herbicide use, but it rapidly increases after use is discontinued (Ball 1992). Herbicides rarely eliminate the entire weed communities, but even sub-lethal doses can markedly reduce seed production (Buhler et al. 1997). An additional concern in both cultivated systems and perennial grasslands is herbicide resistant weed populations (Ball 1992; Buhler et al. 1997).

In grasslands, communities are typically a heterogeneous mix of graminoids, forbs, and shrubs. This poses a problem when attempting to control a weed with herbicides, as non-target species will often be affected. Legumes are a desirable broadleaf plant in both native grasslands and tame pastures, and they are sensitive to many herbicide products marketed to control broadleaf weeds in grasslands used for livestock production (Miller et al. 2015). A further concern to producers is the residual nature of some broadleaf herbicides which can extend the effective window for weed control but may also prevent the re-establishment of desirable forbs like legumes (Miller et al. 2015) as well as delay the opportunity for reseeding.

2.5.1.5 Industrial Disturbance

Oil and gas developments significantly disturb soil during installation. In cases where vegetation and soil are removed, undesirable species have the opportunity to colonize in the absence of competition (Allred et al. 2015). Although this infrastructure has become common among prairie and cultivated landscapes (Allred et al. 2015) the contributions of these disturbances to prairie seed bank ecology has been understudied. Species seeded to reclaim oil and gas disturbances, along with voluntary invasive species, like crested wheatgrass (*Agropyron cristatum*) and sweet clover (*Melilotus* spp.) have established seed banks contributing to the invasion of the adjacent communities (Henderson and Naeth 2005; Simmers and Galatowitsch 2010). Currently there is little research comparing seed banks of reclaimed or restored sites with intense anthropogenic disturbance (Petherbridge 2000), and few studies have observed change in invaded communities (Gioria et al. 2014). Thus, research into seed bank responses here will be novel and supplement a research gap.

2.5.2 Natural Disturbance

Fire was a common disturbance among grasslands in the northern Great Plains prior to European settlement (Archibold et al. 2003), with a fire return interval of approximately every 10-15 years (Wright and Bailey 1982). In northern regions like the Aspen Parkland, fire was functionally important by maintaining rough fescue grasslands which otherwise are susceptible to aspen encroachment (Sheffler 1976; Bailey and Wroe 1974) given the favorable moisture regime (Archibold et al. 2003). Although perhaps less dependent on fire for woody vegetation control, the more arid Mixedgrass prairie was also impacted by periodic fire with a fire return interval of about every 25 years (Wright and Bailey 1982), and this would have created substantial landscape heterogeneity, thereby benefiting a variety of wildlife species. Despite the historical importance of fire, this disturbance is now heavily supressed, and we have a relatively limited understanding of how these ecosystems are impacted by fire. In particular, research into the role of fire in regulating seed bank composition and its role in facilitating secondary succession is limited in North America. Limited evidence from Alberta indicates that legumes often demonstrate marked increases following fire (Bork et al. 2002), suggesting that persistent legume seeds in the seed bank may be released by fire itself, the post-fire environmental conditions, or a combination of the two. Fire itself is known to break the dormancy of many hard-coated legume seeds (Martin et al. 1975) and is considered an important recovery mechanism to allow burned communities to recover (Bork et al. 2002), in part by building up soil nutrients. Post-fire, the fescue grassland community in Alberta shifts towards perennial forbs (Bailey and Anderson 1978), likely resulting from the temporary reduction of dominant later seral grasses and accumulated litter, and the introduction of germination cues in the form of smoke and ash (Ren and Bai 2016a). Other studies suggest small hard seeds of annuals typically survive the heat associated with fire, and the seed rain immediately following the disturbance leads to an increase in seed bank density and diversity (Gonzales and Ghermandi 2008). Romo and Gross (2011) found that burning fescue grassland during or after the growing season can reduce the richness and diversity of the germinable seed bank. The seed bank composition of fescue grasslands has also been linked to their pre-burn histories [burned twice in a 13 to14 year window before the study vs no fire in > 90 years] (Romo and Gross 2011), although this affect post-fire was overshadowed by the recent disturbance. Recent research from fescue grasslands in north-central Saskatchewan identified divergent germinable seed bank responses to smoke and ash treatments (Ren and Bai 2016a), and species-specific germination responses from smoke derived from different plants like wheat, alfalfa, and fescue prairie hay (Ren and Bai 2016b). Germinable seed bank richness and forb richness increased with smoke and ash addition among all soil surface layers [litter and 0 to 5 cm], while other functional plant groupings exhibited unique responses to smoke and ash treatment. Improved germination of certain grassland species [e.g., *Artemisia frigida* and *Conyza canadensis*] exposed to the smoke derived from an herbaceous legume [alfalfa] could have been stimulated by an additional germination cue in the form of NO and NO_2 (Ren and Bai 2016b). Further, prescribed burns influenced the emergence of seeds from the topsoil monitored in field and soil seed bank cores (Ren and Bai 2017). In the field greater foliar cover was attributed to the emergence of early and mid-seral Asteraceae species (*Artemisia frigida, A. ludoviciana, Cirsium arvense* and *Conyza canadensis*), however total germinable seed densities, richness, and diversity were reduced at all depths [litter and 0-5 cm] due to damage (Ren and Bai 2017). In one year of the study, burning had a positive effect on native forb emergence from the top 1 cm of soil (Ren and Bai 2017). This research shows that fire can have diverse effects on seed bank composition, seedling emergence, and plant community assembly in grasslands.

2.5.3 Abiotic influences

Topography and land formations affect soil formation through moisture and nutrient retention at variable positions on the landscape; with thinner more xeric soils at hill tops and sides, and thicker, mesic, nutrient rich soils are found at lower positions. Aspect causes sun exposure to be higher on southern-facing slopes, and to some extent western, which results in relatively dry southern slopes and mesic northern slopes. In turn this affects the aboveground plant communities and their corresponding seed banks.

While aboveground community responses to topography and soils are well understood, the effect this has on seed banks is less understood. Coffin and Laurenroth (1989) explored the relationship between soil texture and soil seed bank composition in a semiarid grassland; fine textured soil had significantly more annuals and coarse textured soil had more perennial grasses, reflecting divergence in the overlying plant community. Despite these differences they concluded that spatial variability in seed bank composition was relatively low, similarly Clements et al. (2007) found no significant difference in seed bank composition between sites with varying soil texture. At the microsite level, small variation in micro-topographic features can influence species specific seed dispersal patterns; where seeds aggregate and germinate in small open areas, often resulting from disturbance (Kinucan and Smeins, 1992), this may enhance recruitment of various plant species.

2.6 Aboveground Floristic Composition Relative to the Belowground Seed Bank

The relationship between aboveground plant species composition and the underlying seed bank composition are conventionally compared using a Sørenson's index of community similarity. Where the number of species two sites have in common (in our case, the seed bank and plant community) multiplied by 2, then divided by the sum of the absolute number of species in each site.

$$S = 2(A \cap B) / A+B$$

Studies on rangeland seed banks show that aboveground species composition tends to weakly correlate with seed bank composition (Eriksson and Eriksson 1997; Tracey and Sanderson 2000; Williams 1984), presumably due to more rapid changes in the aboveground community relative to the seed bank, as well as necessary time lags in the establishment of vegetation from the seed bank following disturbance. However, Hopfensperger (2007) showed that grassland seed banks have the highest similarity to the aboveground community (54±2.7%) when compared to similarity indexes for forests (31±3.7%) and wetlands (47±2.4%). Hopfensperger (2007) also found similarity between above and belowground species richness increased with time elapsed following major disturbance events. This result was attributed to low initial species richness following the disturbance itself and low seed dispersal distances of grassland species (thereby slowing species re-entry) (Hopfensperger 2007). Consequently, non-grazed and lightly grazed pastures tended to have a higher similarity between the seed bank and aboveground vegetation (Tracy and Sanderson 2000). Finally, similarity also tends to be higher in more xeric (water-limited) environments than in more mesic grasslands (Hopfensperger 2007; White et al. 2012). In general, it is not uncommon for less than half of the species occurring in the aboveground vegetation to be found in the germinable seed bank (Eriksson and Eriksson 1997). Low correlations between above and belowground species richness could result from the 'noise' of rare species (Levassor et al. 1990). Additionally, insufficient sample size among studies could explain the low correlation, as soil coring often represents only a small proportion of the total aboveground surface area available, resulting in a failure to sample rare species. To increase the power of the detection analysis of species within the seed bank, there are two solutions: 1) sample α-diversity over a smaller area and intensively sampling that area in isolation, or 2) when calculating overall α-diversity for a pasture, take a large number of randomly distributed samples to account for as much of the landscape heterogeneity (micro- and macro-) as possible.

Grassland seed bank studies often report a number of species that are rare in the aboveground vegetation, but abundant in the seed bank (Tracy and Sanderson 2000). These species are often weedy or ruderal species, and likely represent previously dominant, species with high seed rain, successive vegetation, or propagules immigrating into the community from nearby disturbed lands. Similarly, Wellstein et al. (2007) found that species with greater seed accumulation in the soil, indicating the formation of a long-term persistent seed bank, were rare in the aboveground vegetation. Even in native grasslands non-grazed by livestock, annual weedy forbs are present in the seed bank, and would likely establish following a disturbance (Willms and Quinton 1995). Finding an abundance of these species in the seed bank would be informative to producers, as it suggests caution should be exercised in disturbing these areas.

2.7 Recruitment and Community Assembly

Rules of community assembly play an important role in understanding how individual seeds are incorporated into the aboveground community from the belowground species pool. The above ground community does not express relative abundances of species found in the seed bank (Hopfensperger 2007), or seed bank diversity (Kinucan and Smeins 1992), thus there are constraints on the establishment of individuals and particular species. Theories governing community assembly have numerous competing mechanisms, and therefore, only the most relevant theories pertaining to grassland ecology will be discussed here. When physiological conditions for germination have been met seedlings will have to successfully pass through a number of environmental filters (i.e. factors constraining establishment) to ultimately contribute to the population and associated plant community (Booth and Swanton, 2002). Primary limitations include seed dispersal (can seeds disperse into an environment?) and environmental conditions (soil moisture, soil texture, temperature, seasonal variability, disturbance frequency, etc.); once germinated, competition imposes stress on seedlings (Booth and Swanton 2002).

Interspecific and intraspecific plant competition is important to consider when discussing community assembly. Each species has a unique morphology and life history strategy (Keddy 1990), and within species more competitive genotypes can exist. Simplified competition models assume that competition is symmetrical (i.e. competition between species is proportional to plant size), but competition can be asymmetrical (Schwinning and Weiner 1998) as determined by each species having unique advantages (i.e. environmental tolerances, light interception efficiency, etc.). Thus, competition models that incorporate a competitive hierarchy are more predictive (Keddy 1990). Competitive hierarchies can be devised by grouping species with similar traits and functions (Fargione et al. 2003; Keddy 1990). Competitive plant species are generally larger (i.e. tall grasses vs. basal rosettes), with abundant biomass, and wide canopies (Aniszewski 2010; Keddy 1990). Plants with these traits interfere with their neighbor's ability to collect resources such as light; further inhibiting their development in a form of positive feed-back loop. Similarly, plants with wider and deeper root systems would harvest soil resources more effectively than neighbors; divergent rooting strategies is common among prairie forbs and shrubs, scavenging in spaces void of grass roots (Fargione et al. 2003).

Successful plant recruitment is regulated by numerous stochastic events; thus plants have evolved a number of mechanisms for overcoming these obstacles. If propagation through seed is their sole strategy for reproduction, these species will invest in abundant seed production. These species are sometimes described as 'r-selected' species (Levassor et al. 1990), investing in numerous offspring at low cost to the parent and thereby increasing the probability of having at least one offspring successfully replace the parent. Species that utilize this strategy are often described as weedy, but can include native annuals and biennials that lack vegetative reproduction like pygmy flower (*Androsace septentrionalis*), rock-cresses (*Arabis* spp.), etc. These species can saturate the seed bank and take advantage of the formation of a stochastic niche (i.e. changing microsite) over time. In contrast to these species, are 'K-selected' species, which produce fewer but larger seeds (Levassor et al. 1990). While larger seeds are more energetically costly to produce, they have a higher probability of germinating. This strategy is more important for long-lived organisms (i.e. perennial vegetation) or those who reproduce largely from vegetative means and only require periodic recruitment from seed.

Recruitment from seed banks is often regulated by disturbance events varying in intensity, scale, timing, frequency and duration, which open up a potential niche for establishment. In large-scale disturbances where aboveground biomass is removed over a large area (i.e. following fire or tillage), seedlings will initially be alleviated from competitive stress (Booth and Swanton 2002). In this scenario assembly of the community will follow processes associated with secondary succession (Kinucan and Smeins 1992). Pioneer generalist species with low dormancy, abundant viable seed banks, and weedy habits are the first plants to emerge (Levassor et al. 1990; Tilman 1985). Later seral species tend to be rare or absent, which can impair the rate of succession (Kinucan and Smeins 1992). In the facilitation model of succession (Connell and Slayter 1977), these pioneer species modify the environment, making it more suitable for species of later seres. This assumes the community follows a somewhat linear trajectory, to a point where a stable community will eventually be achieved over time. However, disturbances can alter the trajectory of a community, making the stable endpoint community unattainable (Booth and Swanton 2002) or chaotic (Hastings et al. 1993).

Within grasslands, species can have differential responses to disturbances, or lack thereof, which in turn, can affect the composition and condition of the plant community (Dyksterhuis 1949). When the intensity and frequency of disturbance is low or absent, communities shift towards their climax; in grasslands this often includes plants identified as ‘decreasers’ due to their known sensitivity to disturbance, including grazing by livestock (Dyksterhuis 1949). Intensity of disturbance, environmental stress, and competition can in-part describe the formation of communities by Grime’s CSR theory (1979), where species are classified by their combination of competitive (C), stress tolerance (S), and ruderal (R) life strategies. CSR theory explains why disturbed habitats consist of ruderal species, resource limited environments consist of select species adapted to stress (e.g. low resource availability), and habitats with low disturbance and abundant resources become dominated by competitive species. Connell (1978) explained species diversity responses in the intermediate disturbance hypothesis (IDH), suggesting that maximum species diversity occurred under moderate levels of disturbance where both stress intolerant and tolerant species could co-occur. Rangeland managers widely apply these principles to achieve desired plant communities, including altering plant diversity (Milchunas et al. 1988). Small-scale disturbances such as herbivory can open microsites within the community. Kalamees and Zobel (2002) found plant recruitment from the seed bank often took place within spatial ‘gaps’ of calcareous grassland; this form of regeneration accounted for 36% of plant renewal, with additional recruitment explained by seed rain (46%) and vegetative means (18%).

Certain species can also behave as passengers (take advantage of conditions created by another species) or drivers (make conditions favorable for other species) in community assembly (Helm et al. 2014). Interest in this model has significant applications in the study of alien invasive plant species deleterious to ecosystems; as their introduction can significantly alter the trajectory of communities and facilitate the propagation of itself and other exotic species (Burns 2014; Masters 2014).

Keddy (1990) proposed a centrifugal model of community assembly; where species share a fundamental niche and their growth would be ideal in the preferred ‘central’ habitat characterised by abundant resources and low stress. However, the central community is dominated by the most competitive species; in Keddy’s initial wetland model this is *Typha* (1990). Differences in disturbance and stress cause the community to shift and species find refuge from interspecific competition in a peripheral community (Keddy 1990). Keddy’s model was developed to explain community assembling in wetlands but has also found applications in forests (Keddy and MacLellan 1990) and grasslands (Vujnovic 2000). Vujnovic (2000) applied the centrifugal model to describe the composition of remnant rough fescue grasslands in Alberta and how disturbance encourages invasive species.

In ecology, the mechanisms regulating the patterns and processes of succession are greatly debated and have not been well explained (Tilman 1985). Tilman's theories provide a mathematical and potentially measurable way to track plant species establishment and persistence over time. Tilman's alternative theory for grassland community is the resource ratio hypothesis of plant succession (Tilman 1985), where persistence of species in the community is regulated by the relative availability of resources like water, light, space, nitrogen, phosphorus, potassium, magnesium, trace metals, etc. (Tilman 1985). This model incorporates two important mechanisms, interspecific competition for resources and their long-term limiting supply (Tilman 1985). Species that share a fundamental niche coexist by utilizing fractions of resources complimentary to their neighbors. Further, species occupy unique physical niches within the community's structure (i.e. by possessing different growth forms, canopy height, rooting depths, etc.). Tilman (1985) acknowledges that his model does not account for plant species selection by herbivores, differential colonization abilities, and temperature-dependant growth (i.e. warm-season vs. cool-season grasses). In order for a stable final state in the community to be achieved, resource supply rates need to be locally equilibrated. This means ecosystems with fluctuating inputs of resources (e.g. nitrification, changes in biomass removal, alteration in precipitation with climate change, etc.) will cause species in the community to shift in dominance, thereby affecting the final state (Tilman 1985). These concepts have since been applied to stochastic niche theory (Tilman 2004); where novel species are only incorporated into a community if propagules can survive stochastic mortality and survive on remaining resources. In prairie grassland, established species can inhibit the introduction of new species and individuals through their resources consumption (Fargione et al. 2003). Novel species with similar functional traits will be supressed by existing vegetation (Fargione et al. 2003), but species with novel traits that exploit untapped resources or create resources (i.e. nitrogen fixing species) are more likely to establish.

Common ground exists between these competing theories, as they have each accounted for environmental gradients, competition, and to a lesser extent disturbance. Tilman's (1985) model exclusively used resource gradients, to infer how species will compete for resources, however it has poor predictability when disturbance (change in resources) is imposed as the resulting community will be dependent on the final resource pool. Keddy (1990) and Grime (1979) incorporate both resources and disturbance, Grime's CSR model is simplistic describing the state (hydric to xeric or undisturbed to disturbed) of the ecosystem while Keddy's centrifugal model is more complex taking into account soil properties, competitive hierarchies, types of disturbance, etc. Ideally, a generalized community assembly model would address how functional groups of plants compete for resources among environmental gradients in the fundamental niche and how their interactions will lead to occupation of their realized niche (McGill et al. 2006); which aligns strongly with Keddy's (1990) centrifugal model.

Despite the development of these models, we also have to consider the possibility that the null hypothesis may be true—where assembly is not influenced by competition, functional characteristics, and environmental gradients, but rather that recruitment and survival is stochastic (Gotelli and Graves 1996); driven by demographic factors (i.e. plant age, mortality, etc.) and regional propagule abundance (Fargione et al. 2003). Some aspects of community assembly are stochastic, such as the introduction of a novel species through natural (i.e. wind, birds, herbivores, etc.) or anthropogenic dispersal vectors; however, there also is strong evidence that many mechanisms are regulating the assembly of communities after propagule arrival (Booth and Swanton 2002; Fargione et al. 2003; Keddy 1990; McGill et al. 2006; Tilman 1985).

Concepts of community assembly, such as succession, plant community shifts with disturbance, and transitions to new stable states play an important role in rangeland health monitoring.

2.8 Weed Seed Banks and Invasion

Species with weedy habits tend to be prominent in the seed bank, and their abundance increases with a history of disturbance (Wellstein et al. 2007). Despite the spread of invasive species globally, aspects of weed ecology – especially the contributions of the seed bank to invasion is poorly understood

(Gioria et al. 2014). Invaded communities can exhibit a decrease in the density of native seeds and an overall decrease in species richness (Gioria et al. 2014). High propagule pressure from weedy species also plays a role in overcoming competitive vegetation (Lockwood et al. 2005).

In Alberta enforcement of the *Weed Control Act* requires producers to control or destroy noxious and prohibited noxious weeds, respectively. Noxious weeds of concern to producers include Canada thistle (*Cirsium arvense*), common tansy (*Tanacetum vulgare*), leafy spurge (*Euphorbia esula*), perennial sow thistle (*Sonchus arvensis*), scentless chamomile (*Matricaria perforata*), toad flaxes (*Linaria* spp.), white cockle (*Silene latifolia* subsp. *alba*), and many others, that can opportunistically establish in existing grasslands. Not all weeds of concern are noxious as nuisance weeds can also reduce grassland productivity, as can other undesirable native vegetation that offers few is any benefits for livestock grazing.

Management of weed seed banks in perennial grasslands involves proactive management of the plant community. Reducing the proportion of bare soil is crucial, as this can prevent invasive species from establishing and subsequently populating the seed bank (Clements et al. 2007; Sanderson et al. 2014). In arid communities this often involves protecting the cryptogamic soil crust, which can be degraded under high grazing pressure (Clements et al. 2007). Like cultivated systems, controlling seed input from the existing community can help inhibit seed bank formation, and is achieved through the strategic use of mowing, grazing, or herbicide application (Sanderson et al. 2014). Grazing during the bud or flowering stage of species of concern can improve efficacy (Sanderson et al. 2014). Maintaining functional diversity (i.e. high colonization of all niche space) in plant communities can also reduce their susceptibility to invasion (Renne and Tracy 2007). However, control of local seed input alone does not guarantee weed invasion will not occur, as significant inputs to the seed bank can occur through immigration of seed from outside the local community in the form of seed rain (Booth and Swanton 2002).

2.9 Legumes and Legume Seed Banks

Legumes, from the plant family Fabaceae, have an economically significant role in society, due to their nutritional value and nitrogen fixing ability (Reaume 2009). Roots of legume plants contain *Rhizobia* spp. bacteria within nodules, which fix atmospheric nitrogen (N_2) into organic nitrogen (NO_3^-) (Freedman 2010), a limiting macronutrient for plant growth in most terrestrial ecosystems, including temperate grasslands (LeBauer and Treseder 2008). Organic nitrogen is then transported through the roots and used in protein synthesis. The result of their symbiosis is called biological nitrogen fixation, which supports a large influx of macro nutrients into the base of the food web—primary producers—hence, increasing the amount of biomass an ecosystem can support. Within grazed grasslands, legumes are valued for the increase in forage quality and quantity they convey, particularly in the form of crude protein (Ledgard and Steele 1992).

Recruitment of legumes from the seed bank will be a function of species' biology and their environment. Legume seeds have thick indurate seed coats, which aid in a seed's physical dormancy, but are a barrier to water absorption and are often described as impermeable (Acharya 2006; Baskin et al. 2000; Tracy and Sanderson 2000). Many legumes have a lens (strophiole) adjacent to the hilum (placentation scar on the ovule) that is thin walled. These seeds can become more permeable to water if the lens is degraded (Baskin et al. 2000), thereby aiding germination. *Astragalus cicer, Melilotus alba,* and *Melilotus officinalis* have all been identified as species that imbib water at the lens (Baskin *et al.*, 2000). Breaking dormancy also involves thinning of the seed coat through cold stratification, physical or chemical scarification, heating, and aging, thereby making it easier to absorb water and moisten the embryo (Acharya 2006; Baskin et al. 2000). Embryos of legumes in the subfamily Faboideae (includes: *Astragalus, Dalea, Glycyrrhiza, Lathyrus, Medicago, Melilotus, Oxytropis, Pediomelum,* and *Vicia*), also have properties which contribute to their physiological dormancy (Baskin et al. 2000). These factors contribute to persistence in the seed bank and high legume density in grasslands. Dormant seed banks of legumes are also susceptible to biotic factors that remove seeds from the germinable seed bank such as granivory by microfauna and granivores. Grassland granivores are selective and tend to have a preference for larger-seeded species, with relative selection among legumes [and other plant groups] species specific (Howe and Brown 2000) that in turn influences plant community assembly (Howe and Brown 2001)

2.9.1 Agronomic Legumes

Common legume species found within introduced pastures of western Canada include alfalfa (*Medicago sativa*) and clover (*Trifolium* spp.). Alfalfa is a valued forage crop utilized in hayfields, pastures, and crop rotations. In Alberta common alfalfa (*Medicago sativa)* is prevalent, with yellow alfalfa (*Medicago falcata*) occurring more frequently in southern AB. Common alfalfa has a purple corolla and coiled pods, while yellow alfalfa has a yellow corolla with curved or straight pods; both species have a short head-like raceme bearing the flowers, numerous elongated to prostrate stems, and toothed trifoliate leaves (Moss 2010). White clover is shorter statured (aiding in grazing tolerance) capable of flowering at heights of 5 to 20 cm (Moss 2010). Its leaves are trifoliate, each leaflet has a whitish watermark, and membranous stipules adnate to the petiole (Moss 2010); this species can be distinguished from other *Trifolium* spp. based on leaf characteristics.

Alfalfa is often seeded into newly established pasture mixes where it contributes abundant biomass and improves forage quality (Burity et al. 1989). Alfalfa can fix up to ~200 to 250 kg N ha^{-1} yr^{-1} from atmospheric N_2 (Bell and Nutman 1971; Burity et al. 1989). White clover (*Trifolium repens*) in grazed pastures can fix 55 to 296 kg of N ha^{-1} yr^{-1}, (Ledgard and Steele 1992). Approximately 2% to 26% of biologically fixed nitrogen can transfer to grasses in the community through the decomposition of legume roots and nodules (Burity et al. 1989; Ledgard and Steele 1992). However, alfalfa is also known to be sensitive to grazing (Smith et al. 1988), in part due its preference by cattle, and can therefore decline in abundance with pasture age. While clover is also widely distributed across the Aspen Parkland, and although seeded into many newly established pastures, this species is capable of extensive regeneration (i.e. volunteering) from the soil seed bank, primarily when the soil is disturbed (Barret and Silander 1992). White clover seedlings have relatively high seedling mortality when they germinate in a sod or pasture, but once established, can subsequently propagate through vegetative reproduction via stolons (Barret and Silander 1992). Conditions favouring clover emergence are the availability of microsites with adequate light and moisture (Barret and Silander 1992). White clover is a grazing tolerant species, Tracy and Sanderson (2000) found that cattle grazing can improve its seed bank density. Aboveground, grazing removes taller grasses and increases light availability allowing further propagation (Tracy and Sanderson 2000).

2.9.2 Escaped (Invasive) Agronomic Legumes

Exotic legumes have the potential to become invasive, especially when they establish and reproduce in resource (i.e. nitrogen) limited ecosystems (Riper and Larson 2009). In the process, these species can significantly alter the composition and function of native grasslands creating a nutrient enriched environment that may favor a variety of weedy ruderal species over long-lived native species.

Cicer milkvetch (*Astragalus cicer*; CMV) is an introduced forage most commonly seeded in more mesic regions of western Canada such as the Parkland. However, this legume can also act as a novel invasive in native grasslands, with recent studies pointing to its potential impacts of increasing forage production at the risk of reducing floristic biodiversity and soil carbon (Aniszewski 2010; Carlyle Unpublished). This legume was introduced to North America approximately 85 years ago for its high nutritional value without bloat, long life-span, tolerance to moderately acidic or alkaline soils, winter hardiness, and drought tolerance (Acharya et al. 2006; Peterson et al. 1992). Use of CMV is not very extensive, as it has a particularly slow establishment period, which is caused by a very hard, nearly impermeable seed coat, and subsequent slow seedling development (Acharya et al. 2006). CMV is often used as an alternative to alfalfa (*Medicago sativa*) because it is more frost tolerant and retains its leaves longer, making it a suitable late season forage (Acharya *et al.* 2006).

Characteristics that make CMV potentially deleterious to native grasslands include its growth habit and the modified microenvironment it creates. CMV produces multiple semi-erect to prostrate stems that lay across the soil surface, with a canopy often over 1 m in diameter dense with long pinnate leaves with broad hairy leaflets. Under favorable conditions stems can reach 1.5 m long (Acharya et al. 2006). This results in significant shading (Aniszewski 2010) and increased relative humidity under the canopy. Shading of neighbors makes it a strong competitor when establishing in a new habitat, thereby leading to neighbor loss and potentially eliminating species (Aniszewski 2010). Aniszewski (2010) noted that peak competitive effects occurred in the plant's seventh year of growth when it had reached maximum development. Despite Aniszewski's (2010) findings, he did not think CMV had the potential to become an invasive species outside of its central European range, likely a result of the experiment being conducted in a cultivated field and not among established perennial vegetation. In contrast, CMV has been noted to have invasive properties in the Mixedgrass prairie of SE Alberta on relatively sandy soils and underlain by an elevated water table (Acharya et al. 2006).

Sweet clover (both yellow, *Melilotus officinalis*, and white, *Melilotus alba*) are introduced legumes from Eurasia; commonly found along roadsides and waste ground (Moss 2010). This plant is identified by its racemes of white (*Melilotus alba*) or yellow (*Melilotus officinalis*) flowers extending on a lax elongated rachis (Moss 2010); the leaves are trifoliate with long narrow stipules; these species can be distinguished vegetatively based on stipule length and presence (*M. alba*) or absence (*M. officinalis*) of hairs on the lower leaf surface (Reaume 2009).

In the mid-1900s sweet clover was explored for use as a forage crop (Robinson 1947), and was valued for its salt tolerance (Rogers et al 2008) and ability to withstand waterlogging (Rogers et al. 2008). Moreover, this plant was seeded along many prairie roadways and used in reclamation mixtures for oil and gas disturbance (Simmers and Galatowitsch 2010). Although this plant is regarded primarily as forage, its behavior as an invasive species has been realized more recently. Sweet clover reproduces solely through seed and has a biennial lifecycle. During the first-year seedlings establish a taproot and a small canopy develops; flowering is unlikely but possible at this time (Turkington et al. 1978). A second-year plant produces a large canopy, then flowers and sets seed. By the third year a lignified-skeleton of the previous year's growth remains, and a cluster of seedlings can often be observed growing in close proximity to the parent plant. Individual sweet clover plants can create their own micro-environment and alter the surrounding area by nitrifying the soil, shading neighbors, and increasing relative humidity (Riper and Larson 2009). These characteristics make it a strong competitor against native grasses and forbs, which are often much shorter-statured, and in the case of arid grasslands, adapted to resource limited environments. This process of producing and dropping seed, and facilitating seedling spread, can effectively facilitate invasion of sweet clover together with a number of other (passenger) species that quickly colonize the new micro-site (MacDougall and Turkington 2005; Wolf et al. 2008). In addition, the biennial life cycle of sweet clover can make it an unpredictable supply of forage, as perennials typically offer greater stability in long term forage supply. Furthermore, ingestion of an abundance of sweet clover can have toxic effects (Payne et al. 2015; Turkington et al. 1978).

2.9.3 Native Legumes

Northern grasslands host a wide diversity of native legumes (i.e. *Astragalus* spp., *Dalea* spp., *Lathyrus* spp., *Oxytropis* spp., *Pediomelum* spp., *Thermopsis rhombifolia*, etc.). Native legumes improve forage supply and quality by increasing available nitrogen in the soil, although unlike introduced legumes, they are often at much lower densities, and thus likely provide much smaller overall nitrogen addition. In addition, the retention of these species in grazed grasslands is often overlooked as a supply of forage because many native legumes accumulate secondary plant compounds and can be unpalatable and even toxic to livestock. Nevertheless, select native legume species like *Dalea*, *Vicia*, and a handful of *Astragalus* spp. can be consumed without concern of bloat and other toxic effects (Gunn 1965). Recent research suggests that seeding native legumes into mixed grasslands can improved forage quality, leading to more crude protein than their corresponding native warm-season grasses (McGraw et al. 2004). Interest is growing in the use of seed mixtures containing native grasses and legumes for both perennial forage crops and reclamation mixtures (Jefferson et al. 2005; Mischkolz et al. 2013).

American vetch (*Vicia americana*) is a native legume that is widely adapted in its habitat selection and can be found in xeric native Mixedgrass prairie to mesic grasslands, as well as forest understories, and is phenotypically variable across different environments (Gunn 1965). In xeric environments, American vetch is a relatively short-statured and erect plant that produces few flowers (3-4) per raceme, short tendrils, and thick leaflets (Gunn 1965). In mesic forest understories and introduced pastures, it uses tendrils at its leaf tips to support itself above vegetation, resulting in taller statured plants with broad and thin leaves, and several more flowers per raceme (Gunn 1965). American vetch is sensitive to grazing and tends to be found in grasslands with good range condition (i.e. later seres) (Gunn 1965). American vetch therefore has value as pasture forage and a hay crop (Gunn 1965; Gunn 1979), but its inclusion in seed mixtures is limited.

Another native legume that has significant potential as a forage crop is purple prairie clover (*Dalea purpurea*). This species is native to the Mixedgrass prairie ecoregion, preferring well-drained loamy soils, and often establishing on hill-tops and hill-sides (Moss 2010). Mature plants are described by Moss (2010) as 30 to 80 cm tall with multiple erect or ascending stems, with leaves that are dissected into 3 to 5 linear leaflets. The inflorescence is a compact cylindrical spike 1 to 5 cm long, and the corolla is rose or purple. Fruits are small indehiscent pods containing 1 to 2 seeds, making seed production relatively low per plant. Value and use in forage and reclamation mixtures is currently expanding (McGraw et al. 2004; Mischkolz et al. 2013), as it fixes valuable nitrogen and contributes to native biodiversity.

2.10 Characterization of Seed Banks

Quantifying the persistent seed bank has been approached with a variety of methods, as no optimal strategy has been identified. In part this is because of the challenge associated with identifying the minimum sample size of soil cores needed to adequately represent a given pasture's heterogeneity and accurately quantify composition. Unlike cultivated fields, distributions of seeds in pasture tend to be clustered around the parent plant and these clusters are heterogeneously dispersed across the landscape (i.e. as a Poisson distribution) (Benoit et al. 1989; Bigwood et al. 1988). Thus, large enough sample sizes (either more cores, larger cores, or both) must be sampled to account for this variation (i.e. to overcome within pasture heterogeneity). A wide range of soil core volumes have been used in seed bank research. In most studies, seeds tend to be concentrated in the 1 to 2 cm of soil, which then sharply decreases in abundance with increasing depth (Williams 1984). Some species found in the grasslands of Alberta are an exception as they are more common below the soil surface (Willms and Quinton 1995), such as needle-and-thread grass (*Hesperostipa comata*), pygmyflower (*Androsace* spp.), violet (V*iola* spp.), and silvery cinquefoil (*Potentilla argentea*). Benoit et al. (1989) found there is no significant difference volumetrically between soil core diameters (i.e. all auger sizes detect the same mean abundance of seeds per given volume of soil). Benoit et al. (1989) also found sample variance decreased with decreasing core size and suggested that the use of smaller cores with an increased sampling effort (number of cores) may provide a more accurate estimate of seed bank composition.

2.10.1 Sampling Design and Subsample Size

Optimal sample size for seed bank studies is greatly debated, and in many investigations on grasslands, is believed to inadequately represent the spatial heterogeneity of seed bank composition. Estimates of the aggregate sub-sample size required for obtaining an accurate estimate of seed bank abundance for the entire plant community (or experimental unit) tend to be high, and most studies are instead limited by the green house space, time, and money available to analyze such large volumes of soil. Ambrosio et al. (2004) found that most seed bank sub-sample sizes are over estimated and can be reduced without compromising precision or confidence level associated with the mean. Benoit et al. (1989) reported that sampling up to 75 sub-sample units offered greater precision, but when sampling beyond 75 units, the associated reduction in sampling variance did not compensate for the increased sampling effort incurred.

There are three approaches to sampling seed banks: stratified random sampling, systematic sampling and simple random sampling. In stratified random sampling the sample area (i.e. pasture) is divided into subsections or strata, and sample(s) are randomly taken from each stratum (Benoit et al. 1989). Systematic sampling involves randomly selecting a sampling starting point. From that point a sampling matrix is formed in which the distance between each sample is predetermined (Benoit et al. 1989). In simple random sampling every sample unit has an equal probability of being selected from a predetermined area (Benoit et al. 1989). Sampling variance for clustered sampling is influenced by cluster shape, and sampling variance of systematic sampling is influenced by sampling interval (Benoit et al. 1989). When systematically sampling, the configuration of intervals can cause variation to change with sample size, although eventually variation decreases with larger sample sizes (Benoit et al. 1989). When stratified random sampling, systematic random sampling and simple random sampling methods are compared, systematic sampling is considered superior because it provides a more accurate estimate of the mean with smaller aggregate sub-sample size (Ambrosio et al. 2004).

2.10.2 Germinable Seed Bank Assessments (and Their Limitations)

Quantifying the germinable seed bank involves removing surface soil cores in the field, processing it for the greenhouse, and then observing germination for an extended period. Seedlings are identified to species following germination, counted, removed and discarded. Emergent plants are also observed to make sure they were not derived from vegetative propagules like roots, buds, rhizomes, etc. There are a few strategies that can be used to improve efficacy, such as sieving soil to concentrate the seeds (Ter Heerdt et al.1996) and sieving to remove coarse plant material (including vegetative propagules). In the greenhouse there are two main limitations – space and time (Ter Heerdt et al. 1996). Requirements for all species to germinate may not be met in the time allotted for observation (Coffin and Laurenroth 1989; Ter Heerdt et al. 1996), which can be many months, or even year or more. Species with longer dormancies may remain in the soil and the absence of light and temperature fluctuations found in natural environments could inhibit germination, thereby allowing those species to evade detection.

Studies report germination being highest in the first few weeks; Thompson and Grime (1979) reported 3 weeks and Ter Heerdt et al. (1996) reported 5 to 6 weeks. Longer observation periods allow species with longer dormancy to emerge.

Quantifying the germinable seed bank in the greenhouse has a few inherent caveats that must be kept in mind when interpreting the results. First, the entire diversity and abundance of species dormant in the soil will not be represented, as noted above. Second, the methods used to achieve germination in the greenhouse inevitably will not represent natural disturbances, but perturbation comparable to tilling, which may bias the germination and detection of seeds. Third, aboveground competition is suppressed in the greenhouse, and seeds have ideal condition to break dormancy, therefore, the abundance and diversity of species emerging will not represent typical germination and survival rates occurring in established grasslands.

2.11 Existing Research Gaps in Seed Bank Knowledge

Seed bank surveys are limited in North American rangelands, especially in Western Canada with only a handful of studies (Harker et al. 2000; White et al. 2012; Willms and Quinton 1995) in grazed grasslands. Further study will be required to understand how the seed banks of different communities contribute to the community observed aboveground, as Canadian research in this area lacks in scope and application to many ecological sites. Many studies to date compare the seed bank to the aboveground community and discuss their differences, but offer limited insight into how seedbanks are formed, or test those factors governing plant species recruitment from seed. Research is needed that addresses how the seed bank is assembled, and how filters (including anthropogenic) in the environment effect composition. The influences of management and disturbance history are also important, as past events shape the composition of persistent seed banks and have the potential to influence community trajectory towards either desirable or undesirable endpoints (Renne and Tracy 2007).

Within the Northern Great Plains, I have identified a number of research gaps that impact our understanding and ability to sustainably manage seed banks of northern temperate grasslands. A number of management practices employed by producers such as fertilizer and manure addition, herbicide use, fire, and grazing systems have well understood effects on the aboveground vegetation, but the belowground responses are seldom observed. Herbicides are important management tool to eradicate noxious weeds, the effect on the belowground population of propagules is poorly understood but significantly influences the efficacy of the treatment. As alien invasive plant species, like leafy spurge (*Euphorbia esula*), which is deleterious to rangelands, become more prevalent, an understanding of their population ecology will become very important (Maxwell et al. 1988). The role of seed banks in succession following disturbance, community assembly, and community dynamics (recruitment, maintaining diversity in established grasslands, etc.) is poorly understood. Contributions the seed bank makes to recruitment relative to seed rain and the bud bank under management needs further observation. Seed bank and plant community responses to oil and gas infrastructure installation and reclamation are poorly understood. In fact, little research comparing seed banks among reclaimed disturbances is available.

Population dynamics of specific legume species are poorly understood in perennial grasslands, with the exception of agronomic species like white clover (*Trifolium repens*) (Barret and Silander 1992) and alfalfa (*Medicago sativa*) (Bagavathiannan et al. 2009; 2010; 2011). Agronomic legumes such as sweet clover (*Melilotus* spp.) and cicer milkvetch (*Astragalus cicer*) are poorly represented in the literature; despite their invasive nature little is known about their behavior when introduced to established grasslands. Native legumes are currently growing in recognition as perennial forage crops and potential use in reclamation mixtures (Jefferson et al. 2005; Mischkolz et al. 2013), but many aspects of their biology still require research. Purple prairie clover (*Dalea purpurea*) is a common Mixedgrass prairie legume growing in research interest, but American vetch (*Vicia americana*) has limited research surrounding its biology and ecology despite its continental distribution in North America (Gunn 1965;

Gunn 1970). Overall, seed dormancy and seedling recruitment have been well characterized for agronomic legumes (Acharya 2006; Barret and Silander 1992; Groya and Sheaffer 1981), while this information is generally lacking for native legumes. It is also possible that selective pressures (e.g. anthropogenic influences such selective breeding or managed disturbances) have improved the germinability of agronomic species (De Wet and Harlan 1975; Donohue et al. 2005), which could result in greater recruitment in pastures and improved detection in a germinable seed bank study.

2.12 Conclusion

Current research indicates the ecology of the seed bank is complex, and that different management practices can influence the composition of the seed bank. The most significant trend is that any management decisions that cause disturbance, particularly acute disturbance, can lead to an increase in ruderal species within the seed bank (Wellstein et al. 2007). Grazing plays an important role in maintaining diverse communities both above and belowground (Eriksson and Eriksson 1997; Jacquemyn et al. 2011), emphasizing the importance of herbivores in altering rangeland health and corresponding biodiversity. In general, seed bank surveys that focus on comparing soil seed banks to existing floristic composition have many problems and result in limited new knowledge of seed bank ecology and response to management (Hopfensperger 2007; Sanderson et al. 2007). Consequently, further research is required on seed banks in western Canadian grasslands to better understand those factors regulating seed bank composition and diversity, as well as the importance of seed banks in contributing to above ground plant communities. Although this research attempts to connect many variables affecting seed bank formation, many questions will remain unanswered.

2.13 Literature Cited

Acharya, S.N., Kastelic, J.P., Beauchemin, K.A., and Messenger, D.F. 2006. A review of research progress on cicer milkvetch (*Astragalus cicer* L.). Canadian Journal of Plant Science **86**(1):49-62.

Aniszewski, T. 2010. Canopy behavior of three milkvetch (*Astragalus*) species in acclimation to new habitat. Acta Biologica Cracoviensia Series Botanica **52**(1):45-54.

Allred, B.W., Smith, W.K., Twidwell, D., Haggerty, J.H., Running, S.W., Naugle, D.E., and Fuhlendorf, S.D. 2015. Ecosystem services lost to oil and gas in North America: Net primary production reduced in crop and rangeland. Science **348**(6233):401-402.

Adams, B. W., Ehlert, G., Stone, C., Lawrence, D., Alexander, M., Willoughby, M., Hincz, C., Moisey, D., Burkinshaw, A., Carlson, J. and France, K. 2005. Rangeland health assessment for grassland, forest and tame pasture. Public Lands and Forests Division, Alberta Sustainable Resource Development.

Ambrosio, L., Iglesias, L., Marin, C., and Del Monte, J.P. 2004. Evaluation of sampling methods and assessment of the sample size to estimate the weed seedbank in soil, taking into account spatial variability. Weed Research **44**:224-236.

Archibold, O.W., Ripley, E.A., and Delanoy, L. 2003. Effects of season burning on the microenvironment of fescue prairie in central Saskatchewan. Canadian Field-Naturalist **117**(2):257-266.

Bagavathiannan, M.V., Gulden, R.H., Begg, G.S. and Van Acker, R.C. 2010. The demography of feral alfalfa (*Medicago sativa* L.) populations occurring in roadside habitats in Southern Manitoba, Canada: implications for novel trait confinement. Environmental Science and Pollution Research **17**(8):1448-1459.

Bagavathiannan, M.V., Gulden, R.H. and Van Acker, R.C. 2011. The ability of alfalfa (*Medicago sativa*) to establish in a seminatural habitat under different seed dispersal times and disturbance. Weed Science **59**(3):314-320.

Bagavathiannan, M.V. and Van Acker, R.C. 2009. The biology and ecology of feral alfalfa (*Medicago sativa* L.) and its implications for novel trait confinement in North America. Critical Reviews in Plant Sciences **28**(1-2):69-87.

Bailey, A. W. and Anderson, M. L. 1978. Prescribed burning of a *Festuca-Stipa* grassland. Journal of Range Management **31**(6):446-449.

Bailey, A.W., McCartney, D., and Schellenberg, M.P. 2010. Management of prairie rangeland. Agriculture and Agri-Food Canada.

Bailey, A.W., and Wroe, R.W. 1974. Aspen invasion in a portion of the Alberta Parklands. Journal of Range Management **27**(4):263-266.

Ball, D.A. 1992. Weed seedbank response to tillage, herbicides, and crop rotation sequence. Weed Science **40**(4):654-659.

Barret, J. P. and Silander Jr., J.A. 1992. Seedling recruitment limitation in white clover (*Trifolium repens*; Leguminosae). American Journal of Botany **79**(6):643-649.

Baskin. J.M., Baskin, C.C., and Li, X. 2000. Taxonomy, anatomy and evolution of physical dormancy in seeds. Plant Species Biology **15**:139-152.

Bekker, R.M., Verweij, G.L., Smith, R.E.N., Reine, R., Bakker, J.P., and Schneider, S. 1997. Soil seed bank in European grasslands: does land use affect regeneration perspectives? Journal of Applied Ecology **34**:1293-1310.

Bell, F. and Nutman, P.S. 1971. Experiments on nitrogen fixation by nodulated lucerne. Plant and Soil **35**(1):231-264.

Benoit, D. L., N. C. Kenkel, and Cavers, P.B. 1989. Factors influencing the precision of soil seed bank estimates. Canadian Journal of Botany **67**:2833-2840.

Bigwood, D. W. and Inouye, D.W. 1988. Spatial pattern analysis of seed banks: an improved method and optimized sampling. Ecology **69**(2):497-507.

[BLM] Bureau of Land Management. 2001. Biological soil crusts: Ecology and Management, Technical Reference 1730-2. Denver, CO, USA.

Booth, B.D., and Swanton, C.J. 2002. 50th anniversary—invited article: Assembly theory applied to weed communities. Weed Science **50**:2-13.

Bork, E. W., Adams, B. W., and Willms, W. D. 2002. Resilience of foothills rough fescue, Festuca campestris, rangeland to wildfire. Canadian Field-Naturalist **116**(1):51-59.

Bork, E. W., Grekul, C. W., and De Bruijn, S. L. 2007. Extended pasture forage sward responses to Canada thistle (*Cirsium arvense*) control using herbicides and fertilization. Crop Protection **26**(10):1546-1555.

Briske, D.D. 1996. Strategies of plant survival in grazed systems: a functional interpretation. The Ecology and Management of Grazing Systems. pp 37-67.

Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, K.M., Gillen, R.L., Ash., A.J., and Willms, W.D. 2008. Rotational grazing on rangelands: Reconciliation of perception and experimental evidence. Rangeland Ecology and Management **61**:3-17.

Brummer, E.C. and Moore, K.J. 2000. Persistence of perennial cool-season grass and legume cultivars under continuous grazing by beef cattle. Agronomy Journal **92**(3):466-471.

Buhler, D.D., Hartzler, R.G., and Forcella, F. 1997. Implications of weed seedbank dynamics to weed management. Weed Science **45**:329-336.

Burity, H.A., Ta, T.C., Faris, M.A., and Coulman, B.E. 1989. Estimation of nitrogen fixation and transfer from alfalfa to associated grasses in mixed swards under field conditions. Plant and Soil **114**(2):249-255.

Burns, J.H. 2014. To what degree are invaders drivers or passengers of phylogenetic community structure? Journal of Vegetation Science **25**(6):1311-1312.

Carr, P.M., Poland, W.W., and Tisor, L.J. 2005. Forage legume regeneration from the soil seed bank in western North Dakota. Agronomy Journal **97**(2):505-513.

Caswell, H. 1989. Matrix population models: construction, analysis, and interpretation. Sinauer Associates Inc., Sunderland, Massachusetts.

Cialdella, N., Dobremez, L., and Madelrieux, S. 2009. Livestock farming systems in urban mountain regions: Differentiated paths to remain in time. Outlook on Agriculture **38**(2):127-135.

Clements, D.R., Krannitz, P.G., and Gillespie, S.M. 2007. Seed bank responses to grazing history by invasive and native plant species in a semi-desert shrub-steppe environment. Northwest Science **81**(1):37-49.

Coffin, D.P. and Laurenroth, W.K. 1989. Spatial and temporal variation in the seed bank of a semiarid grassland. American Journal of Botany **76**(1):53-58.

Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. Science **199**(4335):1302-1310.

Connell, J.H. and Slayter, R.O. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. The American Naturalist **111**:1119-1144.

Cox, R. D. and Allen, E. B. 2008. Composition of Soil Seed Banks in Southern California Coastal Sage Scrrub and Adjacent Exotic Grassland. Plant Ecology **198**(1):37-46.

Damschen, E.I., Brudvig, L.A., Haddad, N.M., Levey, D.J., Orrock, J.L., and Tewksbury, J.J. 2008. The movement ecology and dynamics of plant communities in fragmented landscapes. Proceedings of the National Academy of Sciences of the United States of America **105**:19078-19083.

Dornier, A., Pons, V., and Cheptou, O. 2011. Colonization and extinction dynamics of an annual plant population in an urban environment. Oikos **120**:1240-1246.

Dyksterhuis, E.J. 1949. Condition and management of rangeland based on quantitative ecology. Journal of Range Management **2**(3):104-115.

Edwards, A. R., and Younger, A. 2006. The dispersal of traditionally managed hay meadow plants via farmyard manure application. Seed Science Research **16**(02):137-147.

Eschtruth, A.K. and Battles, J.J. 2009. Assessing the Relative Importance of Disturbance, Herbivory, Diversity, and Propagule Pressure in Exotic Plant Invasion. Ecological Monographs **79**:265-280.

Eriksson, A. and Eriksson, O. 1997. Seedling recruitment in semi-natural pastures: the effects of disturbance, seed size, phenology and seed bank. Nordic Journal of Botany **17**(5):469-480.

Facelli, J. M., and Pickett, S.T.A. 1991. Plant litter: its dynamics and effects on plant community structure. Botanical Review **57**:2-32.

Fargione, J., Brown, C.S., Tilman, D. 2003. Community assembly and invasion: An experimental test of neutral verses niche processes. Proceeding of the National Academy of Sciences of the United States of America **100**(15):8916-8920.

Freedman, B. 2010. Environmental Science: A Canadian Prospective. Pearson Canada Inc., Toronto: pg 60-61.

Gardener, C. J., McIvor, J. G., and Jansen, A. 1993. Passage of legume and grass seeds through the digestive tract of cattle and their survival in faeces. Journal of Applied Ecology **30**(1):63-74.

Gioria, M., Jarosik, V., and Pysek, P. 2014. Impact of invasions by alien plants on soil seed bank communities: emerging patterns. Evolution and Systematics **16**:132-142.

Gleason, R.A., Euliss, N.H., Hubbard, D.E., and Duffy, W.G. 2003. Effects of sediment load on emergence of aquatic invertebrate and plants from wetland soil egg and seed banks. Wetlands **23**(1):26-34.

Gonzalez, S. and Ghermandi, L. 2008. Postfire seed bank dynamics in semiarid grasslands. Plant Ecology **199**:175-185.

Goslee, S., Sanderson, M., and Gonet, J. 2009. No persistent changes in pasture vegetation or seed bank composition after fallowing. Agronomy Journal **101**(5):1168-1174.

Gotelli, N.J, and Graves, G.R. 1996. Null Models in Ecology. Smithsonian Institution Press.

Grekul, C. W. and Bork, E. W. 2007. Fertilization augments Canada thistle (*Cirsium arvense* L. Scop.) control in temperate pastures with herbicides. Crop Protection **26**(4):668-676.

Grime, J.P. 1979. Plant strategies and vegetation processes. John Wiley & Sons, New York, USA.

Groya, F.L. and Sheaffer, C.C. 1981. Establishment of sod-seeded alfalfa at various levels of soil moisture and grass competition. Agronomy Journal **73**(3):560-565.

Gunn, C.R. 1965. The *Vicia americana* complex (Leguminosae). Retrospective Theses and Dissertations, Iowa State University. Paper 4086.

Gunn, C.R. 1970. Genus *Vicia* with notes about tribe Vicieae (Fabaceae) in Mexico and Central America. Technical Bulletin 1601, United States Department of Agriculture.

Halfacree, K. 2007. Back-to-the-land in the twenty-first century: Making connections with rurality. Tijdschrift Voor Economische en Sociale Geografie **98**(1):3-8.

Hansen, M. J. and Clevenger, A.P. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation **125**(2):249-259.

Harker, K. N., Baron, V. S., Chanasyk, D.S., Naeth, M.A., and Stevenson, F.C. 2000. Grazing intensity effects on weed populations in annual and perennial pasture systems. Weed Science **48**(2):231-238.

Hastings, A., Hom, C.L., Ellner, S., Turchin, P., and Godfray, H.C.J. 1993. Chaos in ecology: is mother nature a strange attractor? Annual Review of Ecology and Systematics **24**:1-33.

Helsen, K., Hermy M., and Honnay, O. 2015. Changes in species and functional trait composition of the seed bank during semi-natural grassland assembly: seed bank disassembly or ecological palimpsest? Journal of Vegetation Science **26**:58-67.

Henderson, D.C. and Naeth, M.A. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions **7**:639-650.

Helm, A., Kalamees, R., and Zobel, M. 2014. Vegetation patterns and their underlying processes: where are we now? Journal of Vegetation Science **25**(5):1113-1116.

Honnay, O., Jacquemyn, H., Bossuyt, B., and Hermy, M. 2005. Forest fragmentation effects on patch occupancy and population viability of herbaceous plant species. New Phytologist **166**:723-736.

Hopfensperger, K.N. 2007. A Review of Similarity between seed sank and standing vegetation across ecosystems. Oikos **116**(9):1438-1448.

Howe, H.F. and Brown, J.S. 2000. Early effects of rodent granivory on experimental forb communities. Ecological Applications **10**(3):917-924.

Howe, H.F. and Brown, J.S. 2001. The ghost of granivory past. Ecology Letters **4**(4):371-378.

Jacquemyn, H., Mechelen, C.V., Brys, R., and Honnay, O. 2011. Management Effects on the Vegetation and Soil Seed Bank of Calcareous Grasslands: An 11-Year Experiment. Biological Conversation **144**(1):416-422.

Jefferson, P.G., Iwaasa, A.D., Schellenberg, M.P., and McLeod, J.G. 2005. Re-evaluation of native plant species for seeding and grazing by livestock on the semiarid prairie of western Canada. Prairie Forum.

Johnston, A., Smoliak, S. and P.W. Stringer. 1969. Viable seed populations in Alberta prairie topsoils. Canadian Journal of Plant Science **49**(1):75-82.

Kachergis, E., Derner, J., Roche, L., Tate, K., Lubell, M., Mealor, R., and Magagna, J. 2013. Characterizing Wyoming ranching operations: Natural resource goals, management practices and information sources. Natural Resources **4**:45-54.

Kalamees, R. and Zobel, M. 2002. The role of the seed bank in gap regeneration in a calcareous grassland community. Ecology **83**(4):1017-1025.

Keddy, P.A. 1990. Competitive hierarchies and centrifugal organization in plant communities. Perspectives on Plant Competition. Edited by Grace, J.B. and D. Tilman. Academic Press Inc., Sandiego, California, USA.

Keddy, P.A and MacLellan, P. 1990. Centrifugal organization in forests. Oikos **59**(1):75-84.

Kinucan, R.J. and Smeins, F.E. 1992. Soil seed bank of a semiarid Texas grassland under three long-term (36-years) grazing regimes. The American Midland Naturalist **128**(1):11-21.

LeBauer, D.S. and Treseder, K.K. 2008. Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. Ecology **89**(2):371-379.

Levassor, C., M. Ortega, and Peco, B. 1990. Seed bank dynamics of Mediterranean pastures subjected to mechanical disturbance. Journal of Vegetation Science **1**(3):339-344.

Ledgard, S.F. and Steele, K.W. 1992. Biological nitrogen fixation in mixed legume/grass pastures. Plant and Soil **141**:137-153.

Lockwood, J.L., Cassey, P. and Blackburn, T. 2005. The role of propagule pressure in explaining species invasions. Trends in Ecology & Evolution **20**(5):223-228.

López-Mariño, A., Luis-Calabuig, E., Fillat, F., and Bermudez, F.F. 2000. Floristic composition of established vegetation and the soil seed bank in pasture communities under different traditional management regimes. Agriculture, Ecosystems & Environment **78**(3):273-282.

Ma, M., Zhou, X., and Du, G. 2010. Role of soil seed bank along a disturbance gradient in an alpine meadow on the Tibet Plateau. Flora-Morphology, Distribution, Functional Ecology of Plants **205**(2):128-134.

MacDougall, A.S. and Turkington, R. 2005. Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology **86**(1):42-55.

Malo, J.E., and Suárez, F. 1995. Establishment of pasture species on cattle dung: the role of endozoochorous seeds. Journal of Vegetation Science **6**(2):169-174.

Martin, R.E., Miller, R.L, and Cushwa, C.T. 1975. Germination response of legume seeds subjected to moistand dry heat. Ecology **56**:1441-1445.

Masters, J.A. 2014. Invasive plants as drivers and passengers of community change in a disturbed urban forest (Doctoral dissertation, University of Louisville).

Mayor, J. P and Dessaint, F. 1998. Influence of weed management strategies on soil seedbank diversity. Weed Research **38**:95-105.

Maxwell, B.D., Wilson, M.V., and Radosevich, S.R. 1988. Population modeling approach for evaluating leafy spurge (*Euphorbia esula*) development and control. Weed Technology **2**(2):132-138.

McGill, B.J., Enquist, B.J., Weiher, E., and Westoby, M. 2006. Rebuilding community ecology from functional traits. Trends in Ecology and Evolution **21**(4):178-185.

McGraw, R.L., Shockley, F.W., Thompson, J.F, and Roberts, C.A. 2004. Evaluation of native legume species for forage yield, quality, and seed production. Native Plants Journal Fall.

Miller, A.J., Bork, E.W., Hall, L.M., and Summers, B. 2015. Long-term forage dynamics in pastures sprayed with residual broadleaf herbicide: A test of legume recovery. Canadian Journal of Plant Science **95**(1):43-53.

Milchunas, D.G., Sala, O.E. and Lauenroth, W.K. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. The American Naturalist **132**(1):87-106.

Mischkolz, J.M., Schellenberg, M.P., and Lamb, E.G. 2013. Early productivity and crude protein content of establishing forage swards composed of combinations of native grass and legume species in mixed-grassland ecoregions. Canadian Journal of Plant Science **93**(3):445-454.

Molano-Flores, B. 2012. Diaspore morphometrics and self-burial in *Hesperostipa spartea* from loam and sandy soils. The Journal of the Torrey Botanical Society **139**(1):56-62.

Molles, M.C., and Cahill, J.F. 2008. Ecology: concepts and applications: Canadian edition. McGraw-Hill, Dubuque, IA.

Moore, R.J. 1975. The biology of Canadian Weeds. 13. *Cirsium arvense* (L.) Scop. Canadian Journal of Plant Science **55**:1033-1048.

Moss, E.H. 2010. Flora of Alberta, 2nd ed. University of Toronto Press, Toronto.

Neto, M.S., Jones, R.M. and Ratcliff, D. 1987. Recovery of pasture seed ingested by ruminants. 1. Seed of six tropical pasture species fed to cattle, sheep and goats. Australian Journal of Experimental Agriculture **27**(2):239-246.

Otfinowski, R., Kenkel, N.C., and Van Acker, R.C. 2008. Reconciling seed dispersal and seed bank observations to predict smooth brome (*Bromus inermis*) invasions of northern prairie. Invasive Plant Science and Management **1**:279-286.

Payne, J., Livesey, C., and Murphy, A. 2015. Cattle Poisoning: Principles of Toxicological Investigations. Bovine Medicine, 3rd ed. John Wiley & Sons ltd. Sussex, UK: pg 211-224.

Peterson, P.R., Scheafer, C.C., and Hall, M.H. 1992. Drought effects on perennial forage legume yield and quality. Agronomy Journal **84**(5):774-779.

Petherbridge, W.L. 2000. Sod salvage and minimal disturbance pipeline reclamation techniques: implications for native prairie restoration. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Pleasant, J.M.T., and Schlather, K.J. 1994. Incidence of weed seed in cow (*Bos* sp.) manure and its importance as a weed source for cropland. Weed Technology **8**(2):304-310.

Poiani, K.A. and Johnson, W.C. 1988. Evaluation of the emergence method in estimating seed bank composition of prairie wetlands. Aquatic Botany **32**:91-97.

Portnoy, S. and Willson, M.F. 1993. Seed dispersal curves: behavior of the tail of the distribution. Evolutionary Ecology **7**(1):25-44.

Province of Alberta. 2010. Weed Control Act. Her Majesty the Queen in the Right of Alberta, Edmonton.

Reaume, T. 2009. 620 Wild Plants of North America. University of Toronto Press, Toronto: pg 294 and 306.

Ren, L. and Bai, Y. 2016a. Smoke and ash effects on seedling emergence from germinable soil seed bank in Fescue Prairie. Rangeland Ecology and Management **69**(6):499-507.

Ren, L. and Bai, Y. 2016b. Smoke originating from different plants has various effects on germination and seedling growth of species in Fescue Prairie. Botany **94**(12):1141-1150.

Ren, L. and Bai, Y. 2017. Burning modifies composition of emergent seedlings in fescue prairie. Rangeland Ecology and Management **70**(2):230-237.

Renne, I.J. and Tracy, B.F. 2007. Disturbance persistence in managed grasslands: shifts in aboveground community structure and the weed seed bank. Plant Ecology **190**(1):71-80.

Robinson, D.H. 1947. Leguminous forage plants, 2nd ed.

Rogers, M.E., Colmer, T.D., Frost, K., Henry, D., Cornwall, D., Hulm, E., Deretic, J., Hughes, S.R., and Craig, D. 2008. Diversity in the genus *Melilotus* for tolerance to salinity and waterlogging. Plant and Soil **304**(1-2):89-101.

Romo, J.T. and Gross, D.V. 2011. Preburn history and seasonal burning effects on the soil seed bank in the Fescue Prairie. The American Midland Naturalist **165**(1):74-90.

Rowan, R.C. 1994. Are small-acreage livestock producers real ranchers? Rangelands **16**(4):161-166.

Sanderson, M.A., Stout, R., Goslee, S., Gonet, J., Smith, R.G. 2014. Soil seed bank community structure of pastures and hayfields on an organic farm. Canadian Journal of Plant Science 10.4141/CJPS2013-288.

Sanderson, M. A., S. C. Goslee, K. D. Klement, and Soder, K.J. 2007. Soil seed bank composition in pastures of diverse mixtures of temperate forages. Agronomy Journal **99**:1514-1520.

Sayre, N.E. 2004. Viewpoint: The need for qualitative research to understand ranch management. Journal of Range Management **57**:668-674.

Scheffler, EJ. 1976. Aspen forest vegetation in a portion of the east-central Alberta parklands. M.Sc. thesis, University of Alberta, Edmonton, Alberta.

Schwinning, S. and Weiner, J. 1998. Mechanisms determining the degree of size asymmetry in competition among plants. Oecologia **113**(4):447-455.

Simmers, S.M., and S.M. Galatowitsch. 2010. Factors affecting revegetation of oil field access roads in semiarid grassland. Restoration Ecology **18**(s1):27-39.

Smith, S.R., J. H. Bouton and Hoveland, C.S. 1988. Alfalfa persistence and regrowth potential under continuous grazing. Agronomy Journal **81**:960-965.

Smoliak, S. 1974. Range vegetation and sheep production at three stocking rates on *Stipa-Bouteloua* prairie. Journal of Range Management **27**(1): 23-26.

Tannas, K. 2004. Common plants of the western rangelands: Volume 3: Forbs. Alberta Agriculture, Food and Rural Development, Her Majesty the Queen in the Right of Alberta, Edmonton.

Teague, R. Provenza, F., Kreuter, U., Steffens, T., and Barnes, M. 2013. Multi-paddock grazing on rangelands: Why the perceptual dichotomy between research results and rancher experience? Journals of Environmental Management **128**:699-717.

Ter Heerdt, G.N.J, Verweij, G.L., Bekker, R.M., and Bakker, J.P. 1996. An improved method for seed-bank analysis: seedling emergence after removing soil by sieving. Functional Ecology **10**:144-151.

Tilman, D. 1985. The resource-ratio hypothesis of plant succession. The American Naturalist **125**(6):827-852.

Tilman, D. 2004. Niche trade-offs, neutrality, and community structure: A stochastic theory of resource competition, invasion, and community assembly. Proceeding of the National Academy of Sciences of the United States of America **101**(30):10854-10861.

Thomas, A. G. 1985. Weed survey system used in Saskatchewan for cereal and oilseed crops. Weed Science **33**(1):34-43.

Thompson, K. and Grime, J.P. 1979. Seasonal variation in the seed banks of herbaceous species in ten contrasting habitats. Journal of Ecology **67**(3):893-921.

Thompson, K., Band, S.R., and Hodgson, J.G. 1993. Seed size and shape predict persistence in soil. Functional Ecology **7**(2):236-241.

Thompson, K., Bakker, J.P., and Bekker, R.M. 1997. The soil seed banks of north west Europe: Methodology, density and longevity. Cambridge University Press, Cambridge, UK.

Tracy, B.F., and Sanderson, M.A. 2000. Seedbank diversity in grazing lands of the Northeast United States. Journal of Range Management **53**(1):114-118.

Turkington, R. A., Cavers, P. B., and Rempel, E. 1978. The biology of Canadian weeds: 29. *Melilotus alba* Desr. and *M. officinalis* (L.) Lam. Canadian Journal of Plant Science **58**(2):523-537.

Van Riper, L.C. and Larson, D.L. 2009. Role of invasive *Melilotus officinalis* in two native plant communities. Plant Ecology **200**:129-139.

Vujnovic, K., Wein, R., and Dale, M.R.T. 2000. Factors determining the centrifugal organization of remnant Festuca grassland communities in Alberta. Journal of Vegetation Science **11**:127-134.

Wienhold, C.E. and Van der Valk, A.G. 1989. The impact of duration on the seed banks of northern prairie wetlands. Canadian Journal of Botany 67(6):1878-1884.

Wellstein, C., Otte, A., and Waldhardt, R. 2007. Seed bank diversity in mesic grasslands in relation to vegetation type, management and site conditions. Journal of Vegetation Science **18**:153-162.

White, S.R., Bork, E.W., Karst, J., and Cahill, J. 2012. Similarity between grassland vegetation and seed bank shifts with altered precipitation and clipping, but not warming. Community Ecology **13**(2):129-136.

Williams, E.D. 1983. Germinability and enforced dormancy in seeds of species of indigenous grassland. Annals of Applied Botany **102**:557-566.

Williams, E.D. 1984. Changes during 3 years in the size and composition of the seed bank beneath a long-term pasture as influenced by defoliation and fertilizer regime. Journal of Applied Ecology **21**:603-615.

Willms, W.D., and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Willms, W.D., Smoliak, S., and Dormaar, J.F. 1985. Effects of stocking rate on a rough fescue grassland vegetation. Journal of Range Management **38**(3):220-225.

Wolf, J. J., Beatty, S.W., Carey, and G. 2008. Invasion by sweet clover (*Melilotus*) in montane grasslands, Rocky Mountain National Park. Annals of the Association of American Geographers **93**(3): 531-543.

Wright, H.A., and Bailey, A.W. 1982. Fire ecology: United States and southern Canada. John Wiley & Sons.

Xiong, S. and Nilsson, C. 1999. The effects of plant litter on vegetation: a meta-analysis. Journal of Ecology **87**(6):984-994.

Zhan, X., Li, L., and Cheng, W. 2007. Restoration of *Stipa krylovii* steppes in Inner Mongolia of China: Assessment of seed banks and vegetation composition. Journal of Arid Environments **68**(2):298-307.

Study I

Linking plant community, seed bank, soil, and rangeland health's response to producer management in North Central Alberta's Aspen Parkland

In Chapters 3 through 5 multiple multivariate data sets were tied together, including: producer management, rangeland health, soil properties, the aboveground plant community, and belowground seed bank composition. Chapter 3 summarizes rangeland health and producer management. Chapter 4 describes the influence of pasture management and disturbance history on rangeland health, plant community composition, and environmental variables such as soil properties and ground cover. Chapter 5 examines the influence of management and disturbance on the seed bank, acknowledging that these influences are mediated by plant community responses and the microenvironment in which seeds are incorporated into the seed bank.

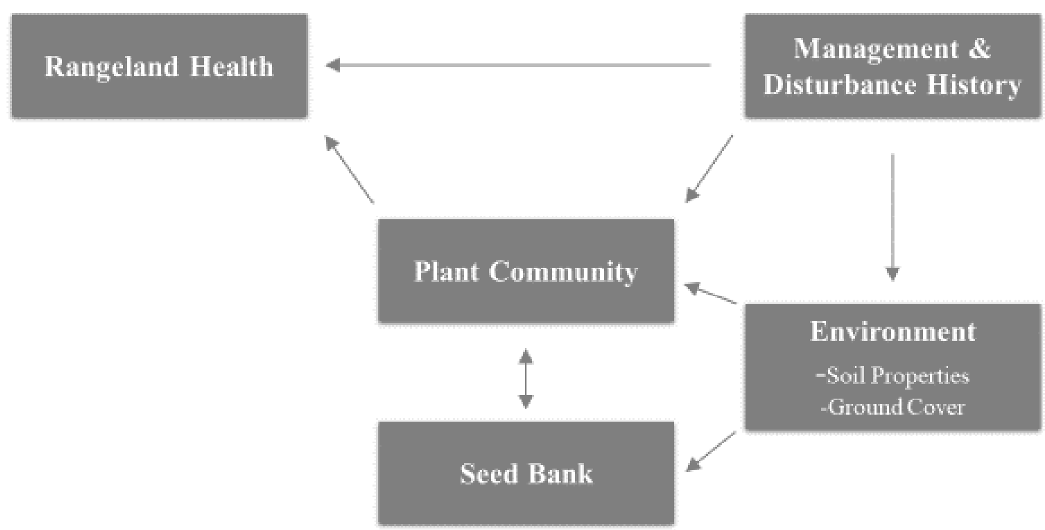


Figure i. Schematic of multivariate data sets examined in the pasture management and seed bank survey summarized for Study I in Chapters 3 through 5. Directional influences of management and the abiotic environment on rangeland health, plant communities, and seed banks are emphasized with arrows.

Chapter 3

Pasture Management History and Corresponding Range Health in Alberta's Parkland

3.1 Abstract

Little information exists on the management history and associated range health of grazing lands in North Central Alberta. We conducted a producer survey of pasture managers and field surveys of range health to address this gap. Pastures were comprised primarily of introduced (tame) grassland in healthy condition, dominated by introduced cool-season grasses and forage legumes. While many grasslands were once cultivated, a subset were identified as non-cultivated, and others had an unknown history, likely due to changes in ownership. Pastures that had never been cultivated were managed with relatively responsible stocking rates (2.14 (±2.91) AUM/ha), when compared to pastures that had been cultivated (6.18 (±0.91) AUM/ha). Forage mixtures were common including grasses like brome (*Bromus inermis* and *B. biebersteinii*), orchard grass (*Dactylis glomerata*), and timothy (*Phleum pratense*), together with legumes like alfalfa (*Medicago sativa*) and clovers (*Trifolium hybridum* and *T. repens*). Farms with horses or mixed herds (i.e. horses and cattle) demonstrated a tendency to practice continuous grazing. Among continuously grazed pastures, some were grazed year-round and supplemental hay was often used. When pastures were grazed during the growing season, we found no significant difference in grazing season length for rotational and continuous systems. Stocking rates of both continuously and rotationally grazed systems were similar, while rotational systems had higher stocking densities. Very high stocking rates were found on pastures used year-round (19.54 (±2.03) AUM/ha) and through the dormant season and winter (20.29 (±3.10) AUM/ha), when compared to growing season use (5.00 (±0.66) AUM/ha). Management inputs were variable, with manuring and harrowing of pastures common, and fertilizing, overseeding, and aerating of soil used less frequently. Herbicide application within the last 3 years occurred on 15.7% of pastures, lower than expected considering 83.3% of pastures contained noxious weeds. Herbicide products containing synthetic auxins (group 4) were those most commonly applied. Ubiquitous presence of noxious weeds reduced rangeland health scores, with 32.4% of pastures

identified as having heavy infestations. Use of prescribed fire was locally rare; when wildfire and evidence of burned woody debris in the soil were combined, 36.3% of pastures had a history of fire. Industrial or infrastructural disturbances in the form of access roads, railways, pipelines, well sites, or mineral extraction, were reported on 48.3% of pastures in the second year of the survey. Overall, these results suggest pastures in the Parkland of western Canada are highly disturbed grasslands in a fragmented and subdivided landscape, where landscapes are managed on small scales by a diversity of people with different management strategies. Tame pasture is known to be prone to overstocking, it is evident that livestock owners in the Parkland are likely exceeding the grazing capacity of their pastures.

3.2 Introduction

Alberta has 40.3% of the Canadian breeding herd of beef cows (Canadian *Agriculture Census* 2011); in Canada the average herd size is 63 head, meaning the majority of cattle producing farms are relatively small operations (i.e. <100 animals). In Alberta, 55% of farms (N=23,855 of 43,234 total) reported they had 'natural land for pasture', covering 6,435,825 ha (Statistics Canada 2014). The Central Parkland Natural Subregion of central and eastern Alberta exceeds 50,000 km^2 and is well known for being an area of diversified agriculture, with fertile soils (e.g. Black Chernozems) giving rise to an abundance of annual cropping (Government of Alberta 2013) (Fig. 3.1). Remaining pockets of pasture land are often used to support small to medium sized cow/calf operations, a trend that is at least partly exacerbated by the increasing presence of hobby farmers (smaller farms, used for pleasure or to supplement income, rather than for primary income) geographically situated near metropolitan regions. Few native grassland remnants persist, and the management of these pockets is important as the Parkland's native grassland communities historically dominated by plains rough fescue (*Festuca hallii*) continue to decline with grazing pressure and anthropogenic disturbance which increases the cover of invasive grass species like smooth brome (*Bromus inermis*) (Sinkins and Otfinowski 2012) Kentucky bluegrass (*Poa pratensis*) (De Keyser et al. 2015; Tannas 2011; Tannas et al. 2015). In contrast to these changes, perennial pastures are expanding into the adjacent boreal, Dry Mixedwood natural subregion, following forest clearing, and there is an ongoing response to the rising demand for arable land in the region (Young et al. 2006).

Management of grazing land for cattle is a product of complex sociological, economic, and ecological factors. Understanding the management regimes employed on private land are important, as management effects plant community composition and productivity (Willms and Jefferson 1993), seed bank characteristics (Johnston et al. 1969; Willms and Quinton 1995), soil properties (Baron et al. 2001;Donkor et al. 2002; Dormaar et al. 1997), aspects of ecosystem functions and services like nutrient cycling (Baron et al. 2002; Naeth et al. 1991) and biodiversity (Tallowin et al. 2005), as well as range condition and associated rangeland health (Willms et al. 1985). While some impacts of management are more acute (e.g. cultivation or herbicide application), grazing practices can influence plant communities more subtly (e.g. type of livestock, season of grazing, grazing system), eventually leading to marked changes in soil and vegetation properties over time (Willms et al. 1985).

Despite the importance of management practices in regulating pasture ecosystem characteristics, many studies are limited to observing one or two select aspects of management in controlled greenhouse or field experiments, such as mowing (Fulkerson and Michell 1987), herbicide application (Grekul and Bork 2007), fertilization (Malhi et al. 2000), or stocking rate (Willms et al. 1985) in various combinations. While these studies are useful in linking pasture soil and vegetation responses to select management activities, they are unlikely to fully represent the complex array of management activities taking place on northern temperature pastures, including on highly fragmented pastures of the Central Parkland and Dry Mixedwood in western Canada. Therefore, it is important to study ecosystem and grassland community responses in relation with pastoral management actions occurring on typical parkland landscapes. However, few studies have been conducted on a large scale examining the impact of contemporary management activities and previous surveys reporting on pasture management in western Canadian rangelands are rare (Chorney and Josephson 2000; Popp et al. 2004).

In Alberta, the Rangeland Health Assessment (RHA) tool (Adams et al. 2005) was developed to help agrologists, producers, reclamation specialists, etc. assess the condition of rangeland plant communities under management regimes (i.e. grazing, industrial disturbance, etc.). A rangeland health score is assigned based on an assessment of plant community characteristics such as the cover and presence of desired plant community components (structural layers, desirable forages, and cover of noxious weeds), hydraulic function (amount of littler), and evidence of erosion. We predict that rangeland health will be influenced by producer management and will aid in the interpretation of plant community responses to management factors in later chapters.

Over the course of two years, 2012 and 2013, we characterized, and quantified the history and management activities of 102 pastures in north central Alberta's central parkland and adjacent boreal mixedwood regions. The large sample size ensured pastures varied in age (recently seeded pastures dominated by productive forages to mature pastures with grazing tolerant species and uncultivated fields) and management history. We documented the available history of these pastures (cultivation, forage species seeded, fire and land use history), type and number of livestock and associated grazing systems and the management inputs producers employed. Producer surveys were followed by a rangeland health assessment. For this chapter, our objectives were to 1) summarize survey responses and identify potential management regimes, and 2) summarize meaningful trends from the rangeland health assessment (RHA scores will be analyzed in greater detail in Chapters 4 and 5).

3.3 Methods

3.3.1. Study Sites

We surveyed a total of 102 pastures during 2012 (N=44) and 2013 (N=58) between May 24 and July 6, distributed across 4 counties (Leduc, Parkland, Strathcona, and Sturgeon County) immediately surrounding the city of Edmonton, Alberta (Figure 3.1). This sampling area was located in north central Alberta's Central Parkland natural subregion, characterized by Black Chernozemic soils (i.e. organic matter of 4-10%), and receives 445 mm of precipitation annually, with about 77% falling during the growing season (April through September) (Fig. 3.2). About half of the pastures sampled occurred in the Central Parkland (N=50), while the remainder occurred within the neighboring boreal natural subregions: Dry Mixedwood (N=50) and Central Mixedwood (N=2). Although precipitation levels are similar, soils in the latter regions are lower in organic matter, resulting in soils varying from Eluviated Black Chernozems to Gray Luvisols. The previously cultivated and seed pastures within the boreal zone resemble the Parkland pastures in composition (Donkor et al. 2002). The large sample size ensured a wide range of pasture types were represented, including old growth pastures (often *Trifolium* spp. dominated) and high-performance pastures containing *Medicago* spp., with a corresponding wide range of management activities.

Sampled pastures were selected at random, acquired through a variety of methods including: consultation with the counties, driving roadsides to visually identify suitable fields, and in some cases, managers referred us to neighbors and family. Suitable pastures had to fit a 260 m long transect, with suitable buffer zones from wetlands, forests, and fence lines (outlined in Chapter 4.3.3 describing the plant community survey) meaning pastures had to be a minimum ~ 10 acres, with larger pastures given preference. If a producer owned or rented multiple pastures, duplicate pastures were only sampled if they were separated spatially (by at least 800 m), although select exceptions (N=2) were made if management was distinctly divergent (i.e. a previous cultivated vs. non-cultivated field; or pastures seeded with different forage mixtures). Acquisition of sites was further constrained by the willingness of landowners to grant permission once their land was identified as a candidate study site, although this happened relatively infrequently with a handful of landowners directly prohibiting entry (N<5). There were also a number of cases where we had to pass on certain potential sampling areas when we were not able to achieve contact with the appropriate people (i.e. we spoke to the wife or children, left consent forms and a survey, but were never invited back; N<50), and many potential sites were visited and revisited, but no owner was at home or answered the door (N<100). Our surveying typically occurred on week days ~ 7

AM to 5 PM, which may have resulted in under recruitment of pasture managers possessing day jobs or busy with other farm activities. Reception from land owners was relatively positive if they were working in their yard at the time of first contact.

3.3.2. Producer Survey

Landowners were surveyed using an in-person interview (see Appendix 3.1), approved by the Research Ethics Office at the University of Alberta, designed to identify historical and current land use practices on individual pastures. Surveys were intended to identify all key management activities that may influence the soil, plant community and associated seed bank composition (discussed in later chapters). Initially, managers were asked how long they managed the pasture in question; when applicable they identified how long the land had been in the family, or under their management, and provided the time of last cultivation. For some pastures that were managed over decades, times of last cultivation were estimates. If land had not been cultivated, or the date of last cultivation was unknown (often the case with grazing lease holders or when the land was cultivated before their possession), this was recorded as well. Other data on management collected included grazing history (number of animals, type of herbivore and timing of use), whether the land had been seeded to introduced forages, when the pasture was last cultivated, fertilized (chemical or manure), or sprayed with herbicide in the last 3 years, and whether the pasture had been otherwise disturbed (burned, pest control, oil and gas disturbance, etc.). Other information may have been volunteered by producers based on their familiarity with management history (i.e. organic management). A final section was allowed for ‘other’ comments on management, where unique management actions or concerns were recorded (i.e. stewardship awards, reclamation concerns, intensive rotational grazing, etc.). When participants were unclear with certain terminology or our motivation for asking, these aspects were clarified. Participants were also given the option of requesting a summary of study results.

During 2013, minor amendments were made to the survey to ask specific questions about weed management (i.e. herbicide product, target weeds, year last treated), determine whether animals had been fed hay in the pasture, manuring, and gather information on pasture/paddock size.

Data regarding farm size, sampled pasture size, and total grazing area were not collected during the interview. In 2016 farm size was inferred from satellite images and county land ownership maps for the legal-land descriptions collected during the survey. Allotments under 80 ac were defined as small holdings and included acreages and smaller hobby farms. Other classifications by lot size included small farms (80-160 ac), medium farms were the standard quarter-section in size and ownership of adjacent allotments was absent (160 ac), and larger farms owned or rented multiple allotments (>160 ac). As actual farm size remained unknown, we were focused on our experimental unit (the pasture) at the time of sampling.

3.3.3. Rangeland Health

Vegetation and soil conditions within each pasture had rangeland health assessment (RHA) within 2 days of completing the producer survey, using the Alberta Environment and Sustainable Resource Development (now Alberta Environment and Parks) Tame Pasture Health Assessment form, which evaluates the abundance of desirable forages (including legumes), weed abundance, site stability and soil erosion, hydrologic function, and nutrient cycling. First the plant community was identified as tame (i.e. grassland dominated by introduced forage grasses) or modified-tame (i.e. included native forbs and grasses, with less than 50% introduced forage cover). Scores are assigned for each category/question, and then tallied to arrive at a total score (see Appendix 3.2), which fell into one of 3 categories (healthy, >75%; healthy with problems, 50-74%; unhealthy, <50%). Score card information allows for both the diagnosis of problematic conditions, and the identification of improvements needed. At the end of the RHA the trend or trajectory of the community (i.e. upward, downwards, stable, or unknown) was assessed, although without a previous reference point the exact trajectory of each community was unknown. Plant community cover and litter abundance were assessed within a 0.25 m^2 (50 x 50 cm) quadrat, while landscape features (soil erosion, anthropogenic bare ground, etc.), and noxious weed and woody species presence, cover, and density, were noted across the transect and pasture.

3.3.4. Stocking Rate

Based on the information given during the interview and aerial photos we were able to calculate stocking rates and densities for 78.4% (n=80) and 80.4% (n=82) of the sampled pastures, respectively. In 2013 we asked producers to describe the number of paddocks used and the area, this question was often answered with little detail as many people could not recall the proportion of their farm in pasture and the sizes of paddocks. Our questions were also designed to ask about the management of the particular pasture we were sampling, but it was apparent that producers often provided a hybrid of information on that pasture and their overall management. For example, we wanted to know the duration of grazing on one pasture but often received a description of their length of grazing season for their operation. Smaller farms that used single pasture systems contained the most adequate descriptions of herd sizes and grazing areas. When rotational grazing was used we calculated the stocking rate two ways, 1) the total grazing area of a farm (determined from aerial photos), herd size, and duration of growing season were used, or 2) when the allotment contained one pasture, often the case when pastures were rented or part of larger operations, the information of duration on that specific pasture and herd size were provided. If pastures had been deferred from grazing over a period of years, the stocking rate of 0 AUM/ha was assigned. Herd sizes included the numbers of different types of livestock, but not the breeds chosen. Thus, we assumed that the animal unit equivalents (AUE) of each livestock type were equal. The AUEs used are as follows: cows and cow-calf pairs = 1.2 AUE, bulls and horses = 1.5 AUE, yearling = 1 AUE, pony = 0.6 AUE, donkey = 0.55 AUE, calf = 0.5 AUE, sheep = 0.2 AUE, and alpaca = 0.1 AUE. When pastures were described as grazed ‘all summer’, often the case with continuous grazing we assumed this was equal to 5.2 months, which was mean length of growing season calculated when adequate information was provided (discussed in 3.5.1.3).

3.4 Statistical Analysis

Producer survey and pasture field data were summarized to provide a quantitative summary (i.e. frequency distribution) on the management and vegetation/soil conditions associated with all pastures. Data were pooled across years because management was not expected to change given that the surveys reflected ongoing activities of landowners over the long-term. Similarly, plant and soil data from perennial pastures were expected to be relatively consistent during the 2 years.

Adequate information on grazing season length was acquired for pastures rotationally (n=29) and continuously (n=26) grazed during the growing season. Median grazing season length was compared using a Kruskal-Wallis test in ‘R’ statistical software (R Core Team 2017) using package *agricolae* (De Mendiburu 2017) ($P<0.05$). Stocking rate and density was log transformed and tested using one-way ANOVA with type III sums of squares (SS) in R against management factors, least squared (LS) means and standard error were found for significant results with Bonferroni corrected contrasts using R software package *lsmeans* (Lenth 2016). In order to identify producers using similar management practices, and link management to rangeland health, we conducted a multiple correspondence analysis (MCA), which uses Euclidean distance to partition categorical data, and was used to ordinate survey data (Greenacre and Blasius 2006). MCA is an analytical tool often used in sociological research (Greenacre and Blasius 2006). This was performed in R using the package *ade4* (Dray and Dufour 2007) and *FactoMineR* (Le et al. 2008). Finally, a cluster dendrogram was performed on the scores of the first 3 MCA axes to show distinct groupings of management regimes reflecting common producer behavior (based on the MCA).

3.5 Results

3.5.1 Management Activities from Producer Surveys

3.5.1.1 Land Ownership

Of the 102 pastures surveyed, we spoke to 73 separate land managers, who were asked a series of questions about ownership, land use history and recent management practices. Of the respondents, 31.5% were female. It was also noted during the interview process that two participants had received stewardship awards, which were displayed on the home quarter. When granted access to multiple pastures, we sampled 2 to 5 (6 in the case of the Blackfoot Grazing Reserve). Across all pastures surveyed, 10.8% of grasslands examined were on crown land belonging to the county, provincial grazing reserve or were designated a natural area (where we sampled an abandoned pasture that was still swathed for hay); with the balance (89.2%) privately owned. On privately owned land (N=91), 7.7% of pastures were rented. Management of natural areas and grazing reserves was primarily the responsibility of the land owner, while management on county land was the responsibility of the lease holder. When lease holders of county land and privately-owned land were combined, 10.8% of all grazed pastures sampled were managed primarily by the renter.

With the study area's close proximity to the city of Edmonton (i.e. within a 50 km radius), large farms with large numbers of livestock were not common. Instead, some pastures (7.8%) were on acreages and small hobby farms (<80 acres); note that an additional 2 pastures were on land units less than 80 acres in size but were rented by larger cattle operations. Sampling pastures on smaller holdings were typically avoided (Fig. 3.3). Based on producer responses to land tenure, the average age of pastures since last cultivation was 20.4 years (n=71), this excluded pastures with unknown and long-term histories, as well as pastures never broken (Fig. 3.4). For privately owned land (n=85), participants reported that land had been in their possession or their immediate families for an average of 39.5 years.

3.5.1.2 Cultivation and Seeding History

The majority of pastures were identified as tame at 88.2% (n=90), while modified-tame communities accounted for 11.8% (n=12). Of the modified-tame communities, 8 pastures were recorded as never previously cultivated, while 2 had an unknown history and 2 were identified as previously cultivated. Overall 7.8% of pastures were cultivated, 75.5% were confirmed to have been cultivated with remainder have an unknown or uncertain history. Non-cultivated pastures were associated with the lowest stocking rate (2.14 (±2.91 SE) AUM/ha), markedly lower than the stocking rate of pastures that had known and unknown cultivation histories (6.18 (±0.91) AUM/ha and 10.91 (±2.06) AUM/ha, respectively) (Table 3.2).

For those pastures with a known seeding history (n=65 or 63.7% of all pastures) (Fig. 3.5), managers provided detailed seed mixtures for 40.2% of pastures, and included a description of plant species with at least one species provided to the genus level (i.e. “alfalfa”, “brome”, “clover”, etc.). Remaining pastures had seed mixes described as either a grass mixture (9.8%), legumes and grasses (5.9%), a grass mix specifying no legumes (2.0%), a pasture or forage mixture (5.9%), with the remainder unknown (27.5%). Additionally, 8.8% were reported as ‘never seeded’, and one manager indicated ‘natural recovery’ of forages from the seed bank. Among the pastures where a seeding history was provided, 54.4% (n= 35) indicated the inclusion of legumes, which increased to 66.2% (n=43) if pasture/forage mixtures were assumed to contain legumes.

The most common forage grasses seeded included bromes (30.7%), timothy (20.0%), and orchard grass (13.8%), while the most common legumes were alfalfa (27.7%) and clovers (23.1%). One land owner stated that a ‘native’ pasture had been plowed under in order to replace it with a high-performance pasture. In only 4.6% (n=3 of 65) of pastures where seeding history was provided had landowners described the specific species present and the proportions thereof seeded. It is therefore possible that many forage species are under-represented in surveys as managers were unlikely to recall all species planted, especially several decades prior, although it is also possible that seeded forage species failed to persist.

Information on an underseeded nurse-crop during forage seeding was volunteered by a small group of producers (3.0%), and we suspect a higher proportion would have reported on this if the question had been on the survey. While pasture overseeding was included on the survey, only one producer reported overseeding alfalfa, and yet another indicated overseeding 2-3 years after initial seeding. Two surveys had a notation that they had intended to overseed but had not.

3.5.1.3 Grazing Management

The majority of pastures were grazed by domestic livestock (96.1%), with only a handful abandoned (3.9%). Cattle and horses were the most common types of livestock (Fig. 3.6); mixed herds of cattle and horses also occurred (5.9%). Donkeys were also present on 2% of pastures. Use of alternative livestock, like sheep and alpacas, was rare (3.9%). Specialty farms producing elk and bison were present in the region but were not sampled.

In assigning pastures to different grazing systems (Fig. 3.8.3), rotational systems were the most common (56.9%), followed by continuous grazing (39.2%). Rotational systems were diverse, and many pasture managers indicated adaptive management (i.e. allowing pastures to 'green up' before grazing, flexible rest periods, etc.), with most systems described as simple-rotational systems. For pastures rotated over the growing season, these pastures were included in systems with an average grazing season of 4.9 ± 0.2 months (Median=5, Mode=5; n=29). For those reporting rest-rotational grazing, the duration of rest periods between grazing events was an average of 4.8 weeks (Median=4; Mode=3; n=32), while other pastures were only grazed once and allowed the remainder of the growing season to rest with a mean occupancy time of 1.9 months (Median=1.5; Mode=1; n=13). High intensity - low frequency (HILF) rotational grazing (5.5%; n=6) and temporary cross fencing (5.5%; n=6) were also utilized in rotational systems.

Continuous systems were defined by constant exposure to livestock throughout the growing season; in some cases animals resided in the sampled pasture all year (20.0%; n=8 of 40). The average grazing period of pastures continuously grazed over the growing season was 5.2 ± 0.2 months (Median=5, Mode=5; n=26), the remainder defined their grazing period as 'all summer' (18.8%; n=32). For pastures grazed continuously over the growing season, we found that the median grazing season did not differ from pastures grazed rotationally over the growing season ($X^2 = 1.263$, $P = 0.26$). After identifying that single pasture systems were common, in 2013 we recorded whether or not animals were fed hay on the pasture overwinter and found this to be 27.6% (n=16 of 58) of pastures. This corresponded with very high stocking rates ($P = 0.003$; Tables 3.1 and 3.2).

Stocking rate (SR) did not differ between rotational and continuous grazing systems, however stocking densities were dissimilar, being higher in rotated pastures (Table 3.2). Pastures used year-round and through the winter had stocking rates much higher than pastures grazed through the growing season only ($P < 0.001$; Tables 3.1 and 3.2). Pastures grazed by horses were also associated with higher stocking rates than pastures grazed by cattle, alternative livestock, and mixed herds ($P < 0.001$; Tables 3.1 and 3.2). Grazing intensity was also inferred from qualitative and quantitative measures of pasture condition through the RHA (Fig. 3.8). Stocking rates did not increase linearly as grazing intensity levels increased despite the significant relationship (Table 3.2). The highest grazing intensity (H) was associated with the highest stocking rate, however the second highest stocking rate was associated with low (L) intensity. Otherwise stocking rate increased from low-moderate to high intensities. Note that when no livestock was present the SR and SD of 0 AUM/ha and 0AU/ha lead to many significant differences (Table 3.1 and 3.2).

3.5.1.4 Herbicide Use and Weed Control

Producers had sprayed herbicide in the last 3 years on 15.7% of pastures (N=16). When asked what rate they applied, most mentioned it was at the "recommended rate" on the product label. The herbicide product used and target weed(s) were also recorded during the 2013 survey. While not requested the year prior, this information was often volunteered in the 2012 survey, and target weeds were inferred from the RHA and plant cover data. Products with a group 4 mode of action, synthetic auxins, were commonly chosen, likely for their systemic and residual properties against perennial noxious weeds, with Grazon® being a popular choice (Table 3.3). Overall, 83.3% of pastures surveyed contained noxious and prohibited weeds, with Canada thistle (*Cirsium arvense*) being the most common (77.5% of pastures; n=79) and was the primary target for herbicide use (Fig. 3.8.6). Not all producers used herbicides to target noxious weeds, with individual cases of hand-pulling and swathing thistle (note that these were often used in addition to herbicide). Herbicide was also used to target nuisance weeds like dandelion (*Taraxacum officinale*) in two cases and western snowberry (*Symphoricarpos occidentalis*) in one case. In Alberta, the *Weed Control Act* is enforced by the county, and although we did not ask if their management of weeds was enforced, managers volunteered that the county had sprayed their pasture in two cases.

3.5.1.5 Nutrient Addition and Other Management Actions

Fertilizer was applied to 8.8% of pastures (N=9) in our study area, though only a handful of producers were able to recall the application rate and is thus not reported further. Among those applying fertilizer, more pastures were treated in spring (55.6%) than fall (44.4%). Notably, manure spreading was more common than fertilization, affecting 24.5% of all 102 pastures, even though this information was not requested in the 2012 survey (i.e. this constituted voluntary information). Harrowing, which is often used for spreading manure, was reported in 33.3% of pastures. History of manure application was associated with higher stocking rates and densities ($P < 0.026$; Table 3.1 and 3.2); this trend was not significant with a harrowing. Mowing or swathing was reported in 8.8% of pastures. Aeration was infrequently used, being reported for only 3.9% of all pastures. Additional information volunteered from producers identified 2.0% of pastures described as using ‘organic’ management. Unique management actions (single observations) included: spreading of drilling mud, mulch spreading, and one pasture was treated with chicken manure.

3.5.1.6 Fire

Fire was known to have occurred based on the surveys in 14.7% of pastures, with 6.9% being wild (or accidental) and 7.8% prescribed. Note that this included 6 pastures from the Blackfoot Grazing Reserve, which were improved using prescribed fire in 1980. Given that fire was common historically and also used as a tool to remove woody cover during early settlement, we recorded the presence of charcoal in the top 15 cm of mineral soil. When combined with pastures identified as recently burned, 36.3% of all 102 pastures had at least some evidence of a history of fire.

3.5.1.7 Other Disturbances

Burrowing mammals like pocket gophers (*Thomomys talpoides*), Richardson's ground squirrels (*Urocitellus richardsonii*), Franklin's ground squirrels (*Spermophilus franklinii*), and North American badgers (*Taxidea taxus*) were perceived as pests for 58.8% of pastures. Grasshoppers were not perceived as a problematic pest in recent years, with some producers referring to a drought in the early 2000s (presumably 2002) (Bonsal and Regier 2007) as leading to problematic grasshoppers.

During 2013, participants were asked if there were industrial disturbances on their land (N=58 pastures; N=42 participants). Of these pastures, access roads and railways were reported in 12.1%, while oil and gas disturbance (indicated by the presence of pipelines) were present in 39.7%, with 29.3% containing wells, and 3.4% had (oil) pumpjacks. Gravel pits were also found though not common (5.2%). Overall, 51.7% of pastures reported no disturbances. Of the 2013 sample group, 7.1% of participants (n = 3of 42) had ongoing 'disagreements' over resource extraction, and 4.8% (n = 2 of 42) brought up the subdivision of land into acreages as a concern when asked about disturbance.

3.5.2 Rangeland Health Summary

Rangeland health assessment (RHA) scores revealed that the average pasture score in north central Alberta's Parkland was healthy (RHA score $\bar{x}$ = 79.8 ± SE 1.3; n =102), with 65.7% (Score ≥75%) of pastures healthy, 30.4% healthy with problems, and 3.9% unhealthy (Score ≤50%). Most grasslands were identified as tame (89.2%), with a smaller subset of modified-tame (11.8%) pasture.

Most pastures (64.7%) had high forage cover (i.e. relative cover >90% for tame pastures or >75% for modified-tame pastures) (Fig. 3.10). Overall, there was limited cover of non-forage plant species like native forbs, or nuisance and noxious weeds. The latter would result in a loss of scoring points for forage species shifts associated with the loss of tall productive forages and their replacement by ruderals, particularly undesirable invasive plants. Scores for hydraulic function and litter abundance represented a highly weighted component of the RHA score at 25%; thus, pastures lacking litter had the potential to have considerably lower total RHA scores. Litter was reduced from the recommended criteria (where the litter layer was visible and uniform across the pasture, ≥ 450 lbs/acre) for 55.9% of pastures sampled. Indicators of erosion (i.e. hoof sheer, wallowing, flow patterns, etc.) were present in 54.9% of pastures. No evidence of macro erosion (i.e. soil movement, material carried off site, obvious water flow patterns, exposed plant roots, etc.) was recorded. Anthropogenic bare soil exceeding 5% was recorded in 21.5% of pastures. The cover and density of noxious weeds, worth 10% of the total RHA score, were criterion where most pastures (83.3%) fell below the maximum score. While noxious weed cover never exceeded 15% in our sampled pastures (which would have led to the lowest possible score for noxious cover), noxious weed density was scored as a 'heavy infestation' in 32.4% of pastures. Due to past cultivation and ongoing grazing, many pastures (71.6%) lacked enough woody cover (relative cover <5%) or density to affect their RHA score.

3.5.3 Management Regimes

Categorical data describing current management (cultivation and fire history excluded) were analysed with an MCA (multiple correspondence analyses) showing the relationship between management activities for the surveyed pastures (Fig. 3.11). Pastures that had not been grazed by livestock in recent years had similar management regimes and were positively associated with the first axis (MCA1), explaining 14.4% of variance among sample sites. Along the second axis (MCA2), which explained 9.1% of variance, there was substantial divergence between pastures continuously grazed and rotationally grazed, in the timing of use, and also the type of herbivores grazed. The third axis (MCA3), which explained 7.9% of variance, exhibited stronger divergence of management inputs like harrowing, feeding hay, manuring, mowing or swathing than the two previous axes (Table 3.6 and Table 3.7).

Pastures that were swathed or mowed, grazed in winter, or aerated, tended to cluster together. County and provincial grazing reserve lands were usually rotationally grazed, and clustered next to rotational grazing. Distinct management regimes within rotational systems did not emerge in the MCA. However, a cluster dendrogram of the MCA axes identified management patterns within the rotationally grazed pastures (Fig. 3.12); rotational grazing was associated with larger farms (>160 ac) with cattle. Vectors for the rangeland health assessment score (RHA) and stocking rate (SR) were divergent. Higher range health scores were associated with grazing reserve land and rotational grazing with cattle, while higher stocking rates and low range health were associated with pastures comprised of small holdings or grazed all year, particularly by horses or mixed herds of animals.

3.6 Discussion

Pastures in North Central Alberta are generally in good health and dominated by competitive forage plants, although some issues were noted associated with increased bare soil and problematic plants, including noxious weeds, which tended to limit the RHA scores. Although documented management actions of producers were highly variable, several distinct management regimes were evident. For example, when livestock was absent year-round other management actions (i.e. mowing, herbicide, fertilization, etc.) were not taken, with the exception of the Wagner natural area which was swathed and bailed in the fall. Overall areas without livestock could be considered abandoned or extensively managed. At the other extreme, small holdings, which we defined as less than 80 acres, were associated with continuous year-round use by horses, and supplemental feed was often provided. These areas were more likely to have lower range health and higher stocking rates. This grouping describes small hobby farmers and acreage owners housing companion animals. In contrast, larger farms (>160 acres) were associated with rotational cattle grazing during the growing season. Managers with multiple pastures and large grazing areas did not feed their animals in the pastures we sampled during the survey and likely transferred them elsewhere overwinter. Harrowing and manure spreading were paired, presumably because harrowing is used to spread manure, and this was associated with pastures where animals were fed hay over winter. Pastures containing alternative livestock for the region, sheep and alpaca, were also associated with manure spreading and harrowing. Similarly, when pastures were grazed or utilized only over the winter and dormant season, there was a tendency for these pastures to be aerated and swathed during the growing season, suggesting landowners may be relying on a small land base to sustain livestock throughout the year. Continuous grazing systems were associated with mixed herds of livestock, horses and cattle, and likely represent hobby farmers housing their companion animals with their beef cattle. Continuous grazing and mixed herds were also associated with allotments that were typically a quarter section in size, within this grouping burrowing mammals and herbicide use were reported.

The majority of pastures (64.7%) received the highest scores for relative forage cover, exceeding 90% cover for tame pastures and 75% cover for modified-tame pastures. Previous cultivation of North Central Alberta pastures and seeding of forages was common. There are sociological, political, and ecological factors responsible for significant losses of native and semi-native grasslands in heavily settled regions. During homesteading, settlers were mandated under the *Dominion Lands Act* (Bailey et al. 2010) to convert land into annual cereal crop production in order to retain deeded ownership of these lands that they otherwise received for free. Cultivation pressure in the region persists; the Plowprint Report by the World Wildlife Fund [WWF] (2016) reported grassland loss due to cultivation as 6.95% between 2011-2012 (year before the study), 3.08% between 2012-2013, and 3.63% between 2013-2014 (final year) in the "Prairie Habitat Joint Venture" region that encompasses the Canadian prairies. Newly converted acres were most commonly planted to alfalfa (19.9%), followed by wheat (19.0%), and canola (12.6%) (WWF, 2016). Native grassland in much of the parkland has also been lost due to encroachment of woody species like aspen (*Populus tremuloides*) following fire suppression (Sheffler 1976; Bailey and Wroe 1974). As a result, the majority of grasslands, results affirmed here, appear to be dominated by tame pastures. Moreover, invasion by cool-season grasses like smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*) can be exacerbated by overgrazing of grazing-sensitive grasses like plains rough fescue (*Festuca hallii*) and western porcupine grass (*Hesperostipa curtiseta*) given the favorable moisture regime

(Archibold et al. 2003) and has likely led to the widespread loss of native grasslands. Observed reductions in health scores were generally tied to modest reductions in forage cover, as well as increases in bare soil and undesirable plants species, including noxious weeds. The latter two observations could also reflect excessive disturbance regimes such as early spring grazing, excessive stocking rates, and an insufficient recovery time between grazing events, as well as the application of manure (and its weed seeds), which would also result in a decline in the vigor (i.e. cover and biomass) of key forage plants and a shift to less palatable, disturbance tolerant plant species.

Remaining pastures in this region likely exist due to an ongoing need for sufficient pasture and forage resources to support livestock on mixed farms (which often include cattle), or companion animals in small hobby farms (e.g. horses) in the highly settled, mostly privately-owned landscape, surrounding the city of Edmonton and associated suburbs. As a result, agricultural landscapes in this region are likely to contain at least some areas of pasture, often on poorer quality soils that are less suited for annual cropping (Samson and Knopf 1994). Alternatively, producers may rotate annual cropland with perennial forages to maintain forage supply, thereby accounting for the abundance of seeded pasture. In a survey by Entz et al. (1995) Canadian prairie farmers reported benefits from including perennial forage cover in their annual crop rotation such as weed control and increased yield. The average age of forage stands reported were 3 to 5 years in mesic regions like south central Manitoba, and 6 to 9 years in southern Saskatchewan; with reductions in forage yield and damage from burrowing pocket gophers being primary motives to cultivate the field (Entz et al. 1995). Periodic ploughing and reseeding can also be employed to rejuvenate pastures (Levassor et al. 1990) if improvements are not realized under nutrient addition, particularly tame pastures and hayfields that are well known to stagnate over time (Lardner et al. 2000), which could therefore help explain the relatively young age of many pastures surveyed. For remaining native and semi-native grasslands there is a risk of loss due to overgrazing, cultivation, and invasive species. In 2003 (Gauthier and Wiken 2003), Alberta was the Canadian prairie province that had retained the most grassland at 43.1%, however in Alberta’s Aspen Parkland only 12.0% of the natural subregion was grassland and we suspect further decline has occurred. Note that this is comparable to the proportional representation of modified-tame grassland pastures (11.8%) we found, and some had a confirmed history of previous cultivation.

Grazing management can vary in complexity, with certain systems requiring high inputs (i.e. infrastructure, labor, monetary, etc.). To reduce complexity in this study, grazing management was classified as continuous or rotational, regardless of the intensity of management. Within rotational systems there was evidence of some systems requiring intensive management (i.e. moving temporary fence every few days, or regularly rotating animals through smaller pastures for HILF systems). In rangeland management there is considerable debate over whether, when and how continuous and rotational grazing systems differ in their ability to sustain plant production and range condition (Briske et al. 2008; Teague et al. 2013). Continuous grazing has been associated with overgrazing of preferred areas while rotational grazing has been associated with controlling over-utilization and altering the timing of use to prevent the loss of desirable forages, biodiversity, and degradation in range condition (Bailey et al. 2010), including riparian health (Popp et al. 2004). Continuous grazing and rotational grazing should be compared with stocking rate in mind (Teague et al. 2013); when we compared the stocking rate of continuous and rotational systems we found they were similar and high. In a survey of Canadian prairie cow/calf producers (Chorney and Josephson 2000), producers reported benefits after switching to rotational grazing (primarily from continuous) such as greater livestock gains, improved forage quality and quantity, reduced overwintering costs, with 88% experiencing greater net farm income. However, to achieve greater economic returns 83% of those respondents reported greater labour costs and 83% faced higher time planning their grazing management (Chorney and Josephson 2000).

Central Parkland grasslands are highly productive, resulting from a favorable moisture regime and black chernozemic soil, thus they are prone to overutilization. Ecologically sustainable stocking rates (ESSR) recommended for the region are typically: 0.74 to 1.75 AUM/ha for Kentucky bluegrass-Smooth brome (depending on successional pathway), 0.86 AUM/ha for Smooth brome-Kentucky bluegrass-

Dandelion and Timothy-Smooth brome, and 2.50 AUM/ha for Alfalfa/Brome-Kentucky bluegrass (Government of Alberta 2013). Note that pastures under both continuous and rotational grazing typically exceeded the ESSR by more than two to three-fold. Pastures with the most mindful stocking rates were never cultivated; these producers were likely managing the forage resource responsibly reducing the need to rejuvenate or improve the pasture. Managers of uncultivated land were often proud of their management and recognized native species housed there were sensitive and valuable. We conclude that regionally producers are likely grazing pastures near their carrying capacity, and when carrying capacity is exceeded they need to feed their animals. Thus, any benefits attributed to rotational grazing could be lost due to overutilization and could result in indistinguishable plant communities.

In our survey few pastures surveyed were grazed over the dormant season, which contrasts with data suggesting this is a common practice in Alberta, with 62% of farms reporting that they use 'extended grazing' management (Statistics Canada*,* 2014). In the present study, there was likely a bias to sample pastures grazed over the growing season during the search, or landowners could have deliberately diverted us away from these pastures fearing that they would not be assessed favorably during a field assessment due to heavy overwinter use levels. A similar bias may be present in the greater representation of larger farms (>160 ac) in the pastures surveyed; although small farms and acreages were very common in the study region, many of these were ignored because of their small area and were therefore not representative of larger operations. Rowan (1994), in a similar survey from east Texas, defined a median ranch size of 271 ac (~108 ha) as a small-acreage ranch; in North Central Alberta, privately owning 271 ac would not be considered a small operation.

In the Canadian Agriculture Census (Statistics Canada 2014) detailed information was reported on the area of land (and in some cases the application rate) treated with manure, fertilizers, fungicides, and herbicides, but cropland and rangeland were not differentiated. The current study surprisingly revealed that manure application was a more prevalent soil amendment than fertilizer, despite the fact that macronutrient, particularly soil nitrogen, is known to strongly constrain plant growth in grasslands

(Lardner et al. 2000; Malhi et al. 2000). It is possible that some people may have misreported the in-situ dispersal of manure for stockpiled manure. However, during the 2013 survey landowners typically allowed us to sample their stockpiled manure without misunderstanding, which supports the fact that about a quarter of pastures are treated with manure. We found that pastures treated with manure were grazed at very high grazing capacities and were likely intensively managed to improve productivity.

It is typically rare for farms to be certified organic (or in transition) with only 0.8 % of Alberta farms reported, or 2.0% nation-wide (Statistics Canada 2011); with our finding of 2.0% of pastures, or 2.7% of land managers, it resembled the national average. It should be noted that our interviewees stated the use of 'organic management', prompted by the questioning of herbicide use and we did not inquire if they were certified.

Noxious weeds were prevalent in a relatively high proportion of pastures (83.3%), with Canada thistle (*C. arvense*) the most ubiquitous, yet herbicide use was only reported in 15.7% of survey sites. Moore (1975) described *C. arvense* as a naturalized weed of the Canadian Prairie Provinces, finding it in 40.7% of surveyed areas in Alberta and Saskatchewan at the time. Our rates of presence of this weed were much greater, at 77.5% across these north central Alberta pastures. None of the pastures surveyed had absolute noxious weed cover exceeding 15%, which would have resulted in the lowest possible scoring and would have been indicative of abuse or neglect of pastures. Noxious weed cover less than 1% was the norm representing 66.7% of pastures. It is possible that we under reported control efforts of producers. In addition to asking if they had sprayed herbicide we should have questioned how they managed undesirable plants and whether they use alternative methods (i.e. targeted grazing, mowing, hand pulling, etc.), as this information was only volunteered rather than requested. In two cases, we were informed that the motive for controlling weeds in their pasture was enforced by their county. Receiving weed notices from counties can hurt a landowner's sense of pride, thus counties like Parkland County, are educating private landowners by leaving informative 'door hangers' (a brochure) when noxious weeds are present before providing a notice, thereby providing landowners an opportunity to pre-emptively remedy the problem. Strathcona County uses noxious weed cover thresholds based on allotment size before issuing notices, and targets problematic areas such as hamlets and acreages where noxious and prohibited noxious weeds are more prevalent. Only a handful of prohibited noxious weeds were present, with only single occurrences of species like field scabious, knapweed, and orange hawkweed. The efficacy of the *Weed Control Act* in educating and motivating people to control problematic weeds has not been assessed formally in Alberta on a provincial or county level despite the important role it plays in reducing the impact of deleterious species. A case study in Australia found that landowners were more likely to control a legislated weed inherently due to its declaration, but compliance from neighbors and the abundance of the weed on their own property contributed to their willingness to control it (Reeve et al. 2015). Consultation with counties identified the lack of consistency in management and enforcement between counties as an issue, however it does allow counties to adapt management to their unique environments, funding, and allows them to make municipal amendments to the list of species.

Among other disturbances in the pastures sampled, not surprisingly most producers reported a prevalence of pests, particularly ground squirrels and moles, and which are often targeted for pest control through poisoning, trapping, and other means. Ground squirrels and pocket gophers are known to lead to a loss in pasture yield and damage the soil surface (Carlson and Crist 1999; Entz et al. 1995). Carlson and Crist (1999) found that pocket gophers mounds could occupy 1% to 6% of pasture area, and mounds were more abundant in lightly grazed pastures. Forage is not only lost due to the cover of mounds and burrows but overlap in the foraging preference of burrowing mammals (primarily studied with prairie dogs) and cattle can lead to reductions in palatable herbage available to livestock; however large numbers of individuals are required to meet the equivalency of one AU, and trade-offs like higher forage quality where overlap occurs can make-up lost productivity (O'Meilia et al 1982; Wuerthner 1997). Proulx (2010) described that ground squirrel populations on the Canadian prairies had reached high densities (>40 adults/ha), and attributed the 'outbreak' to number of socio-economic and environmental factors including: drought (primarily referring to 2000 and 2001), poor pasture management (i.e. overgrazing), lower cattle prices due to BSE (bovine spongiform encephalopathy) in the early 2000s exacerbated overgrazing, banning of strychnine in the early 1990s, loss of predators (including loss of non-target species from strychnine), and loss of smaller family-sized operations. In addition, the perceived fear of livestock injury from burrows caused by larger animals like badgers (Minta and Marsh 1988) can strongly motivate control. In Alberta this philosophy, combined with disease, and habitat loss lead to the extirpation of the prairie dog (Wuerthner 1997). During the survey we did not inquire if they were actively trying to control pasture pests or how.

Anthropogenic infrastructure was commonplace, largely that associated with transportation corridors and energy extraction. The latter features reflect the high-density settlement nature of this region, and also the abundance of oil and gas (Allred et al. 2015), both of which contribute to landscape fragmentation, and potentially the loss of native grassland and therefore the ongoing conversion of land into tame pasture. In central Alberta's Parkland, plains rough fescue does not successfully re-establish following pipeline construction (Desserud and Naeth 2013). Rough fescue recovery is possible when the fescue sod remains intact using minimal disturbance methods; however, historically methods that remove the soil and vegetation were common (Desserud and Naeth 2013). These features create disturbed edges that increase bare soil (Elsinger 2009) and facilitate exotic species (Allred et al. 2015; Desserud and Naeth 2013), which collectively will contribute to the lower than optimal range health scores in more than a third of the study sites. Findings of the RHA in general indicated that most pastures in the study region were relatively healthy as tame pastures. Hansen and Clavenger (2005) found that transportation corridors effectively spread exotic species, particularly in grasslands and this is exacerbated under disturbance. Alberta's counties tend to target these areas and attempt to control high priority species. It should be noted that the majority of road allowances are developed in the Central Parkland, resulting in high connectivity of disturbances and habitat fragmentation.

Our results also revealed a significant presence of fire across the area. Historically, fire was a common disturbance among grasslands in the northern Great Plains prior to European settlement

(Archibold et al. 2003), with a fire return interval of every 10-15 years (Wright and Bailey 1982). In the Parkland, fire was also used as a tool to convert forest and grassland to bare soil suitable for cultivation. Public land managers like the Blackfoot Grazing Reserve were also noted in the surveys to use prescribed fire to control woody plants and improve grazing capacity. Even when fire was reported during the management surveys, assessed pastures for these areas often lacked visual indicators of fire. Pastures that retained charred woody debris in the top 15 cm of mineral soil were likely subjected to fire some time ago and may reflect large-scale fire events such as that in 1895, when fire ravaged much the area east of Edmonton and throughout the Beaver Hills (Kjorlien 1977). Given the known historical importance of fire, it is perhaps surprising that less than half of the pastures examined exhibited evidence of fire.

We have reported on the management actions within pastures of a sizeable portion of Alberta's Parkland and Dry Boreal Mixedwood. It is important to note that there are sociological and economic factors present, but not measured, that likely effect the management decisions of pasture managers. Close proximity to the city of Edmonton influences the price of land, which could influence the size of farms, and therefore their management decisions (i.e. small farms, particularly hobby farms and acreages, are more likely to employ continuous grazing due to a lack of grazing area). Conversely, resources are available to small farms and acreage owners to educate them on the benefits of using simple rotational systems (i.e. switchbacks) and how to monitor forage resources (Alberta Government, Cows and Fish, Ducks Unlimited, Forage Associations, etc.). A survey by Kachergis et al. (2013) found that Wyoming ranchers gained 97% of their knowledge on grazing management from other ranchers despite a preference to acquire information through published sources. This cultural practice could be present in Alberta as well, and it is important to note that there is movement of people back-to-the-land, and they may well lack the fundamental knowledge to sustainably manage pasture resources (Halfacree 2007). In peri-urban areas, the loss and subdivision of arable land and heritage farms to urban sprawl and development effects the persistence of agriculture including cattle operations. Instead, people raise livestock in these areas for the enjoyment of the work or animal husbandry, and they are more likely to have off-farm income to supplement their lifestyle, especially on smaller farms (Cialdella et al. 2009; Rowan 1994; Sayre 2004). Notably, concerns of subdivision and annexation were raised during our surveys, and a handful of people who had recently purchased land volunteered that they were new to the rural lifestyle.

In our survey 31.5% of participants were female, which resembles the Statistics Canada (2011) finding that 29.0% of farm operators in Alberta are female. The proportion of women operating farms in Alberta is higher than the national average at 27.4%. We did not collect information on the manager's age, or their incomes. At the time of sampling we considered individual pastures as the experimental unit, and therefore our survey was not designed to obtain personal information from responders. Statistics Canada (2011) reported that 52.0% of Alberta farm operators had off-farm businesses or income, and 37.8% worked more than 40 hours a week on their land, and 32.8% of younger operators (<35 years) worked off-farm more than 40 hours a week (the highest out of all other age demographics). Being situated so close to the city of Edmonton, it is possible that our study's pastures were influenced by landowners having multiple occupations, which in turn, could have altered the attention to management details.

Finally, future studies linking producer management to effects on rangeland communities and ecosystem function or services should inquire about management goals, motivations and opinions, so better links can be drawn between producer actions and socio-economic variables, education, and attitude (Kachergis et al. 2013; Sayre 2004). Rephrasing questions to ask them about a process (i.e. how do you manage undesirable plants?), instead of directly asking if they have specifically done a certain action (i.e. have you sprayed herbicide?) may yield more information about management. This could be accompanied by supplemental questions. Audio recordings of interviews may reveal large amounts of metadata, and record incidental details that were missed during initial interviews. In this study, a lack of specific information collected regarding total farm size and management limited our analysis in some ways. It was difficult to infer post-interview the total area farmed by producers, whether or not their operations were mixed, or if farming was their sole income. Larger operations did not report to us their total herd size because questions were aimed at pasture management, thus total herd size was not reported for operators. Many assumptions were made regarding stocking rate and density calculations because our initial questioning did not identify livestock breeds (used constant AUEs) or exact pasture dimensions (aerial photos in most cases. Thus, data regarding the stocking rate and density was unknown for about 20% of pastures sampled.

3.7 Conclusions and Management Implications

Managers practice a wide range of activities on these largely private grasslands, which in turn, are reflected in range health scores. Producer behavior can be distinguished based on factors such as management intensity, including whether they practice rotational grazing, use inputs like fertilizer or herbicides, or practice year-long grazing and on-pasture winter feeding. Range health scores also reflect these activities, as while most pastures are considered healthy, nearly one-third of pastures were healthy with problems, mostly reflecting increases in bare soil, a reduced cover of productive forages, and increases in weeds. Pastures with lower range health were associated with higher stocking rates and tended to support horses or mixed herds of livestock in pastures that were on small land holdings and/or grazed year-round. This information provides clarity on the management activities taking place in northern temperate grasslands, their ultimate impacts on range health, and provides insight into the actions necessary to sustain these pastures.

3.8 Literature Cited

Adams, B. W., Ehlert, G., Stone, C., Lawrence, D., Alexander, M., Willoughby, M., Hincz, C., Moisey, D., Burkinshaw, A., Carlson, J. and France, K. 2005. Rangeland health assessment for grassland, forest and tame pasture. Public Lands and Forests Division, Alberta Sustainable Resource Development.

Agriculture Canada. 2014. Human Activity and the Environment: Agriculture in Canada. Statistics Canada Catalogue no.16-201-X.

Alberta Agriculture and Forestry. 2016. AgroClimatic Information Service. Accessed May 10, 2016 for -University of Alberta's South Campus weather station. http://agriculture.alberta.ca/acis/alberta-weather-data-viewer.jsp

Allred, B.W., Smith, W.K., Twidwell, D., Haggerty, J.H., Running, S.W., Naugle, D.E., and Fuhlendorf, S.D. 2015. Ecosystem services lost to oil and gas in North America: Net primary production reduced in crop and rangeland. Science **348**(6233):401-402.

Archibold, O.W., Ripley, E.A. and Delanoy, L. 2003. Effects of season burning on the microenvironment of fescue prairie in central Saskatchewan. Canadian Field-Naturalist **117**(2):257-266.

Bailey, A.W., McCartney, D. and Schellenberg, M.P. 2010. Management of prairie rangeland. Agriculture and Agri-Food Canada.

Bailey, A.W. and Wroe, W.R. 1974. Aspen invasion in a portion of the Alberta Parklands. Journal of Range Management **27**(4):263-266.

Baron, V.S., Dick, A.C., Mapfumo, E., Malhi, S.S., Naeth, M.A., and Chanasyk, D.S. 2001. Grazing impacts on soil nitrogen and phosphorus under parkland pastures. Journal of Range Management **54**(6):704-710.

Baron, V.S., Mapfumo, E., Dick, A.C., Naeth, M.A., Okine, E.K., and Chanasyk, D.S. 2002. Grazing intensity impacts on pasture carbon and nitrogen flow. Journal of Range Management **55**(6):535-541.

Bonsal, B. and Regier, M. 2007. Historical comparison of the 2001/2002 drought in the Canadian Prairies. Climate Research **33**(3):229-242.

Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, K.M., Gillen, R.L., Ash., A.J. and Willms, W.D. 2008. Rotational grazing on rangelands: Reconciliation of perception and experimental evidence. Rangeland Ecology and Management **61**:3-17.

Carlson, J.M., and Crist, T.O. 1999. Plant responses to pocket-gopher disturbances and topography. Journal of Range Management **52**(6):637-645.

Chorney, B. and Josephson, R. 2000. A survey of pasture management on the Canadian prairies with emphasis on rotational grazing and managed riparian areas. M. Sc. Thesis, University of Manitoba, Department of Agricultural Economics and Farm Management, Winnipeg, Manitoba.

Cialdella, N., Dobremez, L., and S. Madelrieux. 2009. Livestock farming systems in urban mountain regions: Differentiated paths to remain in time. Outlook on Agriculture **38**(2):127-135.

De Keyser, E.S., Dennhardt, L.A., and Hendrickson, J. 2015. Kentucky bluegrass (*Poa pratensis*) invasion in the Northern Great Plains: a story of rapid dominance in an endangered ecosystem. Weed Science **64**(3):409-420.

De Mendiburu, F. 2017. agricolae: Statistical Procedures for Agricultural Research. R package version 1.2-6. https://CRAN.R-project.org/package=agricolae

Desserud, P.A, and Naeth, M.A. 2013. Natural recovery of rough fescue (*Festuca hallii* (Vasey) Piper) grassland after disturbance by pipeline construction in central Alberta, Canada. Natural Areas Journal **33**(1):91-98.

Donkor, N.T., Gedir, J.V., Hudson, R.J., Bork, E.W., Chanasyk, D.S., and Naeth, M.A. 2002. Impacts of grazing systems on soil compaction and pasture production in Alberta. Canadian Journal of Plant Science **82**(1):1-8.

Dormaar, J.F., Adams, B.W., and Willms, W.D. 1997. Impacts of rotational grazing on mixed prairie soils and vegetation. Journal of Range Management **50**(6):647-651.

Dray, S., and Dufour, A.B. 2007. The ade4 package: implementing the duality diagram for ecologists. Journal of Statistical Software **22**(4):1-20.

Elsinger, M.E. 2009. Reclamation status of plains rough fescue grasslands at Rumsey Block after well site and pipeline disturbance. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Entz, M.H., Bullied, W.J., and Katepa-Mupondwa, F. 1995. Rotational benefits of forage crops in Canadian prairie cropping systems. Journal of Production Agriculture **8**(4):521-529.

Fulkerson, W.J. and Michell, P.J. 1987. The effect of height and frequency of mowing on the yield and composition of perennial ryegrass-white clover swards in the autumn to spring period. Grass and Forage Science **42**(2):169-174.

Gauthier, D.A., and Wiken, E.B. 2003. Monitoring the conservation of grassland habitats, prairie ecozone, Canada. Environmental Monitoring and Assessment **88**:343-364.

Government of Alberta. 2013. Range plant communities and range health assessment guidelines for the Central Parkland subregion of Alberta. Alberta Government, Red Deer, AB.

Greenacre, M. and Blasius, J. 2006. Multiple correspondence analysis and related methods. Chapman and Hall/CRC, Boca Raton, Florida, USA.

Grekul, C. W. and Bork, E.W. 2007. Fertilization augments Canada thistle (*Cirsium arvense* L. Scop.) control in temperate pastures with herbicides. Crop Protection **26**(4):668-676.

Halfacree, K. 2007. Back-to-the-land in the twenty-first century: Making connections with rurality. Tijdschrift Voor Economische en Sociale Geografie **98**(1):3-8.

Hansen, M.J., and Clevenger, A.P. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation **125**(2):249-259.

Johnston, A., Smoliak, S. and Stringer, P.W. 1969. Viable seed populations in Alberta prairie topsoils. Canadian Journal of Plant Science **49**(1):75-82.

Kachergis, E., Derner, J., Roche, L., Tate, K., Lubell, M., Mealor, R. and Magagna, J. 2013. Characterizing Wyoming ranching operations: Natural resource goals, management practices and information sources. Natural Resources **4**:45-54.

Kjorlien, M.E. 1977. A review of historical information of fire history and vegetation description of Elk Island and the Beaver Hills. Parks Canada, Elk Island National Park, Fort Saskatchewan, Alberta.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2000. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on dry matter yield and forage quality. Canadian Journal of Plant Science **80**: 781-791.

Le, S., Josse, J., and Husson, F. 2008. FactoMineR: An R package for multivariate analysis. Journal of Statistical Software **25**(1):1-18. 10.18637/jss.v025.i01

Lenth, R.V. 2016. Least-Squares Means: The R Package lsmeans. Journal of Statistical Software **69**(1): 1-33. doi:10.18637/jss.v069.i01

Levassor, C., Ortega, M. and Peco, B. 1990. Seed bank dynamics of Mediterranean pastures subjected to mechanical disturbance. Journal of Vegetation Science **1**(3):339-344.

Malhi, S.S., Heier, K., Nielsen, K., Davies, W.E., and Gill, K.S. 2000. Efficacy of pasture rejuvenation through mechanical aeration and N fertilization **80**:813-815.

Minta, S.C., and Marsh, R.E. 1988. Badgers (*Taxidea taxus*) as occasional pests in agriculture. Vertebrate Pest Conference Proceedings Collection VPC13:42.

Moore, R.J. 1975. The biology of Canadian weeds. 13. *Cirsium arvense* (L.) Scop. Candian Journal of Plant Science **55**:1033-1048.

Naeth, M.A. Bailey, A.W., Pluth, D.J., Chanasyk, D.S., and Hardin, R.T. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland. Journal of Range Management **44**(1):7-12.

O'Meilia, M.E., Knopf, F.L, and J.C. Lewis. 1982. Consequences of competition between prairie dogs and beef cattle. Journal of Range Management **35**(5):580-585.

Popp, M., Chorney, B. and Keisling, T. 2004. A Case study on rotational grazing and riparian zone management: Implications for producers and a conservation agency. Journal of Natural Resources and Life Sciences Education **33**:28-34.

Proulx,G. 2010. Factors contributing to the outbreak of Richardson's ground squirrel populations in the Canadian prairies. Proceedings of the 24th Vertebrate Pest Conference, Sacramento, California (pp. 213-217).

Province of Alberta. 2010. Weed Control Act. Her Majesty the Queen in the Right of Alberta, Edmonton.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Reeve, I.J., Coleman, M.J., and Sindel, B.M. 2015. Factors influencing rural landholder support for a mandated weed control policy. Land Use Policy **46**:314-323.

Rowan, R.C. 1994. Are small-acreage livestock producers real ranchers? Rangelands **16**(4):161-166.

Samson, F. and Knopf, F. 1994. Prairie Conservation in North America. BioScience **44**(6):418-421.

Sayre, N.E. 2004. Viewpoint: The need for qualitative research to understand ranch management. Journal of Range Mangement **57**:668-674.

Scheffler, E J. 1976. Aspen forest vegetation in a portion of the east-central Alberta parklands. M.Sc. thesis, University of Alberta, Edmonton, Alberta.

Sinkins, P.A., and Otfinowski, R. 2012. Invasion or retreat? The fate of exotic invaders on the northern prairies, 40 years after cattle grazing. Plant Ecology **213**(8):1251-1262.

Statistics Canada. 2011. Farm and farm operator data: 2011 census of agriculture. Statistics Canada Catalogue no. 95-640-X.

Tallowin, J.R.B., Rook, A.J., and Rutter, S.M. 2005. Impact of grazing management on biodiversity of grasslands. Animal Science **81**(2):193-198.

Tannas, S. 2011. Mechanisms regulating *Poa pratensis* L. and *Festuca campestris* Rybd. Within the foothills fescue grasslands of southern Alberta. Ph.D. Thesis, University of Alberta, Department of Agriculture, Food and Nutritional Science. Edmonton, Alberta.

Teague, R. Provenza, F., Kreuter, U., Steffens, T. and Barnes, M. 2013. Multi-paddock grazing on rangelands: Why the perceptual dichotomy between research results and rancher experience? Journals of Environmental Management **128**:699-717.

Vujnovic, K., Wein, R., and Dale, M.R.T. 2000. Factors determining the centrifugal organization of remnant *Festuca* grassland communities in Alberta. Journal of Vegetation Science **11**:127-134.

Willms, W.D. and Jefferson, P.G.1993. Production characteristics of the mixed prairie: Constraints and potential. Canadian Journal of Animal Science **73**(4):765-778.

Willms, W. D. and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Willms, W.D., Smoliak, S. and Dormaar, J.F. 1985. Effects of stocking rate on a rough fescue grassland vegetation. Journal of Range Management **38**(3):220-225.

Wright, H.A. and Bailey, A.W. 1982. Fire ecology: United States and southern Canada. John Wiley & Sons.

[WWF] World Wildlife Fund. 2016. Plowprint report: Facts & Figures. Accessed February 22, 2017. https://c402277.ssl.cf1.rackcdn.com/publications/947/files/original/plowprint_AnnualReport_2016_Final_REV09192016.pdf

Wuerthner, G. 1997. Viewpoint: The black-tailed prairie dog: headed for extinction? Journal of Range Management **50**(5):459-466.

Young, J. E., Sánchez-Azofeifa, G. A., Hannon, S. J. and Chapman, R. 2006. Trends in land cover change and isolation of protected areas at the interface of the southern boreal mixedwood and aspen parkland in Alberta, Canada. Forest Ecology and Management **230**(1):151-161.

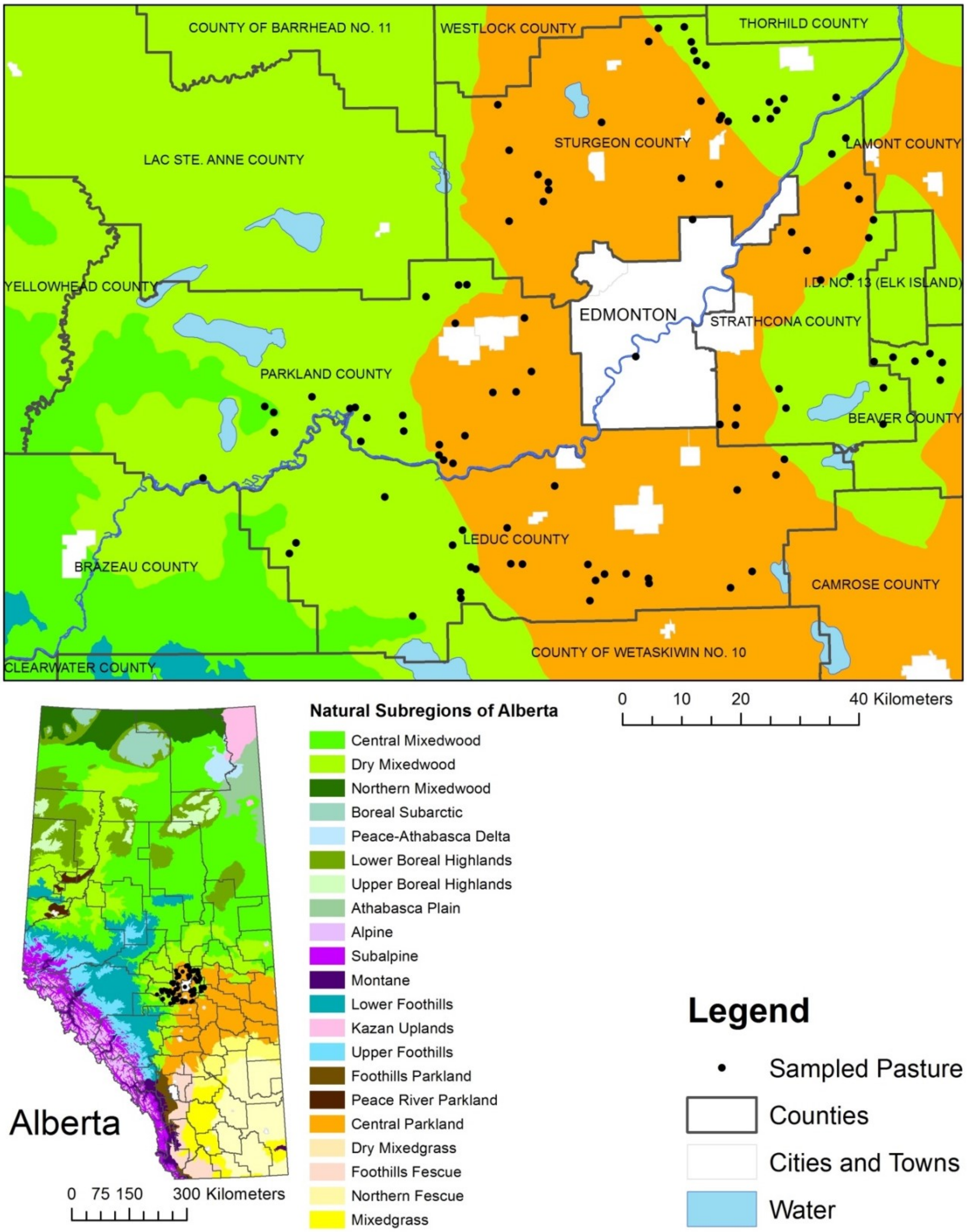


Figure 3.1. Lower map indicates the province of Alberta's natural subregions, and sampling locations. Upper map identifies the Edmonton, AB, metropolitan area with cities, towns, and counties outlined. Sampled pastures (black circles) are located within the Parkland, Sturgeon, Strathcona, and Leduc County; with 6 pastures located in the Blackfoot grazing reserve south of Elk Island National Park.

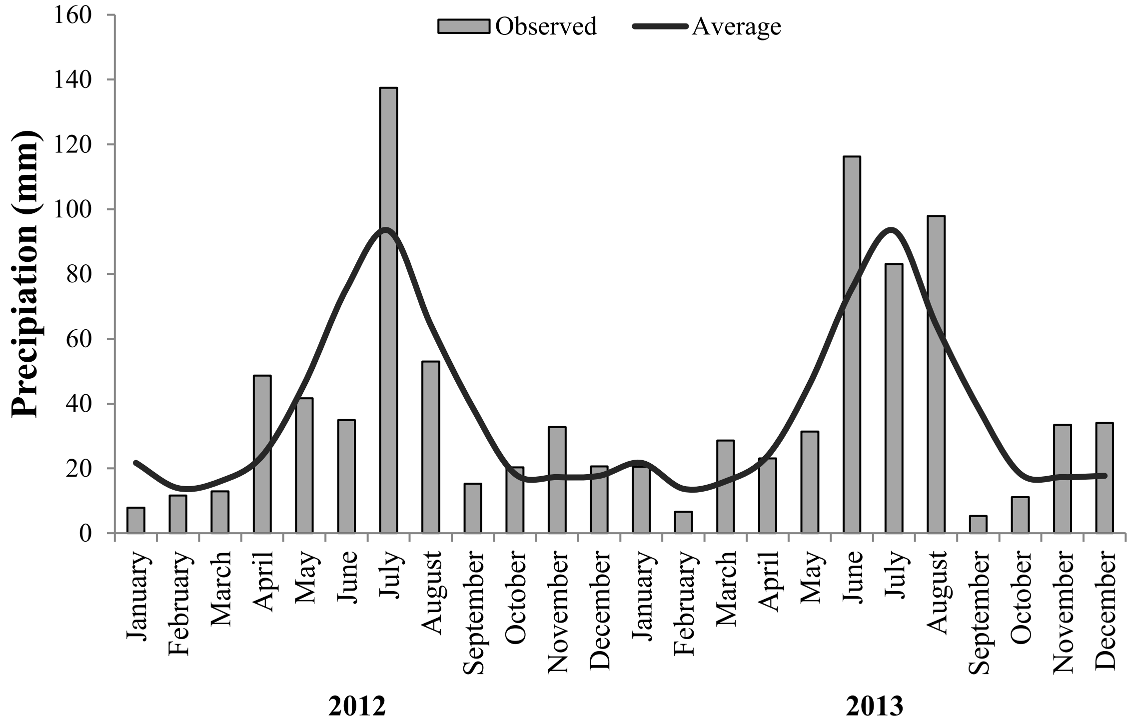


Figure 3.2. Monthly precipitation (mm) at the University of Alberta's South Campus, Edmonton, Alberta in 2012 and 2013 (Alberta Agriculture and Forestry, 2016).

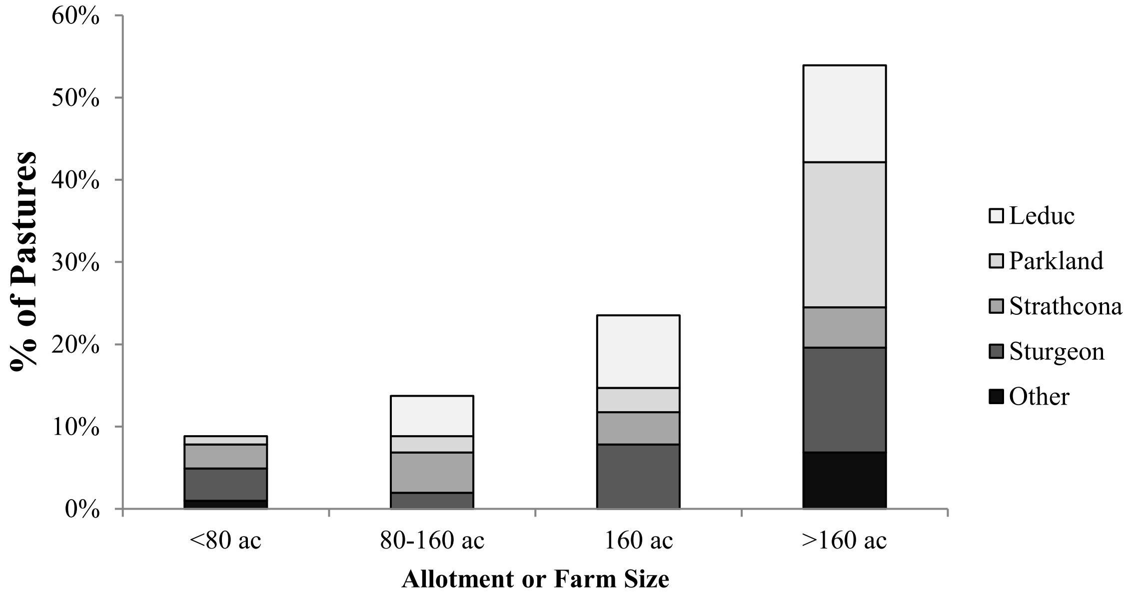


Figure 3.3. Distribution of the total size of the allotment from which each pasture sampled, further stratified by county. Pastures within the boundary of the city of Edmonton or the Blackfoot Grazing Reserve were included in the category 'other'. Pastures associated with areas <80 acres in size include acreages and smaller hobby farms.

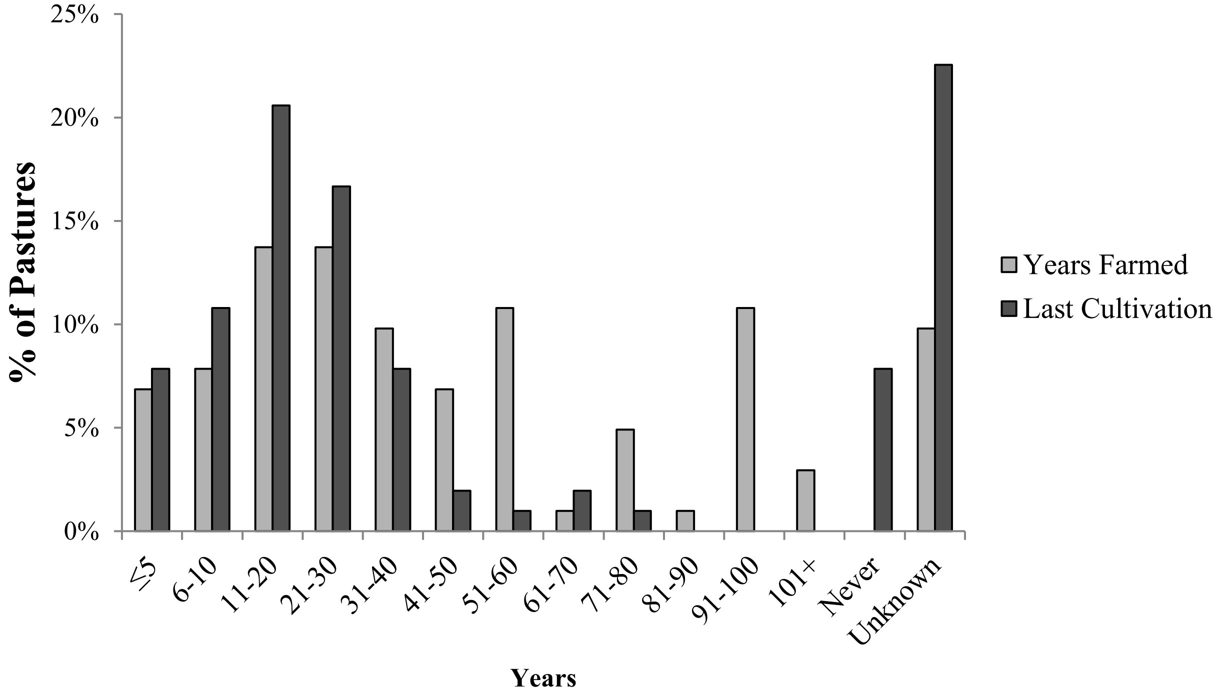


Figure 3.4. Summary of the number of years pastures in North Central Alberta had been farmed by the current family or land manager, and the number of years since last known cultivation.

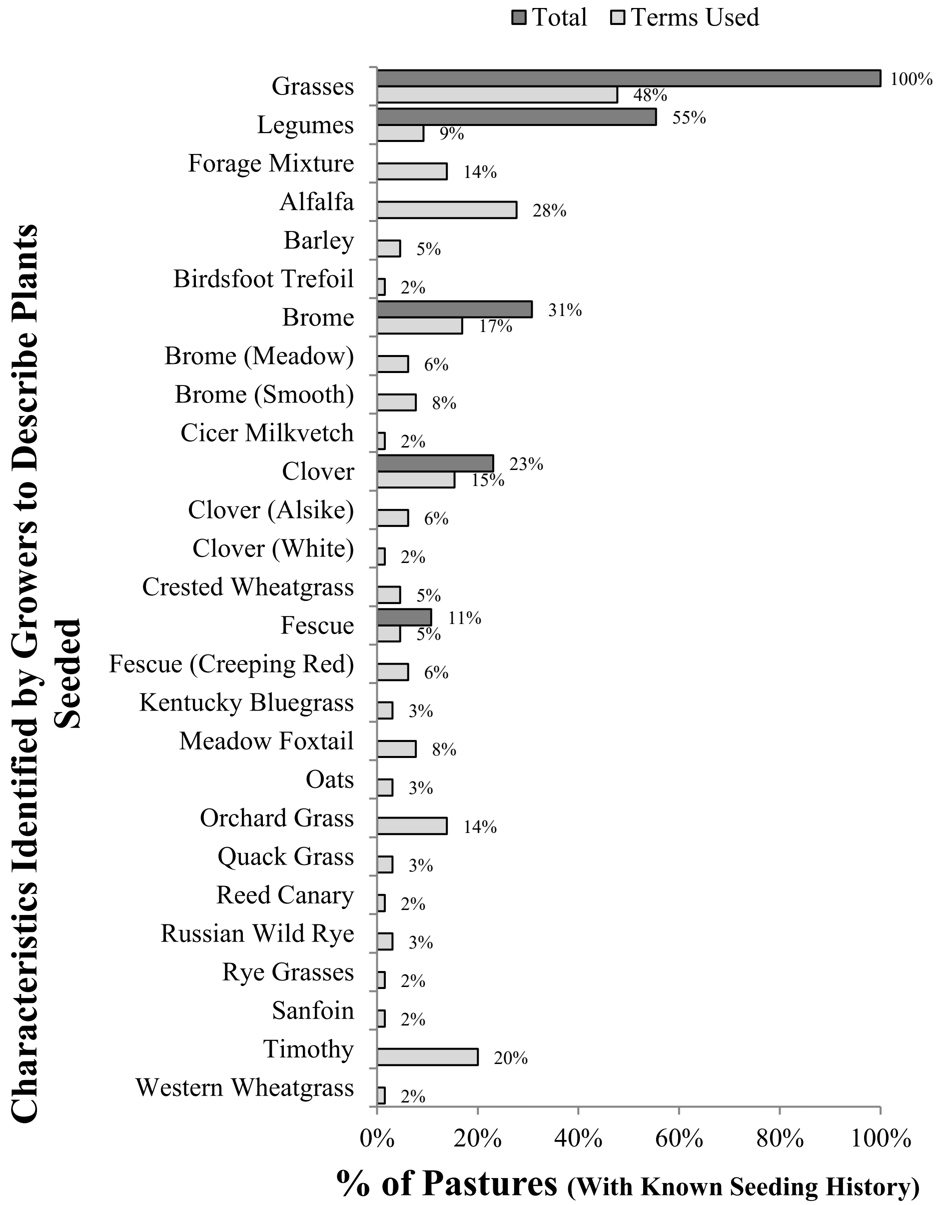


Figure 3.5. Summary of known seeding history for pastures following cultivation where producers were able to recall or estimate the seed mixture (N=65/102). 'Grasses', 'legumes', and 'forage mixture' were generic descriptions of species provided by managers. Totals of similar genera were also grouped together (i.e. brome, clover, and fescue).

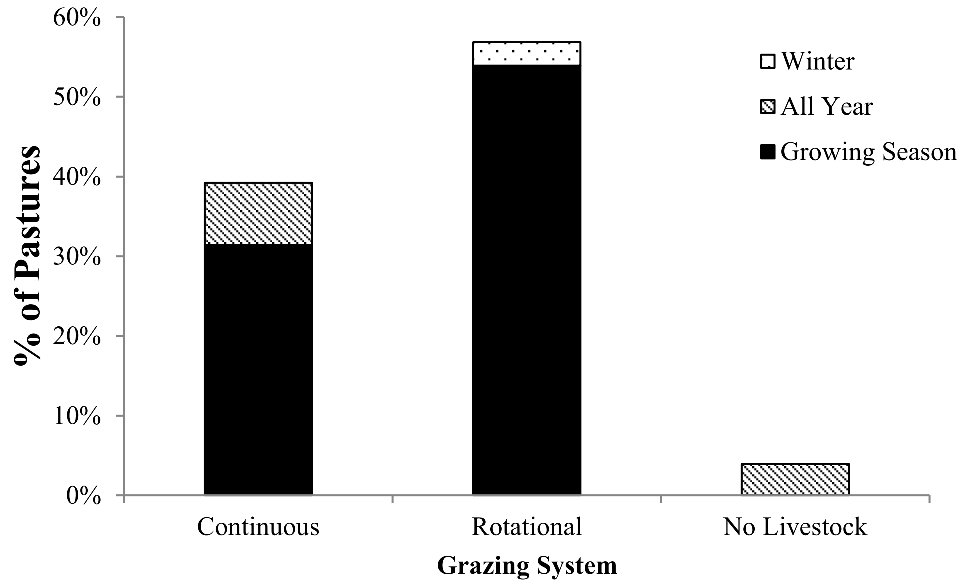


Figure 3.6. Summary of the grazing systems used based on the survey results. Continuous systems included pastures grazed only during the growing season and pastures in which animals were present year-round. Rotational systems included pastures where animals were rotated during the growing season, and pastures grazed only in winter. When no livestock are present, grasslands had been abandoned for multiple years (~10 or more); in at least one case the abandoned pasture was swathed.

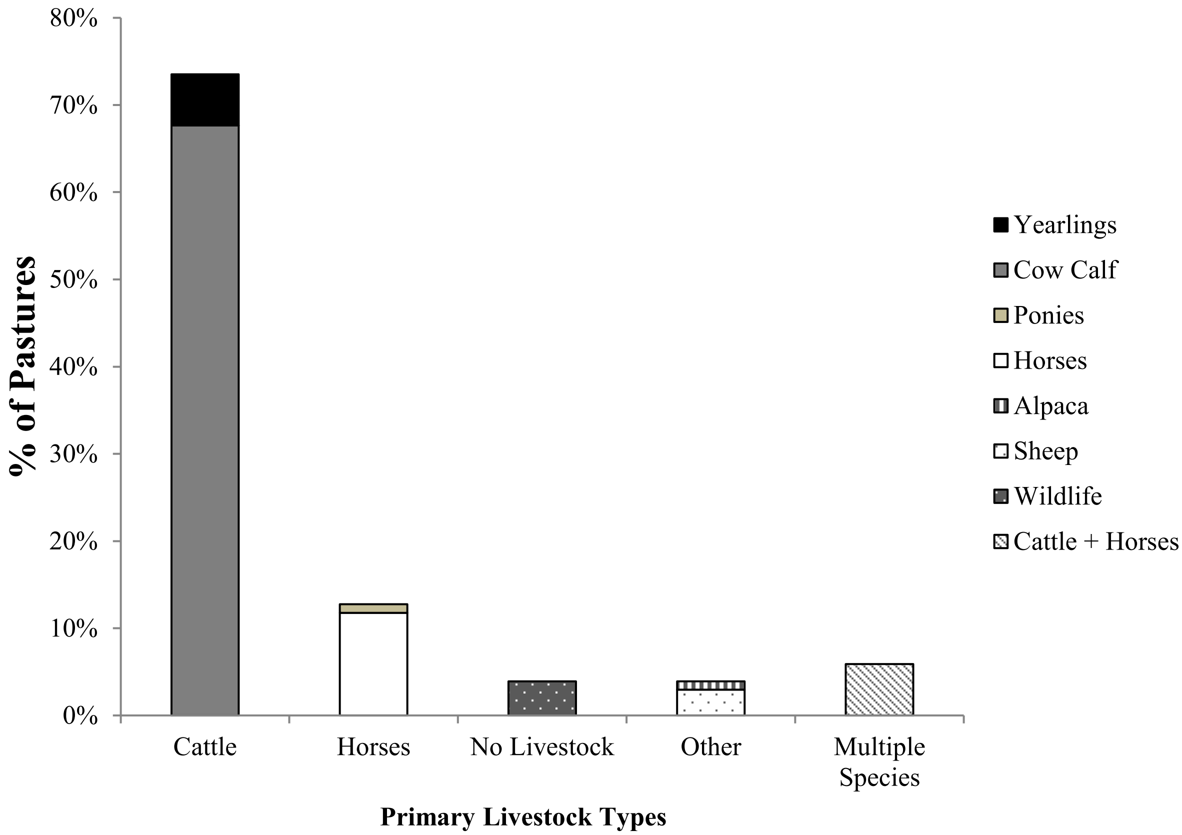


Figure 3.7. Summary of the identity of herbivores grazed in the pastures surveyed in north central Alberta pastures.

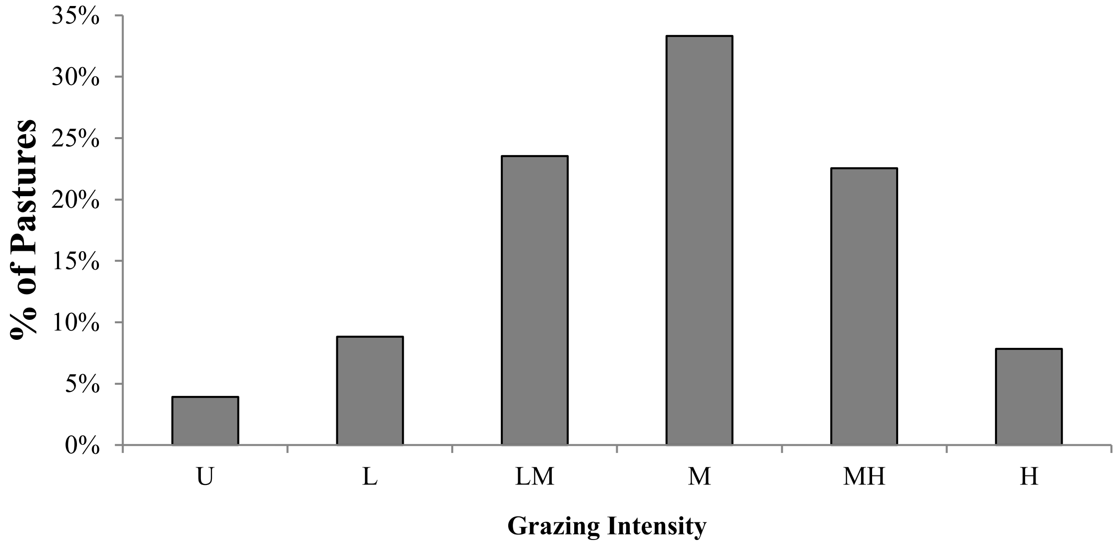


Figure 3.8. Summary of inferred grazing intensities for pastures as determined from the rangeland health assessment based on the observed utilization levels, soil compaction, productivity, species composition, etc. Abbreviations for intensities are as follows: U = not grazed, L = low, LM = low-moderate, M = moderate, MH = moderate-high, H = high.

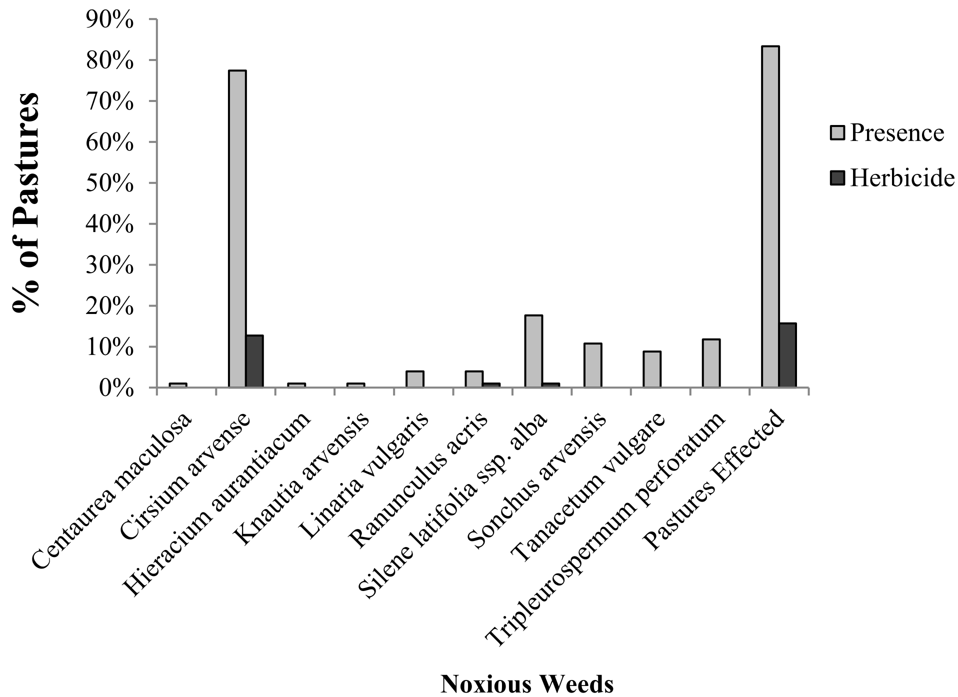


Figure 3.9. Occurrence frequency of noxious weeds detected during RHAs (grey) and the proportion of pastures where specific noxious weeds were targeted for removal with herbicide (black) in the last 3 years. Note that field scabious (*Knautia arvensis*), orange hawkweed (*Hieracium aurnatiacum*), and spotted knapweed (*Centaurea maculosa*) are currently classified as prohibited noxious and require control measures by law.

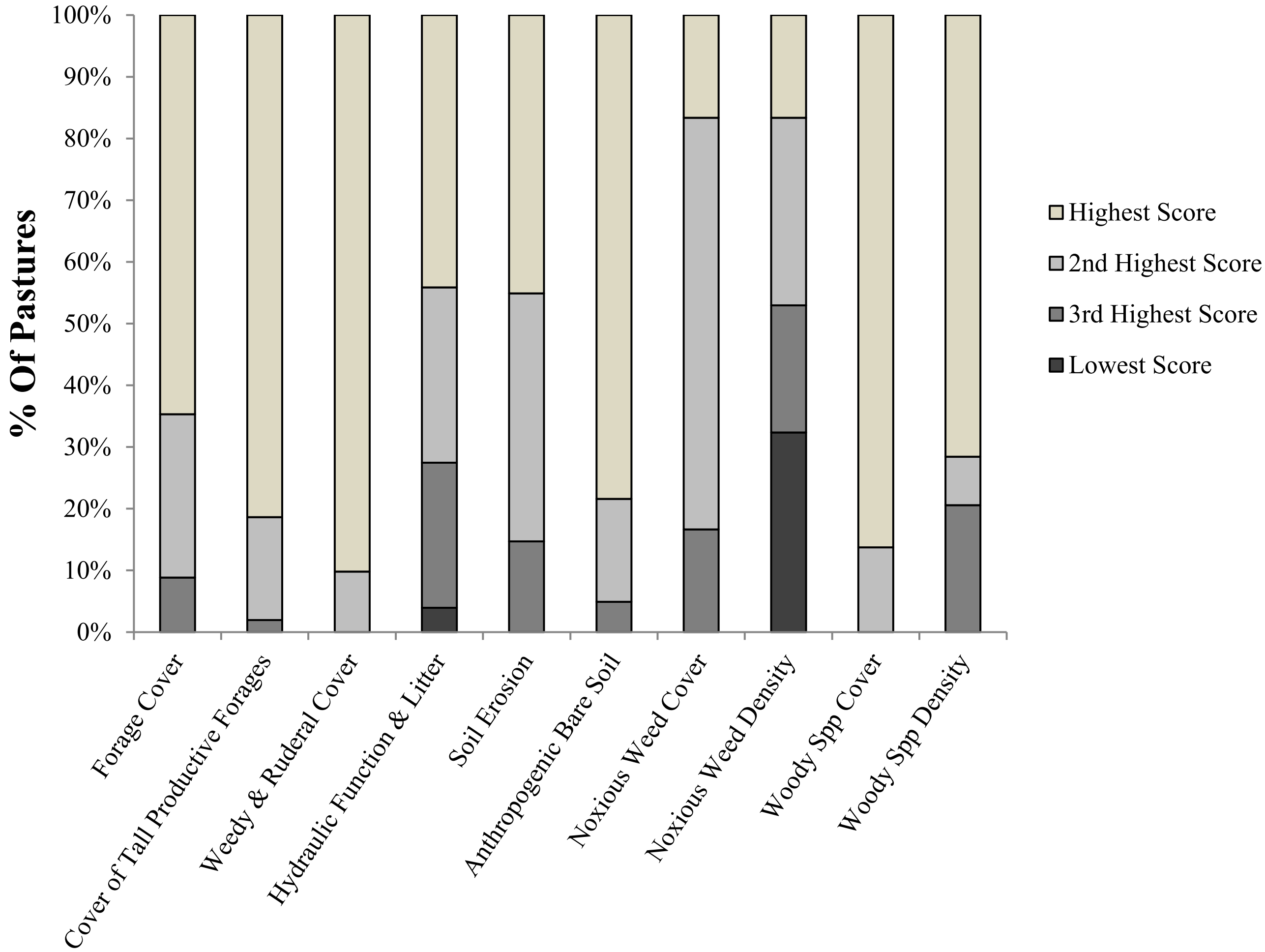


Figure 3.10. Summary of total scores from the rangeland health assessment. Scores are represented as proportions of maximum and ranked from highest (healthiest) to lowest (unhealthy). Scores are further summarized in Table 3.5 and described in Appendix A.2.

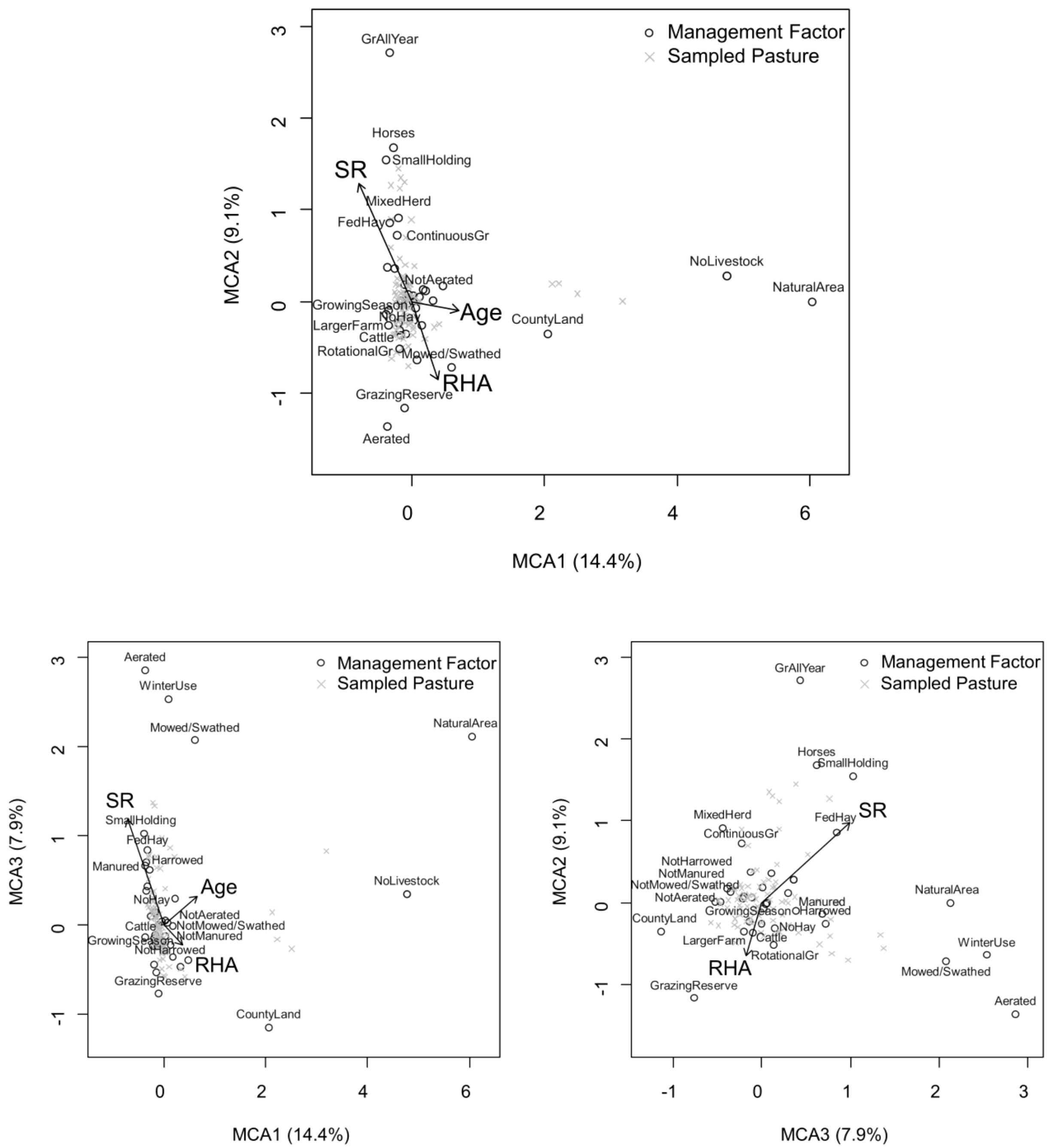


Figure 3.11. Multiple Correspondence Analysis (MCA) ordinations of categorical data representing current pasture management practices (distance=Eigen, dimensions=3). The first 3 axes describe 31.4 % of variation in management. Long-term historical management actions (i.e. fire and cultivation) were excluded, as were non-significant responses ($P < 0.01$). SR = livestock stocking rate; RHA = range health assessment scores. Variable responses summarized in Tables 3.6 and 3.7.

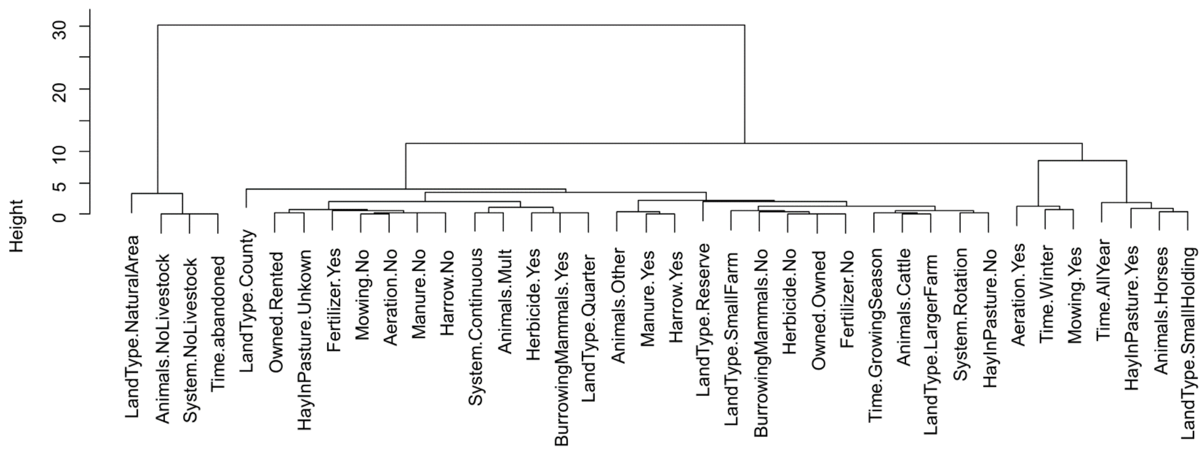


Figure 3.12. Cluster dendrogram of the first 3 MCA axes (distance=Euclidean, clustering method=ward) depicting the hierarchical breakdown of all management factors. Factors in closer proximity within lower levels are more likely to co-occur with one another.

Table 3.1. One-way ANOVA tests for relationships between stocking rate and density with management factors.

	Stocking Rate AUM/ha		**Stocking Density AU/ha**	
Management	**F Value**	**P Value**	**F Value**	**P Value**
Owned or Rented	3.477	0.066	3.460	0.067
Previous Cultivation	**3.426**	**0.038**	2.226	0.115
Grazing System	**102.740**	**<0.001**	**31.712**	**<0.001**
Timing of Grazing	**108.810**	**<0.001**	**44.932**	**<0.001**
System x Timing	**80.716**	**<0.001**	**42.558**	**<0.001**
Herbivore Type(s)	**55.112**	**<0.001**	**32.410**	**<0.001**
Herbicide	0.711	0.402	0.039	0.844
Fertilized	0.132	0.717	0.017	0.896
Manure Spreading	**5.167**	**0.026**	**6.807**	**0.011**
Harrowed	1.231	0.271	0.645	0.425
Aeration	1.070	0.304	1.776	0.187
Swathed or Mowed	0.662	0.418	0.147	0.703
*Fed Hay in Pasture Sampled	**9.569**	**0.003**	1.199	0.279
Burrowing Mammals	0.235	0.629	0.348	0.557
Fire (Survey)	0.005	0.947	1.186	0.280
Fire (Charcoal in Soil)	0.030	0.863	0.011	0.916
Rangeland Health				
Grazing Intensity	**50.535**	**<0.001**	**26.472**	**<0.001**
Health	0.688	0.506	0.033	0.967

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

*Includes only 58 sites from the 2013 survey

Table 3.2. LS Mean (±SE) stocking rate and density in response to pasture management.

Management	Treatment	Stocking Rate AUM/ha	Stocking Density AU/ha
Ownership	Owned	7.00 (±0.86)	1.94 (±0.24)
	Rented	2.85 (±2.58)	0.74 (±0.73)
Cultivation	Cultivated	6.18 (±0.91) a	
	Never Cultivated	2.14 (±2.91) b	
	Unknown	10.91 (±2.06) a	
Grazing System	Abandoned (None)	0.00 (±3.62) b	0.00 (±0.97) c
	Continuous	7.89 (±1.24) a	1.11 (±0.33) b
	Rotational	6.16 (±1.12) a	2.56 (±0.30) a
Timing of Grazing	Abandoned	0.00 (±2.69) c	0.00 (±1.03) b
	All Year	19.54 (±2.03) a	1.76 (±0.78) a
	Growing Season	5.00 (±0.66) b	1.89 (±0.25) a
	Winter	20.29 (±3.10) a	2.92 (±1.19) a
System x Timing	Abandoned	0.00 (±2.70) c	0.00 (±0.97) c
	All Year (Continuous)	19.54 (±2.04) a	1.76 (±0.73) b
	Growing Season (Continuous)	4.86 (±1.04) b	0.94 (±0.37) b
	Growing Season (Rotational)	5.08 (±0.87) b	2.54 (±0.31) ab
	Winter (Rotational)	20.29 (±3.12) a	2.92 (±1.12) a
Animals	Cattle	5.67 (±0.90) b	2.06 (±0.27) a
	Horses	13.65 (±2.30) a	1.47 (±0.69) a
	Multiple	8.93 (±3.45) b	1.09 (±1.03) a
	Sheep/Alpaca	8.41 (±3.45) b	1.66 (±1.03) a
	No Livestock	0.00 (±3.45) c	0.00 (±1.03) b
Manure Spreading	Manured	10.05 (±1.59) a	2.62 (±0.46) a
	Not Manured	5.43 (±0.92) b	1.55 (±0.26) b
Fed Hay	Hay	14.75 (±1.90) a	
	No Hay	5.05 (±1.07) b	
Grazing Intensity	U	0.00 (±3.24) c	0.00 (±1.05) b
	L	7.64 (±2.16) ab	1.82 (±0.70) a
	LM	4.56 (±1.41) b	1.71 (±0.46) a
	M	5.16 (±1.32) b	1.85 (±0.43) a
	MH	7.41 (±1.73) ab	2.29 (±0.56) a
	H	16.86 (±2.29) a	2.10 (±0.74) a

Bonferroni corrected.

Table 3.3. Summary of herbicide products chosen including their active ingredient, herbicide group (mode of action), and systemic/residual properties.

Product	% of Treated Pastures (N=16/102)	Active Ingredient	Herbicide Group	Chemical Family Group	Sys.	Res.
Banvel II ®	6.25	Dicamba	4	Benzoic acid	+	+
Curtail M ®	12.5	Clopyralid	4	Pyridine (Picolinic Acid)	+	+
		MCPA ester	4	Phenoxy-carboxylic-acid	+	+
Grazon ®	37.5	Picloram	4	Pyridine carboxylic acid	+	+
		2,4-D	4	Phenoxy-carboxylic-acid	+	+
Restore ®	12.5	Aminopyralid	4	Pyridine (Picolinic Acid)	+	+
		2,4-D amine	4	Phenoxy-carboxylic-acid	+	+
Roundup ®	6.25	Glyphosate	9	Glycine	+	-
Target/Sword ®	6.25	MCPA	4	Phenoxy-carboxylic-acid	+	+
		Mecoprop	4	Phenoxy-carboxylic-acid	+	+
		Dicamba	4	Benzoic acid	+	+
Tordon ®	6.25	Picloram	4	Pyridine carboxylic acid	+	+
Unknown	18.75	n/a	n/a	n/a		

Sys=Systemic action, Res=Residual soil properties

Note: one pasture was treated with Banvel II and Target/Sword.

Table 3.4. Summary of industrial disturbances reported by landowners in 2013.

Disturbance Type	Reported	% of Pastures (N=58/102)
Access	Roads & Rail	12.1
Mineral Extraction	Gravel	5.2
	Maral	1.7
Oil and Gas	Pipeline(s)	39.7
	Pumpjack(s)	3.4
	Well(s)	29.3
Not Reported	n/a	51.7

Reporting only 2013 survey results.
Pastures can have multiple disturbances (i.e. pastures with wells and pumpjacks also contain pipelines.)

Table 3.5. Summary of mean scores from the rangeland health assessments conducted on 102 pastures distributed across north-central Alberta during 2012-2013. Full details on the range health assessment used can be found in Adams et al. (2009). Also shown are the proportion of tame (n=90) and modified-tame (n=12) pastures falling in the maximum and minimum categories within a criterion. Tame pastures had a known history of cultivation and seeding, while modified-tame were not seeded and therefore comprised of a mix of native and naturalized tame species.

Abbreviated Range Health Question	**Range of Scores (interval)**	**Mean Pasture Score (±SD)**	**% of Pastures with Max Score**	**% of Pastures with Min Score**
1A. Forage cover in tame pasture	5-12	10.9 (±2.0)	73.3	7.7
1B. Forage cover in modified tame pasture	0-9	8.0 (±1.8)	75.0	0
2.1. Forage species shifts	0-14	12.6 (±3.2)	81.4	2.0
2.2. Weed distribution	0-14	13.3 (±2.1)	90.2	0
3. Hydrologic function and nutrient cycling	0-25	17.5 (±7.7)	44.1	3.9
4.1. Evidence of soil erosion	0-10	7.9 (±2.1)	45.1	0
4.2. Area of bare soil (adjusted for region)	0-5	4.4 (±1.3)	78.4	4.9
5.1. Noxious weed cover	0-5	3.0 (±1.2)	16.7	0
5.2. Noxious weed density	0-5	2.0 (±1.8)	16.7	32.4
6.1. Woody species regrowth	0-6	5.6 (±1.0)	86.3	0
6.2. Woody plant density distribution	0-4	3.0 (±1.6)	71.6	20.6
Max/mean score (tame pastures)	100	80.5 (±13.3)		
Max/mean score (modified tame pasture)	97	74.9 (±14.3)		

Table 3.6. Summary of management factors associated (i.e. correlated) with the MCA axes (Fig. 3.11). Only significant factors (P < 0.05) are listed.

	MCA 1		MCA 2		MCA 3	
Management	r^2	**P Value**	r^2	**P Value**	r^2	**P Value**
Aeration			0.08	0.005	0.33	<0.001
Burrowing Mammals			0.05	0.029		
Fertilizer						
Grazing System	0.93	<0.001	0.36	<0.001		
Harrowing	0.06	0.012			0.25	<0.001
Hay	0.07	0.022	0.16	<0.001	0.21	<0.001
Herbicide						
Herbivores	0.93	<0.001	0.52	<0.001		
Land Type	0.52	<0.001	0.36	<0.001	0.23	<0.001
Manure	0.05	0.026			0.15	<0.001
Mowing/Swathing			0.05	0.022	0.41	<0.001
Ownership						
Timing of Grazing	0.93	<0.001	0.64	<0.001	0.23	<0.001

Table 3.7. Significant management factors (P < 0.05) for MCA axes (Fig. 3.11).

		MCA 1		MCA 2		MCA 3	
Management	**Factor**	**β-Estimate**	**P Value**	**β-Estimate**	**P Value**	**β-Estimate**	**P Value**
Aeration	Yes			-0.30	0.005	0.58	<0.001
	No			0.30	0.005	-0.58	<0.001
Burrowing Mammals	Present			0.09	0.029		
	Absent			-0.09	0.029		
Fertilizer	Yes						
	No						
Grazing System	No Livestock	1.74	<0.001				
	Rotational	-0.86	0.042	-0.28	<0.001		
	Continuous			0.23	<0.001		
Harrowing	Yes	-0.14	0.012			0.21	<0.001
	No	0.14	0.012			-0.21	<0.001
Hay	Yes			0.28	<0.001	0.26	<0.001
	No			-0.21	0.006	-0.25	<0.001
	Unknown	0.20	0.007				
Herbicide	Yes						
	No						
Herbivores	Cattle	-0.48	0.004	-0.36	<0.001		
	Horses			0.50	<0.001	0.18	0.017
	Multiple			0.18	0.021		
	No Livestock	2.11	<0.001				
	Other						
Land Type	County	0.52	<0.001			-0.52	0.046
	Grazing Reserve			-0.50	0.003		
	Larger Farm			-0.16	0.002		
	Natural Area	2.62	<0.001			0.75	0.033
	Quarter			0.14	0.032		
	Small Farm						
	Small Holding			0.64	<0.001	0.32	0.002
Manure	Yes	-0.13	0.026			0.17	<0.001
	No	0.13	0.026			-0.17	<0.001
Mowing/Swathing	Yes			-0.17	0.023	0.44	<0.001
	No			0.17	0.022	-0.44	<0.001
Ownership	Owned						
	Rented						
Timing of Grazing	None (Abandoned)	1.94	<0.001				
	Growing Season	-0.67	<0.001	-0.32	<0.001	-0.37	<0.001
	All Year			0.92	<0.001		
	Winter					0.68	<0.001

Table 3.8. Significant biplot vectors ($P < 0.05$) describing pasture age, health, and stocking rate under current management and disturbance history (Fig 3.11).

	MCA 1		MCA 2		MCA 3	
Biplot	**r**	**P Value**	**r**	**P Value**	**r**	**P Value**
Pasture Age	0.21	0.034				
RHA Score	-0.23	0.021	-0.25	0.010		
Stocking Rate	-0.24	0.017	0.38	<0.001	0.38	<0.001

Chapter 4

Using producer surveys to link pasture management with vegetation composition, soil properties and rangeland health

4.1 Abstract

Northern temperate pastures experience a complex history of management factors, yet little is known of the extent to which these physical, management and social factors regulate plant communities and soil characteristics. In this study, plant community composition, range health, and soil properties from 102 pastures in Alberta's Central Parkland and adjacent Boreal region were related to management history collected from retrospective producer surveys. Producers were asked to identify pasture history (e.g. date of last cultivation, what species pastures were seeded with, if a fire event had occurred, etc.), contemporary grazing management practices (i.e. timing of grazing, grazing systems, livestock grazed, etc.), and other management (i.e. herbicide application, manure spreading, etc.) that could affect plant community communities and their soil.

Cultivation history was the primary driver of the plant community where previously cultivated pastures were dominated by *Poa pratensis* L. and *Bromus inermis* Leyss. (colloquially called tame pasture or grassland), eliminating many native species from the forage sward. Remaining semi-native grassland, identified as modified-tame during the rangeland health assessment, were altered by invasive cool-season grasses likely resulting from a history of excessive stocking. Soil fertility (C, N, and OM) was highest in tame communities, while modified-tame communities were associated with sandier soil, indicating historically productive soils were likely converted. Comparatively, grazing strategies had limited significant impact on plant communities and soils, and this was likely caused by excessive stocking in both continuous and rotationally grazed pastures. Ground cover was responsive to grazing management, where growing season grazing resulted in a thinner litter layer with less cover, and bare ground was twice as high with continuous stocking when compared to rotationally grazed pastures, which would translate into lower ecological function and rangeland health.

Fertilizer use reduced overall broadleaf plant cover, primarily from legumes, but also reduced ruderal grass and introduced ruderal forb cover, which corresponded with lower richness and diversity. Nutrient inputs resulted in higher litter cover and higher cover from plant shoots and crowns at ground level, likely supressing niche space for weeds. Manuring and harrowing were often paired resulting in similar effects on plant communities and soils increasing soil fertility (C, N, and OM) and salinity. Manure addition was associated with a handful of weedy indicator species, likely resulting from seed that passed through herbivore digestive tracts or propagation on stockpiled manure. Herbicide treated pastures had high introduced grass cover and were associated with seeded species like *Festuca rubra* and *Schedonorus pratensis*; *Cirsium arvense* was frequently reported as a target species (Chapter 3) and an indicator species for herbicide use. Compared to non-treated pastures, total noxious weed cover was marginally reduced, while legume and introduced ruderal forbs were also unaffected. It is possible that diverse methods of herbicide application (spot vs. broadcast spray) influenced the efficacy of reducing broadleaf cover at a landscape level but has positive implications for maintenance of legume populations. Finally, pastures identified through producer interviews as burned (diverse ignition sources) had dissimilar plant communities from pastures that had not burned in recent memory. In contrast, plant communities of pastures with indicators of a historical burn (charred woody debris in top 15 cm of top soil) were not dissimilar from pastures lacking evidence of a historical burn. Burned pastures had greater cover from native plants, attributed primarily to woody species and native forbs which corresponded with greater richness and diversity.

Rangeland health was higher in pastures with greater total cover from graminoids (primarily seeded, introduced grasses), low introduced ruderal forb cover, and low plant species richness. This likely resulted from the *Tame Pasture Health Assessment*'s emphasis on productive forage species. Hence, health was associated with factors like cultivated (tame) pastures or dormant season use, while year-round stocking of livestock was associated with the lowest health scores due to soil erosion, bare soil, noxious weeds, and loss of hydraulic function (litter).

4.2 Introduction

Temperate grassland plant communities and their response to management factors have been extensively studied using controlled experiments with treatments designed to isolate vegetation responses to variation in perturbations including: grazing intensity, frequency, timing or duration; fertilizer or manure application; or weed control. While these approaches are effective in isolating the effects of specific treatments, they cannot assess a multitude of management factors simultaneously impacting pastoral systems or the effects that site quality and grazing requirements or cultural factors such as holding size, off-farm employment and tending of companion animals can exert on management decisions. In north central Alberta's Central Parkland, exists a mosaic of residual grassland with diverse vegetation composition and divergent disturbance histories. Approximately 75% of pastures have a known history of cultivation and had plant communities dominated by cool-season introduced grasses and relatively few legumes, which fewer had no history or were uncultivated sustained communities containing native grasses and forbs (Chapter 3; Pyle et al. 2017). Pasture area was relatively small, >50% exceeded 65 ha but many were smaller acreages and small hobby farms. Previously cultivated and seeded pastures supported a stocking rate of 6.2 AUM, compared to 2.1 AUM in those without a history of cultivation, high stocking rates were similarly found in both continuously and rotationally grazed pastures (Chapter 3). Pasture management was variable, with manuring and harrowing common, and fertilizing, overseeding and aerating infrequent. Most contain noxious weeds, although the use of herbicides is limited. While prescribed burning is rare, most have evidence of fire (recent memory or historically).

Grazing can promote healthy functional grasslands under responsible management (Milchunas et al. 1988), but at excessive levels, can cause undesirable community shifts over time (Willms et al. 1985). Repetitive defoliation of palatable species can inhibit their persistence and competitiveness (Dyksterhuis 1949) and is particularly problematic for grazing-sensitive grasses in Alberta's Parkland such as plains rough fescue (*Festuca hallii*) or forage legumes like alfalfa (*Medicago sativa*). For native fescue grasslands in the Parkland of western Canada, this can cause non-reversible shifts within the plant community to new stable states (Briske et al. 2005; Laycock 1991; Westoby et al. 1989). Within fescue grasslands specifically, heavy grazing can lead to domination by cool-season grasses (Vujnovic et al. 2000) and/or weedy ruderal species (Grime 1979), together with grazing-tolerant invasive plants such as smooth brome (*Bromus inermis*) (Sinkins and Otfinowski 2012) and Kentucky bluegrass (*Poa pratensis*) (De Keyser et al. 2015; Tannas 2011; Tannas et al. 2015). Concerns have been raised with both the latter species, because despite being highly productive and desirable forages, these introduced grasses may impede the conservation of native grassland (Elsinger 2009, De Keyser et al. 2015; Deserrud and Naeth 2014; Gifford and Otfinowski 2013; Sinkins and Otfinowski 2012), in part because their greater production may allow livestock managers to employ greater stocking rates than they would otherwise use. Grazing induced changes in pasture composition are exacerbated under high grazing pressure (Smoliak 1974; Willms et al. 1985) and continuous grazing (De Bruijn and Bork 2006) and persist in the Parkland even after long-term recovery (Sinkins and Otfinowski 2012). Grazing systems are diverse, and each producer will adapt rotations, stocking rates, and land use strategies (i.e. water placement, fencing, etc.) to achieve unique management goals. In many cases however, there is a tendency for management to follow a utilitarian perspective, managing for a narrowly defined plant community dominated by a few tall and productive forages (Fuhlendorf et al. 2012). While conflicting perspectives exist on the benefits of rotational grazing relative to continuous grazing (Briske et al. 2008; Teague et al. 2013), continuous grazing remains common in western Canada (Josephson 1993). Josephson (1993) found that implementing rotational grazing in southwestern Manitoba improved net farm income per acre, and at that time remained an underutilized conservation tool.

Although grazing is a primary concern for livestock producers, it is not the only disturbance that can influence plant communities in northern temperate pastures of central Alberta. These landscapes have been markedly altered by European settlement, which first used (and then supressed) fire, and together with widespread land-use conversion into annual cropland, led to extensive modification of the northern rough fescue grasslands once covering most of the region (Bailey et al. 2010; Coupland and Brayshaw

1953). Native grassland conversion into cropland is driven by agricultural commodity prices and soil quality; although rates vary, the prairie pothole region loses about 1.33% of uncultivated grassland per year (Rashford et al. 2011). The Plowprint Report by the World Wildlife Fund [WWF] (2016) reported grassland loss due to cultivation as 6.95% between 2011-2012 (year before the study), 3.08% between 2012-2013, and 3.63% between 2013-2014 (final year) in the "Prairie Habitat Joint Venture" region that encompasses the Canadian prairies. Newly converted acres were most commonly planted to alfalfa (19.9%), followed by wheat (19.0%), and canola (12.6%) (WWF, 2016). Proportionally, Alberta and Manitoba contain the most fescue grassland at around 12% to 11.5%, while as little as 5.9% of the remaining northern fescue prairie is thought to remain in Saskatchewan (Gauthier and Wiken 2003). Other cultivated areas have been converted into introduced forages to support either a sizeable cattle industry, or other livestock and companion animals that are increasing coincident with hobby farm establishment, suburban sprawl, and industrial disturbances (i.e. access road, gravel pits, oil and gas, etc.) leading to slow degradation of remaining Parkland patches (Rowe 1987). Thus, many of the grasslands that remain in the region are semi-natural or comprised primarily of introduced forages, and all areas, particularly those previously cultivated, may exhibit a prevalence of agronomic weeds. In many cases the Parkland region has been referred to as an endangered ecosystem and some postulate that aside from marginal remnants on poor quality ecosites, native Parkland formations may become extinct in the future (De Keyser et al. 2015; Rowe 1987). Gossner et al. 2016 found that land-use intensification homogenized grassland communities at multiple trophic levels and taxa (e.g. soil micro-fauna, plants, and arthropods), meaning the loss of diverse functional ecosystems caused trophic cascades as the native habitat is functionally modified.

Use of prescribed fire in the region is a tool rarely used to promote grassland health but is still occasionally used to remove woody vegetation and facilitate forest conversion into grassland. Where pastures are considered relatively poor in productivity, managers are more inclined to plow the land and reseed to high yielding forages, if not switch to annual cropping. Alternatively, some managers may try to rejuvenate ‘decadent’ pastures (Lardner et al. 2000; Lardner et al. 2001; Lardner et al. 2002; Malhi et al. 2000) using a variety of strategies, including burning, heavy fertilization, or the use of herbicides to control undesirable weeds and encroaching woody vegetation (aspen, snowberry, *Rosa* spp., etc.) (Bowes 1981; Bowes and Spurr 1996). Fertilizer is an amendment commonly applied to improve pasture performance and can release pastures in central Alberta from nitrogen deficiency (Malhi et al. 2000). In comparisons of various treatments, heavy applications of fertilizer were most effective in restoring forage yields of decadent forage stands (Lardner et al. 2000) but came at a significant economic cost to producers. In contrast, only minor benefits were found from the use of burning and aeration in renovating pastures (Lardner et al. 2000). Spreading manure can also promote more abundant palatable forage and lead to greater forage availability (Blonski et al. 2004). While low amounts and/or infrequently applied manure may be capable of maintaining pasture composition (Bork and Blonski 2012), excessive or improperly sourced manure can place vegetation at greater risk of invasion by persistent undesirable species noxious weeds (Pleasant and Schlather 1994). Harrowing can also accompany manure spreading in order to distribute thick manure more evenly on treated areas, thereby adding mechanical to nutrient addition impacts on vegetation and soil. Aeration or ‘spiking’ (Lardner et al. 2000) is used to reduce the negative impacts of trampling and soil compaction by cattle, as well as alleviate sod-bound soil of poor air entry and is an alternative to cultivating. However, a study by Malhi et al. (2000) found that mechanical aeration of central Alberta pastures did not improve forage production. Other common amendments producers impose include swathing or mowing, which can provide short-term control of perennial noxious weeds (Trumble and Kok 1982) or collect a hay-crop of otherwise unutilized forage. Locally, irrigated pastures are rare as soil moisture is not often a limiting factor in northern temperate pastures.

Prevalence of noxious weeds is often a symptom of problematic management (i.e. excessive stocking rates and lack of recovery) causing deterioration of initial pasture conditions, and thus, reduced competitive ability of forages and increased niche availability due to changes in microsite conditions (i.e.

bare soil, litter cover, nutrient availability, etc.). Weeds can have complex relationships with desirable forages, which in turn, vary markedly with environmental (soil and climate) conditions (McLeod et al. 2015). Presence of noxious weeds can decrease forage yield (Grekul and Bork 2004). Strategies to control weeds can include direct control with herbicides or mowing and contribute to positive forage yield increases (Grekul and Bork 2007), but also come at the expense of altering ecosystem function. The latter includes removing beneficial legumes along with broad-leaf weeds (Grekul et al. 2005; Grekul and Bork 2007), as well as limiting legume recovery potential (Laird 2014; Miller et al. 2015). In Alberta, control of weed populations can be enforced through the *Weed Control Act* (Government of Alberta 2010). While herbicides are often an effective tool to cause immediate reductions in weed cover, integrated weed management strategies are often the best, and include a combination of herbicides, fertilization to enhance competition from forages, and carefully timed defoliation with cattle to reduce weed populations (Grekul and Bork 2007; De Bruijn et al. 2010; De Bruijn and Bork 2006).

Rangeland health assessments (RHA) are a tool developed and used in Alberta to measure the response of plant communities and associated soils to pasture management over time, as it is based on a series of questions reflecting the status of the plant community relative to fulfilling certain functions. Lower health scores are intended to alert the manager to existing or emergent concerns and help them adapt their management to overcome these. In Alberta, the Rangeland Health Assessment (RHA) protocol was introduced to address short comings in the traditional 'range condition' method, where site stability, soil, and divergent successional trajectories were not considered (Adams et al. 2005). Values and benefits of healthy rangelands for livestock producers include: lower feed costs, renewable and reliable forage, stability of forage during drought, lower maintenance and input costs (i.e. weed control, fertilizers, etc.), and reduced concern for noxious weeds (Adams et al. 2005). Plant communities abundant in palatable forage species for tame pastures, and in the case of native grasslands or modified native grasslands, native grasses, are quantified as healthier when they have more productive forage species, particularly large-statured species, with fewer disturbance-adapted species with undesirable characteristics (i.e. annual weeds and unpalatable perennial forbs). Hydraulic function and litter accumulation are heavily weighted variables in RHA, as litter offers numerous important functions including moisture retention by preventing run-off and evaporation (Deutsch et al. 2010b; Sharafatmandrad et al. 2010), covering bare ground, creating habitat for micro-flora and fauna, and facilitating seed bank formation (Facelli and Pickett 1991; Willms and Quinton 1995). In the Parkland, litter is particularly important for maintaining soil moisture in June and July, exhibiting positive effects on community productivity (Deutsch et al. 2010a). While excessive litter loss is also known to directly reduce herbage production (Deutsch et al. 2010b; Willms et al. 1986; Willms et al. 1993), tame grasslands in the Parkland can exhibit improved productivity temporarily under reduced litter (Deutsch et al. 2010b). As standing and fallen litter decrease under increasing grazing intensities, they serve as an indicator of over use (Naeth et al. 1991). Decreased litter is also associated with increased bare ground (Naeth et al. 1991), which can exacerbate grazing induced erosion. The tame pasture assessment is most suitable for central Alberta given the large amount of land once cultivated but now in perennial pasture. Ultimately, RHAs may provide an effective tool to link management activities with pasture agro-ecological function, and in the process, highlight opportunities for improvement in management.

The objective of this study is to assess plant community and soil responses across a large sample of pastures in northern temperate pastures of the Central Parkland and neighboring Dry Mixedwood natural subregions and interpret those data in relation to specific management history data collected directly from producers managing those areas. Second, this assessment will use observed plant community and soil characteristics to further understand the relevance of RHAs, with particular attention to the latter's responsiveness to management actions. This information will provide key insight on the impact of various management actions on pasture biophysical conditions (vegetation and soils), including metrics of rangeland health.

4.3 Methods

4.3.1 Study Site Selection

We surveyed a total of 102 pastures during 2012 (N=44) and 2013 (N=58) between May 24 and July 6, distributed across four counties (Leduc, Parkland, Strathcona, and Sturgeon County) within an 80 km radius surrounding the city of Edmonton, Alberta (Figure 3.1). The middle of the sampling area is located at the northern extent of north central Alberta's Central Parkland natural subregion, which is characterized by Black Chernozemic soils (i.e. organic matter of 4-10%), and receives 445 mm of precipitation annually, with about 77% falling during the growing season (April through September) (Fig. 3.2). About half of the pastures sampled occurred in the Central Parkland (N=50), while the remainder occurred within the neighboring boreal natural subregions: Dry Mixedwood (N=50) and Central Mixedwood (N=2). Although precipitation levels are similar, soils in the latter regions are lower in organic matter, resulting in soils varying from Eluviated Black Chernozems to Gray Luvisols. The previously cultivated and seeded nature of pastures within the boreal zone make them strongly resemble the Parkland pastures in composition (Donkor et al. 2002). The large and well-distributed sample size ensured a wide range of pasture types were represented in the survey, and included both older, decadent pastures (often *Trifolium* spp. dominated) and more recently established high-performance pastures containing *Medicago* spp., with a corresponding wide range of management activities.

Pastures were selected using a stratified random approach and were separated by at least 800 m. Pastures were acquired through consultation with municipal county staff, then driving roadsides to visually identify suitable fields, and in some cases, managers referred us to neighbors and family. Suitable pastures had to fit a 260 m long transect, with suitable buffer zones from wetlands, forests, and fence lines (outlined in 4.3.3 describing the plant community survey) meaning pastures had to be a minimum of ~ 10 acres, with larger pastures given preference. If a producer owned or rented multiple pastures, duplicate pastures were only sampled if they were separated spatially, although select exceptions (N=2) were made if management was distinctly divergent histories (i.e. a previous cultivated vs. non-cultivated field) or if pastures were seeded with different forage mixtures and when the land was last cultivated. Acquisition of sites was further constrained by the willingness of landowners to grant permission to their land, which was denied less than 5% of the time. Finally, sampling locations were only selected if pastures were an adequate size (i.e. large enough for the sampling transect; see Fig. 4.1), if there was evidence of grazing in the past (i.e. we made an effort to avoid sampling hay fields), and there was a preference to choose larger pastures that contained cattle over smaller single pastures on hobby farms with horses. Further information on management factors is provided in Chapter 3.

4.3.2 Determining Producer Management

Producer management information was acquired for 102 pastures through a retrospective, in-person interview, described in detail in Chapter 3. The interview (see Appendix 3.1), approved by the Research Ethics Office at the University of Alberta, was designed to identify all historical and current land use practices on the pastures in question. Surveys were intended to identify all key management activities that may influence the soil, plant community and associated seed bank composition (discussed in Chapter 5). If land had never been cultivated, or the date of last cultivation was unknown (often the case with grazing lease holders or when the land was cultivated before their possession), this was recorded as well. Other data on management collected included grazing history (number of animals, type of herbivore and timing of use), whether the land had been previously seeded to introduced forages, when the pasture was last cultivated (tilled, aerated or harrowed), fertilized (chemical or manure), or sprayed with herbicide in the last three years, and whether the pasture had been otherwise disturbed (burned, impacted by oil and gas disturbance, etc.).

4.3.3 Plant Community and Rangeland Health Assessment

Following the in-person interview, a field assessment was conducted. During field sampling, areas of each pasture were avoided that could cause edge effects such as field margins (>10 m from fences), wetlands (>30 m), and areas strongly influenced by forest (>10 m). To initiate sampling, a randomly selected point in the pasture was located that met our criteria, and was relatively uniform in ecosite conditions (aspect, slope, elevation, drainage, soils, etc.) and remained distant from disturbances (roads, well sites, feeding areas, etc.) in a representative area of the pasture. From that point a 260 m long ‘W-transect’ was formed (Fig. 4.1), as adapted from Thomas (1985). Plant community composition was assessed at 9 equidistant locations along the W-shaped transect using a 50 x 50 cm (0.25 m^2) quadrat. Foliar cover by individual plant species, together with ground cover (litter, bare soil, manure, rock, moss lichen, and basal vegetation (stems, shoots, and crowns)) was visually estimated to the nearest 1 percent in each quadrat (%); cover <1% was recorded as trace (0.1%). Ground cover totalled 100%, while foliar cover was estimated independently by species. In addition, litter depth was measured at 5 points in each frame (4 corners and centre). Cover of plant species were partitioned into biologically significant groups such as: total native cover and total introduced plant species cover; total broadleaf (forbs) cover vs graminoids (grasses, sedges, rushes); and functional groups such as total legumes, ruderal grasses, noxious weeds, introduced ruderal forbs, introduced grasses (often seeded as forage), native ruderal forbs, native perennial forbs, native perennial grasses, and native grass-likes (sedges, rushes, etc.).

Rangeland health was assessed using the *Tame Pasture Assessment Form* prepared by Alberta Environment and Parks, formerly Alberta Environment and Sustainable Resource Development (Government of Alberta 2010), was described in Chapter 3. In brief, the RHA evaluated pasture conditions based on five criteria, including the status (composition and structure) of existing vegetation to the desirables (i.e. tall, productive forages) and non-desirables (e.g. weedy, woody and non-palatable species), the abundance of litter, base soil, and evidence of erosion. For reference the RHA form used in the assessment of pastures is provided in Appendix A.2, and resultant RHA scores for all pastures summarized in Chapter 3. When classifying pastures as tame or modified-tame, we were more lenient with classifying pastures as modified-tame based on the guidelines which specified pasture plant composition had to be comprised of more than 50% native cover. This was modified further where native grass cover was present, especially of plains rough fescue (*Festuca hallii*) or intermediate oatgrass

(*Danthonia intermedia*), or high native forb cover (which was usually over 50%), each of which led to assignment of plant communities to the modified-tame category.

One additional amendment made while assessing range health was to include all introduced and potentially seeded forages like creeping red fescue (*Festuca rubra*), Kentucky bluegrass (*Poa pratensis*), and white clover (*Trifolium repens*), as desirable forages, as these species all contribute to favorable scoring in sections 1A, 2.1, and 2.2 (see Appendix A.2). Many species described as grazing-induced forages were actively seeded by producers and therefore should not be discounted in the RHA. During assessment, pastures receiving heavier grazing would have received lower scores under Q2.1 (i.e. assessment of forage species shifts by scoring the cover of tall productive forages) because desirable forages would have been underperforming, having short stature and low productivity. Additionally, native grasses (which were relatively rare) were included in the forage cover for pastures classified as tame, because although guidelines suggest otherwise, native species can still contribute to the agro-ecological function of these pastures. Naturalized dandelion, which despite being recognized for having forage value, was not included in desirable forage cover while scoring.

4.3.4 Soil Sampling and Properties

Soil cores (n = 10 to 15) were plunged randomly across each field, and after the surface LFH (i.e. mulch) was removed, the 0-15 cm mineral soil layers were combined to produce one composite sample for each field. Samples were dried at 55°C, sieved at 2 mm, and later assessed for soil physical properties, including % organic matter (OM), total nitrogen (N), total carbon (C), pH, electrical conductivity (EC), and texture. Levels of OM were quantified by burning 10 g of soil in a furnace at 450°C for 4 hr and measuring the subsequent mass loss. EC and pH were measured in a soil solution that was one-part soil and two-parts water. Soils were shaken for at least 30 minutes before measuring pH, and the soil solution filtered before measuring EC. Total carbon and nitrogen were measured using a LECO TruSpec CN elemental analyzer (LECO Corporation, St. Joseph, MI, USA). Samples were ground to a powder with a ball mill to ~ 0.1 mm (Retsch MM400 Mixer Mill, Retsch, Haan, Germany) and fumigated with HCl beforehand to remove inorganic C present as carbonate in alkaline soils (note that all soils were treated similarly). Soils from north Central Alberta typically had OM exceeding 5% for the majority of sites, thus all soil samples were pre-treated before texturing. OM was removed by applying small volumes of hydrogen peroxide to ~60 g of soil until soils achieved a color change and the reaction ceased (Lavkulich and Wiens 1970; Mikutta et al. 2005). Texturing was then performed on pre-treated soils using the hydrometer method (Bouyoucos 1927), where 40 g of soil and 4 g of sodium hexametaphosphate (Calgon) were suspend in 1 L sedimentation tubes, and the proportion sand, silt and clay subsequently quantified. In 2013, soil compaction was measured at 45 sites using a soil surface penetrometer.

4.4 Statistical Analysis

4.4.1 Plant Community and Soils

Two approaches were used to assess the impact of management factors on pasture characteristics, including vegetation attributes (richness, diversity, native, introduced, etc.), soil characteristics (OM, C, N, pH, EC, texture), and range health conditions. The first was a direct test of management factors on pasture biophysical attributes using ANOVA, while a more in-depth assessment of management impacts on plant species composition was conducted using multivariate analytical techniques.

To facilitate parametric analysis, continuous plant community, soil, and environmental variables from all sites were initially examined visually for normality, with residuals tested using the Kolmogorov-Smirnov testusing the *lillie.test* function from the *nortest* package (Gross and Ligges 2015), as well as homogeneity of variances using Levene's test in R software (Glass et al. 1972; R Core Team 2017). Many variables required transformation before analysis with univariate methods. Square root (total broad leaf cover, legume cover, introduced ruderal forb cover, soil surface compaction) and logarithmic (Pielou's evenness, soil OM, sand, clay, basal vegetation cover, litter depth) transformations were used for positively skewed data, while a square (Simpson's diversity) transformation was used for negative skew.

The effect of each management factor on each continuous vegetation and soil variable (e.g. soil organic matter, litter depth, indices of diversity, etc. [variables listed in Table B.1]) was then tested in a one-way ANOVA using Type III sums of squares and LS (least-squared) means because of unequal sample sizes within each level of management factors and RHA categories (results in Appendix B). LS means and contrasts were derived from the *lsmeans* package (Lenth 2016). Where data were unable to be transformed [total native cover, total introduced cover, ruderal grass cover, noxious weed cover, native ruderal forb cover, native perennial forb cover, native perennial grass cover, graminoid (including sedges, rushes, etc.) cover, woody (shrubs and trees) cover, species richness, soil C, soil N, bare ground cover, and manure cover], a Kruskal-Wallis test was used in R with in the *agricolae* package (De Mendiburu 2017), which also provided Bonferroni adjusted mean ranks for subsequent contrasts. Variables that met assumptions without transformation were total graminoid cover (including Poaceae and grass-like taxa), seeded (i.e. introduced vegetation) grass cover, Shannon's diversity, soil C:N ratio, soil pH, % silt, and litter cover.

Detailed plant community composition across all 102 pastures was analysed using a combination of permutational multivariate analysis of variance (perMANOVA), non-metric multi-dimensional scaling (NMDS), and indicator species analysis (ISA). Differences in plant community composition in relation to the principle management questions were tested using perMANOVA in R with the *adonis* function in the *vegan* package set to run 999 permutations (Oksanen et al. 2017). Due to the unbalanced experimental design of management factors (i.e. it was impossible to know survey responses in advance of the producer interview) and differences in multivariate spread among factors (Anderson 2005), thus we tested each management factor individually. When testing for differences in community composition among pastures where animals were given supplemental feed, we only analysed pastures sampled in 2013 (N=58) due to the absence of this question in surveys performed the year prior. Data were also analyzed this way for the ISA (i.e. separately by management factor). After testing for plant community differences within individual management factors, we tested for all interactions among the latter. Once significant factors and interactions were identified, contrasts were performed among the treatment levels (e.g. cultivated, not cultivated, unknown cultivation history) within each management factor with perMANOVA.

NMDS was used to graphically explain the relationships between plant species composition, management factors obtained from the producer surveys, ancillary environmental variables, and rangeland health metrics from all 102 pastures. Ordination was performed in R software using the *metaMDS* function in *vegan* using the Bray-Curtis distance metric. Given the large number of variables, assessment of ordinations was limited to the first two dimensions. Resulting ordinations were graphically displayed using joint biplots, together with vectors for major plant species and centroids of environmental variables having significance at $P < 0.05$ determined by the *envfit* function in the *vegan* R package. In R the proportion of variance described by each axis is not available. An ISA (indicator species analysis) was used to identify specific plant species that responded significantly increased in response to individual management factors, using the *indicspecies* package (De Caceres and Legendre 2009).Significant indicator species were included in the NMDS plots describing significant management factors (Figure 4.4).

In the final step, unique plant community types were identified from all the pasture composition data using the Bray-Curtis distances of plant community cover, and clustered in a dendrogram using the silhouette widths of ward distances between sites (Borcard et al. 2011). Similar sites were then analysed with an indicator species analysis. Plant communities were then described by their dominant species followed by an ISA. This information is presented in Appendix B.3.

4.4.2 Rangeland Health

RHA scores, both total and for each category, were tested with one-way ANOVA using Type III SS (sums of squares) and LS (least-squared) means for every management factor and each RHA category. Plant community characteristics and soil properties were tested as predictors of RHA score using multiple linear regression (MLR); variables were eliminated using a forward step-wise selection process.

Raw RHA scores were tested using perMANOVA against management factors, using both Euclidean and Bray-Curtis distance, to detect significant shifts in RHA parameters (i.e. forage cover, hydraulic function, erosion, etc.). NMDS was used to describe relationships between pasture RHA scores using Euclidean distance. To identify RHA parameters that responded negatively to management factors, we ran a multi-pattern analysis on inversed RHA scores.

The relationship between the range health and plant community are discussed in Appendix B.4 because many relationships between plant community characteristics and RHA questions are likely correlational.

4.5 Results

4.5.1 Plant Community

Pastures in north central Alberta were dominated by cool-season, introduced, forage grasses (Fig. 4.2), with Kentucky bluegrass (*Poa pratensis*) and smooth brome (*Bromus inermis* subsp. *Inermis*) contributing the greatest foliar cover (Table 4.1). Median cover of legumes was second highest followed by introduced ruderal forbs, relatively low amounts of native perennial grass and forb, as well as noxious weeds. All other cover groups (e.g. shrubs) contributed very little to overall cover. Importantly, pastures in the Central Parkland did not differ in floristic composition from pastures in the boreal (Dry Mixedwood and Central Mixedwood) ($P = 0.191$; Table B.3.2). Only a handful of indicators were indicative of natural regions with prickly rose (*Rosa acicularis*) occurring more commonly in the boreal, and agronomic weeds such as shepherd's purse (*Capsella bursa-pastoris*) and knotweed (*Polygonum aviculare*) occurring in the parkland ($P < 0.02$; Table B.3.3).

History of previous cultivation had a significant effect on plant community composition ($P = 0.016$; Table 4.2); which reflected the two types of grasslands, tame and modified-tame ($P = 0.002$; Table B.4.1), identified through the RHA. Uncultivated pastures retained cover of numerous species of native grasses and forbs (Table 4.4), and the total cover of native plants was also higher due to greater native perennial grasses, forbs, and graminoids ($P < 0.05$; Tables 4.6 to 4.9). Correspondingly, total cover of introduced species was lower in uncultivated fields, primarily through a reduction in introduced forage grasses. Cultivation history also effected all measures of species diversity tested, with richness, Shannon's diversity, and Simpson's diversity highest in uncultivated pastures ($P < 0.007$), and Pielou's evenness lowest ($P = 0.023$) (Tables 4.10 and 4.11). Richness, diversity, and evenness of pastures with an 'unknown' cultivation history resembled pastures with a known cultivation history.

Different grazing systems and timings was not significantly correlated to plant community composition or indices of diversity, although introduced species were an indicator of continuously grazed pastures ($P = 0.039$) and native species and graminoids (sedges, rushes, etc.) were more abundant in pastures that were abandoned ($Ps < 0.051$) (Table 4.5). Grazing induced increases in the forage white clover (*Trifolium repens*) and was indicative of both continuous and rotational grazing systems ($P = 0.017$; Table B.1.2.1). When single pastures were used year-round, weedy forbs like common plantain (*Plantago major*) become more abundant ($P = 0.035$; Table B.1.2.1). Total cover of grasses and graminoids combined remained lowest in pastures grazed by horses at 55.8%, while other herbivores were associated with cover ≥ 64.7% ($P = 0.055$; Tables 4.6 and 4.7).

Qualitative assessment of grazing intensity revealed some trends in plant community response. Pastures non-grazed or grazed at low to low-moderate intensities retained native perennial grasses ($P = 0.043$), while native perennial forbs were removed at the highest grazing intensity ($P = 0.027$; Table B.1.2.1). Higher grazing intensities were associated with ruderal species like foxtail barley (*Hordeum jubatum*), common pepper grass (*Lepidium densiflorum*), and stinkweed (*Thalaspi arvense*) ($P < 0.05$; Table B.1.2.1).

Although herbicide treatment was not a significant indicator of plant community differences ($P = 0.232$; Table 4.2), an indicator analysis found that Canada thistle (*Cirsium arvense*) ($P = 0.015$) was an indicator of herbicide treated pastures along with red fescue (*Festuca rubra*) and meadow fescue (*Schedonorus pratensis*) (Table B.1.2.1). Recently sprayed pastures were associated with high cover contributions from introduced species (> 90%), with total graminoid cover primarily attributed to forage grasses (Ps < 0.05; Tables 4.6 to 4.9), and select noxious weeds (marginally significant, P = 0.089; Table 4.8). There was low expression of both native ruderal and perennial forbs (Ps < 0.05). Native ruderal forbs were a weak indicator of pastures that had not been sprayed recently (P = 0.065). Herbicide use was linked to lower Shannon's diversity (P = 0.027), while there were trends for decreased Simpson's diversity and overall richness as well (Tables 4.10 and 4.11).

Application of fertilizer was associated with an abundance of meadow brome (*Bromus biebersteinii*) (P = 0.031), while non-treated pastures had an abundance of the legume white clover (*T. repens*) and alsike clover (*T. hybridum*) (Ps < 0.002; Table B.1.2.1). Overall, fertilized pastures had lower forb cover including significant reductions in legumes and introduced ruderal forbs. Use of fertilizer was associated with lower plant community richness and diversity (Shannon's and Simpson's) (Ps < 0.018) (Tables 4.10 and 4.11).

Manure addition had a near significant influence on plant community composition (P = 0.061) (Table 4.2), with introduced ruderals like peppergrass (*Lepidium densiflorum*), stinkweed (*Thalapsi arvense*), and the noxious weed white cockle (*Silene latifolia* sbsp. *Alba*), all favored by manure (Table B.1.2.1). Manured pastures had higher cover from introduced species and lower native cover with cover of native perennial forbs and woody species significantly reduced (Ps < 0.05). Harrowing had no effect on community composition but followed similar trends in cover and indicator species. There was a marginally significant trend at P < 0.1 for increased evenness when pastures were harrowed, aerated, and swathed or mowed (Table 4.10 and 4.11). Ruderal grasses were more abundant in aerated pastures (P = 0.009; Table 4.8) but contributed little to vegetation cover at 3.4±1.0 % (Table 4.9).

Use of supplemental feed in pastures was correlated with divergent plant communities (P = 0.033; Table 4.2) as indicated by the inclusion of weedy mustards in community cover (Table B.1.2.1). Feeding hay was associated with reductions in total broadleaf cover and native species cover (Ps < 0.05; Tables 4.6 and 4.7), and likely contributed to near significant reductions in legume cover and native perennial forbs (Ps ≤ 0.066; Tables 4.8 and 4.9). Loss of forb and native cover corresponded with lower richness and diversity (Ps < 0.015; Tables 4.10 and 4.11).

Burrowing mammals, which were frequently identified as pasture pests, were associated with near significant shifts in plant community composition (P = 0.099) (Table 4.2). Presence of burrowing mammals were associated with higher introduced ruderal forb cover (P = 0.044) and lower woody cover (P = 0.02) (Tables 4.8 and 4.9). No indicator species were detected for pastures with small mammal activity, pastures without burrows included strawberry (*Fragaria virginiana*), creamy peavine (*Lathyrus ochroleucus*), and prickly rose (*Rosa acicularis*) (Ps < 0.05; Appendix B.1.2.1).

Pastures identified as burned through the survey were significantly different from pastures lacking indication of fire (P = 0.003), with woody species more abundant in pastures exposed to fire (P = 0.019; Table 4.5). Burned pastures also had higher cover from native species, contributed by native perennial forbs, with more woody species, and reduced introduced ruderal forb cover (P < 0.05; Tables 4.6 to 4.9). Abundance of native species and the inclusion of shrubs corresponded with higher richness and diversity in pastures with reported fire (Ps < 0.025; Tables 4.10 and 4.11). Pastures that included charred woody debris within the top 15 cm of soil were not associated with different plant communities, but there were responsive functional groups and indicator species. Legume total foliar cover was higher in pastures with charcoal (P = 0.02; Table 4.8and 4.9), with the legumes creamy peavine, red clover (*Trifolium pratense*), and American vetch (*Vicia americana*), along with the forbs strawberry and prickly rose, all indicative of a history of burning (Table 4.4.).

NMDS of plant community foliar cover (Fig. 4.3; distance = Bray-Curtis, dimensions = 2, stress = 0.23) identified distinct gradients in plant communities. Increasing the NMDS to three dimensions resulted in a reduction in stress to 0.17, but given the complexity in data, a simpler 2-D solution was considered more desirable. Our observed stress level of 0.23 is considered adequate for a low-dimension ordination, although stressed (Legendre and Legendre 1998). Relationships between significant management centroids and indicator species were explored in Fig. 4.4 for cultivation, feeding animals, fertilizing, harrowing and manure spreading ($P < 0.05$) (Table B.1.1.1). Pastures identified as never cultivated correlated positively to MDS1, which included significant responses from native perennial and ruderal forbs, woody species, total graminoids, total native species, pasture species richness, and soil lichen cover. Pastures diverging from non-cultivated areas along MDS1 (i.e. with a history of cultivation) were associated with higher soil fertility (C, N, and OM) and characterized by higher cover of introduced species including quackgrass (*Elytrigia repens*) and dandelion (*Taraxacum officinale*). Tame pastures abundant in introduced, seeded, forage grass cover were associated with high litter cover, along MDS2 (Fig. 4.3). Theses pastures had the highest overall RHA scores, with RHA categories of forage cover, cover of tall productive forages, and woody plant density corresponding with higher individual scores. Introduced forbs and legumes were more abundant where bare soil exposure and high C:N ratios were detected. Clustering of Bray-Curtis distances between pasture communities identified 27 unique plant communities encountered during the survey (Fig. B.3.1). The numerous communities would be challenging to describe, thus a penultimate peak of 10 communities was chosen with a second highest silhouette width (Fig. B.3.2). An indicator species analysis identified species associated with the unique communities, most communities contained Kentucky bluegrass and smooth brome, but its rank in dominance and co-dominant species differed in each community (Table B.3.1).

4.5.2 Soil Properties and Microsite Characteristics

Overall, soil properties were not very sensitive to pasture management, responding to only a handful of management conditions (Table 4.12). Total carbon, nitrogen, and organic matter responded to manure spreading and harrowing ($Ps \leq 0.022$), with more nutrients available when either activity occurred (Table 4.13). Manure spreading also increased soil salinity (EC) ($P = 0.025$). Harrowed pastures were abundant in silt and clay ($Ps < 0.039$), with lower proportions of sand ($P = 0.032$) indicating loamier soils. A higher proportion of silt was indicative of herbicide use ($P = 0.025$). Pastures with an unknown cultivation history had the highest proportions of clay, while non-cultivated pastures had the lowest proportions of clay ($P = 0.029$; Table B.5.1 and B.5.2). It is important to note that significant differences in soil texture were likely not caused by the management action, but indicative of site conditions that facilitate or cause the management action. The carbon to nitrogen (C:N) ratio was affected by the presence of burrowing mammals and both indicators of fire (survey and charcoal in soil) ($Ps < 0.05$). When burrowing animals were present there was a lower C:N ratio. Where fire had occurred the C:N ratio was higher. Higher intensity grazing, as determined through the rangeland health assessment form, was linked with high soil salinity ($P = 0.038$). A similar trend for increasing soil C and N with increasing grazing intensity was apparent but remained marginally significant ($Ps < 0.063$). Soil compaction data were only available for a subset of pastures (N=46), and exhibited minor sensitivity to management conditions, with the lone exception of pastures managed by land owners having greater compaction, while rented pastures had less soil compaction ($P = 0.023$). Soil pH was non-responsive to the management factors assessed.

Ground cover variables responded strongly to management factors. Continuous grazing resulted in more bare soil ($P = 0.023$), while pastures that were abandoned had lower bare soil and a thicker litter layer ($Ps < 0.05$). Abandoned pastures had litter cover and depth similar to pastures used by livestock during the winter. Pastures that had been fertilized were characterized by lower basal plant cover (stems, shoots, crowns at ground level) and higher litter cover ($Ps < 0.05$). Pastures that received manure application were more abundant in manure cover ($P = 0.008$). Harrowing was associated with more bare soil, a thinner litter layer, and more manure cover ($Ps < 0.05$). When hay was provided in pasture, more manure was present on the soil surface ($P = 0.01$). Litter accumulation was lower in pastures with burrowing mammals ($Ps < 0.012$). Pastures with both reported fire and fire indicated by charred woody debris in the soil had lower manure cover ($Ps < 0.033$) and a thicker litter layer ($Ps < 0.009$).

4.5.3 Rangeland Health Response to Management

Total rangeland health scores are heavily influenced by forage productivity and responded to cultivation history, timing of grazing, and the interaction of timing and grazing system ($P \leq 0.05$; Table 4.16). Cultivated fields had the highest RHA scores while pastures with unknown cultivation history had lower scores, while no difference existed between those pastures cultivated and non-cultivated. When pastures were grazed year-round, RHA scores were significantly lower than other timings (and their interaction with grazing system) (Table 4.17). Pastures with a recent history of abandonment or winter grazing (i.e. as part of a rotational system) had higher RHA scores. Fertilized pastures had marginally higher RHA scores than unfertilized pastures ($P = 0.08$; Table 4.16).

PerMANOVA with both Euclidean and Bray-Curtis distance measures yielded comparable results (Table 4.18), and showed that rangeland health responded to grazing systems, timing of grazing, the herbivores grazed, recent fire (indicated during the interview) and grazing intensity (quantified during the RHA) ($P < 0.05$). Marginally significant responses were found for previous cultivation and fertilization ($P < 0.1$; Table 4.18).

The indicator analysis of inversed RHA scores identified which aspects of the health assessment were affected by management actions (Table 4.19). Declines in the score for total forage cover occurred in pastures that were never cultivated ($P = 0.001$), which we identified as containing numerous native unpalatable perennial forbs. Loss of tall productive forage cover was found in non-cultivated fields, abandoned pastures, and under high grazing pressure ($P < 0.049$). Reduced litter and hydraulic function occurred in pastures where grazing occurred during the growing season (i.e. was not indicative of pastures utilized during the dormant season or abandoned). Litter was also reduced in pastures with high grazing pressure. Erosion occurred in pastures that were grazed year-round ($P = 0.002$) and was largely indicative of a moderate-high grazing pressure (P = 0.044). Increases in anthropogenic bare soil occurred in pastures that were grazed year-round, where animals were fed hay, pastures were aerated, and pastures were grazed at high intensities. Cover of weedy species and disturbance induced species were not indicative of any particular management actions, though there was a trend for these to increase under high grazing pressure (P = 0.098). Noxious weed cover did not respond to management, but noxious weed density was significantly higher in pastures that were rented or grazed year-round (P = 0.034). There were also moderate increases in weed density when alternative livestock (i.e. sheep and alpaca) were present (P = 0.82). Higher woody cover was marginally responsive to many management conditions including pastures where animals were not fed hay (P = 0.090), burrowing mammals were absent (p = 0.089), and when charred woody debris was found in the soil (P = 0.091), woody cover was significantly higher where fire was reported during the interview (P = 0.001).

NMDS of raw RHA scores (distance = Euclidean, dimensions = 2, stress = 0.14) demonstrated significant relationships between sites and responses to environmental variables (Fig. 4.5). Management factors with significant centroids for the first two dimensions were fire (reported in survey), feeding hay on pasture, and grazing intensity based on vegetation assessments (P < 0.05, Table B.2.2.1). Reports of previous fire and lower grazing pressure each responded positively to NMDS 1 while higher grazing pressure responded negatively to NMDS 1. No significant relationships between health and soil properties (i.e. texture, C, N, etc.) were found. Some ground cover characteristics were significant, including basal vegetation cover (space occupied by shoots and stems) and bare soil exposure. Basal vegetation cover was higher with pastures that had high scores for litter (hydraulic function) and bare soil (i.e. had little anthropogenic bare ground). Interestingly, this vector also correlated with an abundance of noxious weed seeds (presented in more detail in Chapter 5), suggesting noxious weeds were occurring in pastures with abundant perennial forage cover and litter, but not necessarily in pastures with abundant bare soil. The vector for total RHA score corresponded with the vector for litter depth, plant community richness, woody cover, native graminoids cover and pastures that scored high in the noxious weed density and cover categories (meaning noxious weeds were likely rare or absent), as well as pastures that had abundant cover of desirable forages (i.e. scored well in the category of forage species shifts). Thus, abundance of noxious weeds and non-forage cover appear responsible for declines in rangeland health in our study area. While a lack of trees and shrubs in tame pastures should be associated with higher RHA scores, we observed the opposite trend, where lack of woody cover was indicative of higher grazing intensities, and instead was associated with ruderal graminoid and introduced ruderal forb cover.

4.6 Discussion

4.6.1 Management and Disturbance Legacies

Results of this study showed that both the vegetation composition and soil characteristics of northern temperate pastures were correlated to management factors of livestock producers in central Alberta, Canada, and led to the formation of up to 10 unique plant community types based on cluster analysis (described in Appendix B.3). In general, plant communities within these pastures were mostly dominated by cool-season forage grasses that are often seeded forages, however these species can also be voluntary and increase cover in response to prolonged disturbance and abundant moisture with species like smooth brome (*Bromus inermis*), Kentucky bluegrass (*Poa pratensis*), and creeping red fescue (*Festuca rubra*). These species were dominant in 6 of the 10 communities described. Other cool-season forage grasses that were common, but are known to decrease overtime with grazing pressure, included orchardgrass (*Dactylis glomerata*), common timothy (*Phleum pratense*), and meadow brome (*Bromus bieberstienii*) (Government of Alberta 2010; Government of Alberta 2013). Decreaser forage species like orchardgrass (7 pastures) and meadow brome (8 pastures) were dominant in distinct communities. As expected in these heavily grazed small pastures, overall these decreaser forage grasses were less common than the invasive increaser forages. Pastures located in the Central Parkland and boreal (Dry Mixedwood and Central Mixedwood) natural subregions did not differ in composition, which is not unexpected as forested areas in the boreal fringe have largely been managed the same as parkland areas (Donkor et al.

2002), including being seeded with similar forages and were susceptible to propagation of similar invasive grasses and ruderals over the ~150 year history of agricultural settlement in the region. Conversion of boreal forest into tame (seeded) grassland may also reflect the need for a greater agriculture footprint required to support the peri-urban area of Edmonton, AB and the added pressure of smaller farms (<160 acres) and acreages (< 80 acres) occupied by commuters and hobby farmers' grazing animals for enjoyment and a secondary form of income (Cialdella et al. 2009; Rowan 1994; Sayre 2004; Chapter 3). Pastures within the boreal natural region were likely deforested. Deforestation events within a comparable ecoregion in central Saskatchewan reported a loss of soil organic carbon of 30 Mg C/ha (Fitzsimmons et al. 2004). Thus, conversion of these areas to pasture resulted in the formation of simpler plant communities dominated by cool-season forages with less potential to store carbon.

Plant communities and the biophysical properties of cultivated vs uncultivated and tame vs modified-tame pastures were divergent for similar reasons. Modified-tame pastures were partly native and historically tended to be left uncultivated due to their biophysical nature; however, their condition as native grasslands was often altered by numerous introduced species, likely resulting from improper grazing management in the pasture's history (e.g. excessive stocking) and grazing-induced community change overtime (Smoliak 1974; Vujnovic et al. 2000; Willms et al. 1985). Nevertheless, these modified-tame and non-cultivated pastures retained some native grass cover and had higher species diversity and retention of native forbs than tame pastures. A cultivation event in a pasture's history was associated with the loss of many native species, particularly native grasses that are known to decrease with disturbance like plains rough fescue (*Festuca hallii*) and western porcupine grass (*Hesperostipa curtiseta*), which in the Central Parkland would have been the dominant species pre-European settlement (Coupland and Brayshaw 1953). Common timothy (*Phleum pratense*) emerged as an indicator of cultivation, while invasive Kentucky bluegrass (*Poa pratensis*) and smooth brome (*Bromus inermis*) had a neutral response to cultivation (and all other management factors) as they have naturalized across the region. Previously cultivated pastures had clay-rich soils, while uncultivated fields did not have sandier soils, modified-tame pastures we significantly sandier. Hence native grassland species are finding refuge in marginal areas, particularly where soils are coarse. This also indicates pastures that were previously cultivated and have reverted back to semi-native communities had non-arable soils where native species are more adapted to edaphic conditions. Soil carbon, nitrogen, and organic matter were all greater in tame pastures than modified-tame pastures. Tame pasture in the region is likely providing valuable ecosystem services in the forms of carbon storage and nutrient cycling and tends to provide greater ecological services than annually cropped land (Mapfumo et al. 2002). However, we cannot discount the possibility that most tame pastures were on land of greater production potential. In north central Alberta, retention of native grassland is rare, and in general areas that were never cultivated are described as non-arable due to landform and biophysical properties (i.e. rockiness, high salinity, coarse textured soil, etc.). Analysis of soil properties found modified grasslands had coarser textured soils abundant in sand, meaning these areas were likely less suitable for tame pasture land and annual cropping. Areas currently in tame pasture and cropland were likely fertile plains rough fescue grassland historically (Coupland and Brayshaw 1953; Gauthier and Wiken 2003; Vujnovic et al. 2000). This finding clarifies further the results from Alberta (Bork 2015; Hewins et al. 2018), where native grassland was described as containing a larger carbon pool than both neighboring cropland and tame pasture on the same ecosites. Additionally, tame pasture tends to be stocked with high animal densities, receive nutrient inputs (manure and fertilizer), and supplemental feed can be provided in simple single-pasture systems, and it is therefore possible that nutrient inputs from livestock under high land use intensity have altered soil nutrients (Baron et al. 2001; Baron et al. 2002).

In our survey most of the variance among plant communities across pastures was explained by cultivation history rather than grazing management (i.e., continuously vs rotationally grazed). There are several potential explanations for this observation. First, our management survey of livestock producers did not provide enough detailed questions to derive stocking rate for all pastures. Second, our estimated stocking rates showed both rotationally and continuously grazed pastures were susceptible to high levels of grazing but did not differ significantly (Chapter 3; Pyle et al. 2017). Although research has shown that continuously grazed pastures are at risk for degradation due to overuse (De Bruijn and Bork 2006, Teague et al. 2013), with the prevalence of smaller farms in central Alberta including numerous small hobby farms surrounding the city of Edmonton, it is likely that there were livestock managers using simple rotational systems that were overstocked. *Bromus-Poa* grasslands that dominate the region are more productive with moderate to high intensity defoliation events, however six weeks of rest is required for biomass recovery (Donkor et al. 2003). Our qualitative assessment of grazing intensity using the rangeland health assessment identified pastures grazed at higher intensities had greater soil salinity and were associated with ruderal and salt tolerant species, like stinkweed (*Thlaspi arvense*) and foxtail barley (*Hordeum jubatum*), respectively. Pastures that were not grazed or grazed at low and low-moderate intensities retained native perennial grass species, while native perennial forbs were not associated with the highest grazing intensity. Herbivore type did not affect plant communities or soils. Cattle and horses were the most common livestock in the region and are known to co-occur in pasture (Chapter 3). These herbivores, in addition to sheep, alpacas, and other livestock, differ in foraging behavior and digestive efficacy, and their presence could therefore be expected to result in divergent plant communities (Rook et al. 2004). However, the only trends that emerged were reduced total graminoid cover in pastures grazed by horses, potentially due to their close grazing habit resulting from incisors (Gordon 1989), and thin or reduced litter layer in pastures grazed by alternative livestock (sheep and alpaca), presumably again due to their tendency to crop plants at a very low height (Allden and McDWhittaker 1970).

Changes in soil properties under different management conditions have been described in many localized studies, with a greater focus in general on the effects of specific grazing management practices (Donkor et al. 2002; Dormaar et al. 1997; Henderson et al. 2004; Mapfumo et al. 1999; Mapfumo et al. 2000). In our survey differences in grazing systems and timing unexpectedly did not correspond with differences in soil properties, similar results were reported by Mapfumo et al. (2000). Once again, this is likely reflective of diverse stocking rates, herbivores, etc. associated with our pastures. Orthic Black

Chernozems in the Aspen-Boreal region are susceptible to trampling which causes compaction of the topsoil under high stocking densities, meaning rotational systems using higher stocking densities for shorter durations can still have the negative impacts on soil that would be anticipated under continuous grazing (Donkor et al. 2002). Dark Gray Luvisols, which formed under deciduous aspen forest, typically have higher clay content and soil organic matter which makes these soils susceptible to compaction as well (Donkor et al. 2002; Mapfumo et al. 1999). Overall Donkor et al. (2002) found that short duration rotational grazing (SDG) compared to continuous grazing did not improve soil properties associated with water infiltration in the Parkland-Boreal region, as suggested by promoters of SDG (Savory and Parsons 1980); similarly standing biomass and fallen litter did not improve (Donkor et al 2002). We found that producers practicing rotational grazing were utilizing higher stocking densities than continuously grazed pastures and also utilized high stocking rates (Chapter 3), which could have similarly negated improved pasture soils. Dormaar et al. (1997) reported that grazing can decrease total soil carbon and nitrogen in Alberta's Mixedgrass prairie and postulated that changes in plant community were regulated by microclimate (i.e. litter cover). In our survey, assessment of ground cover intended to measure microenvironment responses to management, was responsive to both the type of grazing system and timing of herbivory. Presence of grazing during the growing season resulted in a thinner litter layer with less cover, and bare ground was twice as high under continuous grazing compared to rotationally grazed areas. These differences under long-term management have the potential to alter nutrient cycling, ecological function, and overall health of the plant community.

Fertilizer is a tool available to rejuvenate pastureland, but one that can shift the competitiveness of legumes (Lardner et al. 2000; Lardner et al. 2001; Lardner et al. 2002; Malhi et al. 2000). While rejuvenated pasture was not associated with significantly different plant community composition, the dynamics of plant functional groups were responsive. Treated pastures were associated with lower broad leaf cover, especially from legumes. Ruderal grasses and introduced ruderal forbs were also lower under fertilization. The lack of legumes in the community suggests either producers may be responding to the lack of legume cover, corresponding to pastures lacking productivity or long-term fertilizer application could also reduce the competitiveness of legumes (Aydin and Uzun 2005) and forbs (Schellberg et al. 2001) if the pasture is released from nitrogen deficiency, as more competitive grasses exploit the available nitrogen (Aydin and Uzun 2005; Schellberg et al. 2001). Fertilized pastures had high litter cover and lower basal vegetation cover. High litter cover likely supressed potential niche space for ruderal grasses and weedy introduced annuals and biennials (Facelli and Pickett 1991). Fertilization of pasture in the Parkland has been shown to improve biomass of desirable forages and supress Canada thistle (*Cirsium arvense*) when combined with mowing (Grekul et al. 2007). Schellberg et al. (2001) found lower species richness in fertilized pastures due to the elimination of forbs; this corresponds to our finding of lower floristic richness and diversity in fertilized pastures. In a three year trial, Aydin and Uzun (2005) found that the proportion of legumes [dry weight] decreased from 47.0% to 5.3%, which is comparable to the 3.1% cover found in our pastures, suggesting significant legume loss can occur with short-term fertilization. Aydin and Uzun (2005) found that legume biomass only increased with phosphorus addition, and this suggests that applying additional phosphorus could compensate for crude protein losses due to legume reductions with nitrogen fertilization. Fertilized pastures also had soil nitrogen levels comparable to non-fertilized pastures, which contradicts results found by long-term fertilization studies (Schellberg et al. 2001). We are limited in our ability to explain fertilization history because our survey did not ask producers what their motivation was for treating pastures, the frequency of fertilization, when the last fertilization occurred, or sample actual biomass to measure productivity or forage sward quality. Yield benefits from fertilization, in addition to mechanical methods, can also be relatively short lived (Lardner et al. 2000), necessitating retreatment to maintain them.

Pastures that were manured and harrowed tended to have limited cover contributed from native species, and greater cover from introduced species. Manuring has the potential to introduce agronomic weeds through the introduction of endozoochorous seeds (Malo and Suarez 1995) and through ruderal species that establish on stockpiled manure (Menalled et al. 2005). However, there were few weedy indicator species associated with these management factors, but the presence of noxious weed, white cockle (*Silene latifolia* ssp. *alba*), may be of management concern (McNeill 1977). Although manure tends to be dense with ruderal seeds (Dastgheib 1989; Pleasant and Schlather 1994), some studies report it is not an important weed source (Menalled et al. 2005; Pleasant and Schlather 1994) unless animals are consuming feed with noxious weed species (Pleasant and Schlather 1994). Like fertilization, nutritive additions in manure could be improving the competitiveness of forage grasses (Blonski et al. 2004), intercepting light and soil resources supressing ruderal forb seedlings from establishing from a weed seed bank (López-Mariño et al. 2000). Spreading manure and harrowing had profound effects on parkland pasture soils, increasing soil carbon, nitrogen, organic matter and salinity. Replicable results have been found in other studies, where organic material in manure improves soil organic matter and can increase the carbon pool by an average of 9% in the top 10 to 20 cm of soil (Conant et al. 2001). Single applications of manure can improve available soil nitrogen and improve plant growth; however, their results are short lived as available nitrogen can be utilized within a single growing season (Bork et al. 2013). Repeated manure addition can increase prairie soil salinity, and this is exacerbated under non-irrigated conditions (i.e. typical pasture and feedlots) (Hao and Chang 2003). Harrowing is often paired with manure spreading to physically spread manure (Chapter 3), likely explaining many of the comparable responses, however there were some variables that corresponded to only harrowing. For example, harrowing was linked to reduced litter depths and higher bare soil cover likely resulting from the mechanical perturbation and movement of fallen litter (Mills and Sina Adl 2006), which in-turn could have improved the litter's surface area and increased decomposition due to improved photodegradation (Barnes et al. 2012; Rozema et al. 1997) and microbial activity (Beare et al. 1992). Alternatively, the relatively thinner litter layer could be indicative of litter compaction due to intensive land use.

Supplemental animal feeding with hay, primarily bale grazing, was associated with changes in plant communities and biophysical properties. Where animals were fed hay, there was a reduction in total plant species richness and diversity corresponding with reductions in total forb cover and native cover.

Ruderal mustards and *Chenopodium* spp. were associated with feeding, presumably as a result of propagule distribution in hay or higher grazing pressure, while orchardgrass (*Dactylis glomerata*) which is known to decrease under grazing pressure was associated with pastures where no feeding occurred. These changes were likely induced by higher herbivore pressure as these pastures were associated with higher stocking rates (Chapter 3) and higher basal manure cover (a proxy for higher cattle use). Higher stocking rates in pastures where animals are fed can result in higher soil bulk density [compaction] from trampling (Stephenson and Veigel 1987), however we did not find this effect. Winter feeding on pasture does have benefits, such as improved soil nitrogen and phosphorus, and can be a more effective pasture rejuvenation method than manure spreading (Jungnitsch et al. 2011), however these improvements were not realized in this study. It is possible that the presence of ruderal weeds is induced by the higher stocking rate and *in-situ* spreading of manure, or they could be introduced from the supplemental feed. Strewing hay across fields has been demonstrated as a method for spreading seed and can be used to improve grassland botanical diversity (Edwards et al. 2007) and act as a vector for spreading invasive or introduced species (Dutt et al. 1982). It is also important to note that large disturbances in pastures would have been avoided choosing a uniform vegetated area. Studies reporting the impacts of cattle on feeding areas are typically observing the immediate area and travel paths which typically have markedly altered vegetation and soils (Simek et al. 2005).

Soil aeration of pastures was uncommon in our study region despite potential benefits such as reduced soil compaction (Cournane et al. 2011), which was not realized in the current study. However, our ability to test this was limited by a small proportion of pastures experiencing aeration and smaller subset of total pastures tested (N = 45/102) in the second year. Similar findings (i.e. ineffective aeration treatments) were reported by Malhi et al. (2000) who also examined central Alberta soils. Aeration in combination with fertilization has been reported to improve Parkland pasture productivity more that fertilizer alone, while aeration by itself did not improve productivity (Lardner et al. 2000). Aeration treatments come at the cost of increased presence of annual weeds, greater soil exposure, and decreased forage production the same year as the treatment potentially breaking weed seed dormancy (Lardner et al. 2001). Similarly, we found greater cover of ruderal grass species in aerated pastures. Mowing and swathing were also uncommon as the pastures surveyed here were typically used for grazing rather than hay fields; pastures surveyed here were either abandoned (i.e. not grazed) for some time or stocked during the dormant season and hayed at least once during the growing season. Due to the small sample size results should be interpreted with caution.

Herbicide use was not associated with distinct plant communities but did affect the cover of multiple plant functional groups. Pastures treated with herbicide within the last 3 years had high cover from introduced species (> 90%), primarily from forage grasses; where seeded tame forage grasses creeping red fescue (*Festuca rubra*) and meadow fescue (*Schedonorus pratensis*) emerged as strong indicators. This suggests herbicides released these grasses from weedy competition (Grekul and Bork 2007), or producers with these pastures were more likely to use herbicides in an attempt to maximize grazing capacity (Bowes 1981). Herbicide use was related to the near elimination of native ruderal and native perennial forbs, while introduced ruderal forbs and legumes were unaffected. Noxious weeds were marginally reduced under herbicide treatment, perhaps a legacy of their resiliency and difficulty to control without repeated and integrated techniques. Our survey of producers identified that target species were diverse and not necessarily noxious weeds (e.g. dandelion (*Taraxacum officinale*) or buckbrush (*Symphoricarpos* spp.)), alternatively there were abundant noxious weeds that were infrequently targeted despite their presence (e.g. common tansy (*Tanacetum vulgare*) and white cockle (*Silene latifolia* ssp. *alba*)) (Chapter 3). Despite this, Canada thistle (*Cirsium arvense*) was a strong indicator of herbicide affected pastures and was the primary target weed (Chapter 3). In the Parkland, Canada thistle has naturalized (Moore 1975) and significantly reduces palatable forage through anti-herbivory mechanisms (epidermal spines) (Grekul and Bork 2004); however, this weed can be effectively controlled with herbicide (Grekul and Bork 2005) and integrated weed management (De Bruijn and Bork 2006; Grekul and Bork 2007). Improved cover from graminoids results in a more palatable and productive forage sward and the retention of legumes with herbicides, although unexpected, is important for maintaining pasture productivity. In small scale studies, broadleaf herbicides typically reduce the cover and biomass of legumes (Bowes 1981; Bowes 1982; Grekul and Bork 2007; Miller et al. 2015), grasses can compensate (Bowes 1981), but net reductions in net biomass may still occur (Miller et al. 2015). Legume seedling emergence and establishment can be affected 15 to 24 months after treatment in Parkland soils (Miller 2013). Substantial loss of legumes may inevitably result in producers renovating pastures by overseeding legumes. On larger scales, producers are more likely to target problematic areas for treatment to save time and money. In our survey we did not specifically ask how producers applied herbicide products, but based on our meta-data, we found some producers indicated that they had spot and/or broadcast sprayed affected areas. These two methods could significantly alter plant communities; with pastures spot treated theoretically retaining more legume cover and diversity.

Burrowing mammals are common pasture residents, often perceived as pests because they create bare ground within the vicinity of their burrows which opens up the community to undesirable species and reduces forage yield (Carlson and Crist 1999; Entz et al. 1995). Colonies of burrowing mammals like Richardson's ground squirrels were once an important component of grassland ecosystems, creating habitat for a variety of other small mammals and providing a food source to meso-fauna (Bylo et al. 2014). Pastures containing burrowing rodents did not have significantly different plant communities, though soil conditions did vary. A lower C:N ratio and thinner litter layer could indicate biophysical conditions preferred or exploited by burrowing pests, as burrowing mammals have been found to utilize areas overgrazed by livestock (Bylo et al. 2014) or the former could arise from the repeated disturbance imposed by colonies of burrowing mammals (Agnew et al. 1986; Whicker and Detling 1988). Soil is often exposed near burrows, but their grazing can affect the entire area of a colony as they forage aboveground and clip vegetation to help with predator detection (Whicker and Detling 1988). In Mixedgrass prairie, Bylo et al. (2014) found ground squirrel populations were greater in well-drained uplands where grazing intensity was higher and plant biomass was reduced; while the less frequently reported badger preferred lowland where grazing pressure was lower, resulting in greater litter cover and plant community biomass.

Pastures identified as burned through the producer interview were dissimilar in plant community composition from pastures that were not burned in recent memory, while pastures containing charred woody debris in the topsoil were similar in composition; this suggest that more recent fire events had residual effects on the plant community that were at the time of sampling. Burned pastures had greater cover from native plants, attributed to greater cover from woody plants and native perennial forbs while introduced forbs contributed less cover. Greater diversity from native broad leaf species corresponded with greater species richness and diversity. The soil had an elevated carbon to nitrogen ratio and a relatively thicker litter layer. These recent fires included both prescribed and wildfires, with most instances of prescribed fire occurring on public land (two instances on private land) within the confines of a grazing reserve. Unfortunately, the dates of most fires were not reported, and we were not able to observe community changes over time. Historically, fire was credited with the maintenance of fescue grasslands in the mesic Central Parkland region (Archibold et al. 2003) with a fire return interval of every 10-15 years (Wright and Bailey 1982), as these communities are susceptible to encroachment by woody species like aspen (*Populus tremuloides*) and western snowberry (*Symphoricarpos occidentalis*) (Bailey et al. 1990). These results suggest fire in the Parkland improves the functional diversity of native plants and may be associated with more structural diversity (shrubs) and community heterogeneity than typical cultivated and seeded tame pastures. Bailey and Anderson (1978) reported *Festuca-Hesperostipa* grassland communities expressed dominance of perennial forbs up to three years post fire, similarly Ren and Bai (2017) attributed this to forb emergence from the seed bank post-fire. Perhaps improvements in native perennial forb cover are a legacy effect of fire. Reductions in introduced ruderal forbs suggest fire aided in the reduction of agronomic weed species from the seed bed (Ditomaso et al 2006). There is limited research reporting the effects of fire on Parkland soils (Anderson and Bailey 1980), however, an elevated carbon to nitrogen ratio is indicative of nitrogen limitation. In productive tallgrass prairie, an elevated carbon to nitrogen ratio can occur under frequent fire (Ojima et al. 1994), however the fire events that occurred were likely singular and stochastic. Pastures that contained charred woody debris within the topsoil were assumed to have burned at point in their history and were associated with some important community responses. Mainly, legume cover was higher in these pastures and this was associated with native species with a climbing (vine) growth habit like cream peavine (*Lathyrus ochroleucus*) and American vetch (*Vicia americana*), and red clover (*Trifolium pretense*), which is introduced and stoloniferous. Limited evidence from here in Alberta indicates that legumes often demonstrate marked increases following fire (Bork et al. 2002), suggesting that persistent legume seeds in the seed bank may be released by fire itself, the post-fire environmental conditions, or a combination of the two. Similar to pastures identified as burned in the survey, the carbon to nitrogen ratio was also greater but the difference was reduced. However, it is important to note that charred organic matter was visually identified during sieving and preparation for other analyses of soil properties. Using other techniques to identify char such as electron microscopy and UV-oxidation, we would have likely found more soil from pastures with Black Chernozems possessing charred organic matter as it is present in particles as fine as silt, making up a significant proportion of Chernozems (Ponomarenko and Anderson 2001).

4.6.2 Rangeland Health and Management

Higher rangeland health scores were associated with greater total cover from graminoids (primarily seeded forage grasses), reduced cover from introduced ruderal forbs, and decreased plant species richness. This comes from the tame pasture RHA form placing emphasis on the importance of forage species (grasses and legumes); healthy pastures likely had less floristic diversity because forbs where outcompeted by forage grasses or they were reduced under efforts to renovate pastures through inputs like fertilizer and herbicide (Grekul and Bork 2007). Rangeland health responded to many aspects of management, where healthier pastures were associated with cultivation and dormant season use while pastures that were stocked year-round had the lowest RHA scores. There was a tendency for scores to be lower in non-cultivated pastures because the retained native perennial forb diversity would have contributed negatively to the overall score because they do not contribute palatable forage (i.e. lower score for total forage cover and tall productive forage cover). Loss of tall productive forage cover also occurred in pastures that were abandoned [not grazed] and holistically described as heavily grazed. Reduced litter and hydraulic function occurred in pastures where grazing occurred during the growing season and this was exacerbated under higher grazing pressures. Pastures that were stocked year-round were associated with lower scores of the rangeland health assessment's questions observing bare soil, erosion, noxious weed density, and hydraulic function [litter abundance]. Erosion features such as hoof shear, wallowing, and trailing were more prevalent when pastures were grazed year-round, which likely resulted in greater bare soil. However, these signs of pasture degradation were not detected in our analysis of ground cover variables. It appears that the rangeland health assessment was more sensitive to deleterious management conditions than a suite of other variables used. During the tame pasture health assessment, the observer is encouraged to look beyond the transect and walk around the pasture and observe the plant community, identify patches of invasive noxious weeds, identify areas of woody encroachment, and check for areas of overuse or disturbance (i.e. feeding stations, trails, loading areas, etc.) (Government of Alberta 2010). During our soil and vegetation sampling we avoided areas of high disturbance and selected areas with uniform plant communities.

Higher woody cover was responsive to many management conditions including pastures where animals were fed hay, where burrowing mammals were absent, recent fire (reported during the interview), and when charred woody debris was found in the soil. Woody species were likely reduced in pastures where animals were provided supplemental feed due to grazing pressure, and likely reduced in pastures with animal burrows for similar reasons as burrowing mammals tend to select sites with higher grazing pressure (Bylo et al. 2014). In areas with a history of fire, higher woody cover is likely indicative of areas that were previously woodlands and there has been regrowth since the initial disturbance (Bailey et al. 1990). This was known to be the case for the Blackfoot Grazing Reserve that used prescribed fire to improve grazing capacity, and there were a handful of pasture where wildfires had occurred – these pastures were usually located within woodland openings.

Permutational ANOVA found that RHA scores from all categories in concert are sensitive to differences in grazing management (i.e. grazing systems, timing of grazing, the types of livestock grazing, and grazing intensity) and a history of fire reported in the survey. Ordination of RHA scores showed cover of shoots/stems was higher where there was abundant litter and less bare soil which corresponded with an abundance of noxious weed seeds in the soil (presented in more detail in Chapter 5), suggesting noxious weeds were occurring in pastures with abundant perennial forage cover and litter, but not necessarily in pastures with abundant bare soil. The vector for total RHA score corresponded with the vector for litter depth, plant community richness, woody cover, native graminoids cover and pastures that scored high in the noxious weed density and cover categories (meaning noxious weeds were likely rare or absent), as well as pastures that had abundant cover of desirable forages (i.e. scored well in the category of forage species shifts). Thus, abundance of noxious weeds and non-forage cover appear responsible for declines in rangeland health in our study area. While a lack of trees and shrubs in tame pastures should be associated with higher RHA scores, we observed the opposite trend, where lack of woody cover was indicative of higher grazing intensities and instead was associated with ruderal graminoid and introduced ruderal forb cover. One trend that emerged from the analysis of RHA scores was that higher scores in the categories observing woody species cover and density (indicating less encroachment) were often associated with deleterious management conditions and did not correspond with higher overall rangeland health scores.

4.6.3 Reflection on Methods

It should be noted that our treatments were reflective of management conditions naturally occurring on small, mixed and larger farms, and we had no control over the intensity of the management actions taken. In controlled experiments observing similar factors, more significant and concise effects may be found when other confounding variables are controlled. This is particularly evident in the diversity of stocking rates, grazing systems, and herbicide products used across pastures sampled in this investigation (Chapter 3). As controlled, manipulative studies do not provide insight into local management concerns, nor do they define how producers are affecting their forage resources and soil on a broad scale, the survey conducted here provides novel insight into management actions on pasture, vegetation and soil responses.

During our survey riparian and forested areas within pastures were often avoided, seeking uniform grassland areas that could fit a 260 m transect. Riparian and forested areas in the Parkland-Boreal area are sensitive to cattle disturbance and could have provided valuable insight into overall pasture health if these ecosites and plant communities were assessed separately (Fitch and Adams 1998; Fitch et al. 2003; Miller et al. 2010). Qualitative observations and photos during surveys show evidence of heavy animal loading in patches of tall woody vegetation with compromised mid-structural layers (shrubs). Riparian areas were avoided, but severe hummocking of moist soil adjacent to these areas was often observed. Grazing-tolerant introduced grasses like *Poa* and *Bromus* common in tame grassland are likely less sensitive to the excessive stocking rates observed in the region than native vegetation surrounding wetlands and forest understories, and likely colonized these areas if they were exposed to high grazing pressure.

Based on the current format of the RHA, it is assumed that tame pastures or modified-tame pastures with less than 5% woody cover would be healthier. Our data did not support this idea as there was a trend for health to decrease when woody cover was low or non-existant, resulting from heavy utilization. The current rangeland health assessment also assumes a positive linear relationship between litter abundance and rangeland health. However, there are accounts in literature where too much litter can reduce plant community productivity (Deutsch 2010b; Hilger and Lamb 2017).

Further, it would have been beneficial to link plant communities and soil responses to management with a better understanding of the sociology (motivation, philosophy, economics, etc.) behind management actions. For example, our RHA assumed that remnant native grasslands or uncultivated areas were similarly managed as tame pasture due to the prominence of introduced grasses in the region. However, there were occasional cases where landowners recognised greater sensitivity of these areas, but the design of the human ethics approved survey was not detailed enough to capture these landowner values and these conversations remain only as memories.

4.7 Conclusions and Management Implications

Various pasture management practices have been extensively studied for impacts on vegetation, soils, and other environmental impacts, but typically using tightly controlled manipulative studies. Within the small peri-urban pastures in north central Alberta, the primary influence on communities was the historic use of tillage and seeding rather than current grazing systems. Despite chronic overgrazing, plant communities within these pastures were in relatively good health (dominated by productive forages) based on the criteria for tame grasslands. In conversation with growers an understanding of the cultural aspects influencing management decisions became apparent but were not always captured in survey questions. These include the relative importance of these small pastures to the economic livelihood of the managers.

4.8 Literature Cited

Agnew, W., Uresk, D. W., and Hansen, R. M. 1986. Flora and fauna associated with prairie dog colonies and adjacent ungrazed mixed-grass prairie in western South Dakota. Journal of Range Management **39**(2):135-139.

Allden, W. G. and McDWhittaker, I. A., 1970. The determinants of herbage intake by grazing sheep: the interrelationship of factors influencing herbage intake and availability. Crop and Pasture Science **21**(5):755-766.

Anderson, M. J. 2005. Permutational multivariate analysis of variance. Department of Statistics, University of Auckland **26**:32-46.

Anderson, H. G. and Bailey, A. W. 1980. Effects of annual burning on grassland in the aspen parkland of east-central Alberta. Canadian Journal of Botany **58**(8):985-996.

Aydin, I., and Uzun, F.. 2005. Nitrogen and Phosphorus fertilization of rangelands affects yield, forage quality and the botanical composition. European Journal of Agronomy **23**(1):8-14.

Bailey, A. W. and Anderson, M. L. 1978. Prescribed burning of a *Festuca-Stipa* grassland. Journal of Range Management **31**(6):446-449.

Bailey, A. W., Irving, B. D. and Fitzgerald, R. D. 1990. Regeneration of woody species following burning and grazing in aspen parkland. Journal of range management **43**(3):212-215.

Bailey, A.W., McCartney, D. and Schellenberg, M.P. 2010. Management of prairie rangeland. Agriculture and Agri-Food Canada.

Barnes, P. W., Throop, H. L., Hewins, D. B., Abbene, M. L. and Archer, S. R. 2012. Soil coverage reduces photodegradation and promotes the development of soil-microbial films on dryland leaf litter. Ecosystems **15**(2): 311-321.

Baron, V.S., Dick, A.C., Mapfumo, E., Malhi, S.S., Naeth, M.A., and Chanasyk, D.S. 2001. Grazing impacts on soil nitrogen and phosphorus under parkland pastures. Journal of Range Management **54**(6):704-710.

Baron, V. S., Mapfumo, E., Dick, A. C., Naeth, M. A., Okine, E. K. and Chanasyk, D. S. 2002. Grazing intensity impacts on pasture carbon and nitrogen flow. Journal of Range Management **55**(6):535-541.

Beare, M. H., Parmelee, R. W., Hendrix, P. F., Cheng, W., Coleman, D. C. and Crossley, D. A. 1992. Microbial and faunal interactions and effects on litter nitrogen and decomposition in agroecosystems. Ecological Monographs **62**(4):569-591.

Blonski, L.J., Bork, E.W., Blenis, P.V. 2004. Herbage yield and crude protein concentration of rangeland and pasture following hog manure application in southeastern Alberta. Canadian Journal of Plant Science **84**(3):773-783.

Borcard, D., Gillet, F., and Legendre, P. 2011. Numerical ecology with R. Springer Science and Business Media, New York, USA.

Bork, E.W. 2015. Baseline quantification of carbon stored in Alberta rangelands. Final report to the Alberta Livestock and Meat Agency Ltd, Research and Development Program.

Bork, E. W., Adams, B. W., and Willms, W. D. 2002. Resilience of foothills rough fescue, Festuca campestris, rangeland to wildfire. Canadian Field-Naturalist **116**(1):51-59.

Bork, E.W., and Blonski, L.J. 2012. Short-term native grassland compositional responses following liquid hog manure application. Canadian Journal of Plant Science **92**(1):55-65.

Bork, E. W., Lambert, B. D., Banerjee, S. and Blonski, L. J. 2013. Soil mineral nitrogen responses following liquid hog manure application to semiarid forage lands. Canadian Journal of Soil Science **93**(3):369-378.

Bouyoucos, G.J. 1927. The hydrometer method as a new method for the mechanical analysis of soils. Soil Science **23**(5):343-354.

Bowes, G. G. 1981. Improving aspen poplar and prickly rose-covered rangeland with herbicides and fertilizer. Canadian Journal of Plant Science **61**(2):401-405.

Bowes, G. G. 1982. Changes in the yield of forage following the use of herbicides to control aspen poplar. Journal of Range Management **35**(2):246-248.

Bowes, G.G., and Spurr, D.T. 1996. Control of aspen, balsam poplar, prickly rose and western snowberry with metsulfuron-methyl and 2, 4-D. Canadian Journal of Plant Science **76**(4):885-889.

Briske, D.D. Fuhlendorf, S.D., and Smeins, F.E. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management **58**:1-10.

Bylo, L. N., Koper, N., and Molloy, K. A. 2014. Grazing intensity influences ground squirrel and American badger habitat use in mixed-grass prairies. Rangeland Ecology and Management **67**(3):247-254.

Carlson, J.M., and Crist, T.O. 1999. Plant responses to pocket-gopher disturbances and topography. Journal of Range Management **52**(6):637-645.

Cialdella, N., Dobremez, L., and S. Madelrieux. 2009. Livestock farming systems in urban mountain regions: Differentiated paths to remain in time. Outlook on Agriculture **38**(2):127-135.

Conant, R.T., Paustian, K. and Elliott, E.T. 2001. Grassland management and conversion into grassland: effects on soil carbon. Ecological Applications **11**(2):343-355.

Coupland, R.T., and Brayshaw, T.C.1953. The fescue grassland in Saskatchewan. Ecology **34**(2):386-405.

Cournane, F.C., McDowell, R.W., Littlejohn, R.P, Houlbrooke, D.J, and Condron, L.M. 2011. Is mechanical soil aeration a strategy to alleviate soil compaction and decrease phosphorus and suspended sediment losses from irrigated and rain-fed cattle-grazed pastures? Soil Use and Management **27**(3):376-384.

Dastgheib, F. 1989. Relative importance of crop seed, manure and irrigation water as sources of weed infestation. Weed Research **29**(2):113-116.

De Bruijn, S.L., and Bork, E.W. 2006. Biological control of Canada thistle in temperate pastures using high density rotational cattle grazing. Biological Control **36**(3):305-315.

De Bruijn, S.L., Bork, E.W. and Grekul, C.W. 2010. Neighbor defoliation regulates Canada thistle (*Cirsium arvense*) in pasture by mediating interspecific competition. Crop Protection **29**(12):1489-1495.

De Caceres, M., Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. Ecology, URL http://sites.google.com/site/miqueldecaceres/

De Keyser, E.S., Dennhardt, L.A., and Hendrickson, J. 2015. Kentucky bluegrass (*Poa pratensis*) invasion in the Northern Great Plains: a story of rapid dominance in an endangered ecosystem. Weed Science **64**(3):409-420.

De Mendiburu, F. 2017. agricolae: Statistical Procedures for Agricultural Research. R package version 1.2-6. https://CRAN.R-project.org/package=agricolae

Desserud, P.A. and Naeth, M.A. 2014. Predicting grassland recovery with a state and transition model in a natural area, Central Alberta, Canada. Natural Areas Journal **34**(4):429-442.

Deutsch, E. S., Bork, E. W. and Willms, W. D. 2010a. Separation of grassland litter and ecosite influences on seasonal soil moisture and plant growth dynamics. Plant Ecology **209**(1):135-145.

Deutsch, E. S., Bork, E. W. and Willms, W. D., 2010b. Soil moisture and plant growth responses to litter and defoliation impacts in Parkland grasslands. Agriculture, Ecosystems & Environment **135**(1):1-9.

Donkor, N.T., Gedir, J.V., Hudson, R.J., Bork, E.W., Chanasyk, D.S., and Naeth, M.A. 2002. Impacts of grazing systems on soil compaction and pasture production in Alberta. Canadian Journal of Plant Science **82**(1):1-8.

Dormaar, J.F., Adams, B.W., and Willms, W.D. 1997. Impacts of rotational grazing on mixed prairie soils and vegetation. Journal of Range Management **50**(6):647-651.

Dutt, T. E., Harvey, R. G. and Fawcett, R. S. 1982. Feed quality of hay containing perennial broadleaf weeds. Agronomy Journal **74**(4):673-676.

Dyksterhuis, E.J. 1949. Condition and management of rangeland based on quantitative ecology. Journal of Range Management **2**(3):104-115.

Edwards, A. R., Mortimer, S. R., Lawson, C. S., Westbury, D. B., Harris, S. J., Woodcock, B. A. and Brown, V. K. 2007. Hay strewing, brush harvesting of seed and soil disturbance as tools for the enhancement of botanical diversity in grasslands. Biological Conservation **134**(3):372-382.

Elsinger, M.E. 2009. Reclamation status of plains rough fescue grasslands at Rumsey Block after well site and pipeline disturbance. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Entz, M.H., Bullied, W.J., and Katepa-Mupondwa, F. 1995. Rotational benefits of forage crops in Canadian prairie cropping systems. Journal of Production Agriculture **8**(4):521-529.

Facelli, J.M., and Pickett, S.T.A. 1991. Plant litter: its dynamics and effects on plant community structure. Botanical Review **57**:2-32.

Fitch, L., and Adams, B.W. 1998. Can cows and fish co-exist? Canadian Journal of Plant Science **78**(2):191-198.

Fitch, L., Adams, B.W., O'Shaughnessy, K. 2003. Riparian areas and grazing management. Cows and Fish Program.

Fitzsimmons, M. J., Pennock, D. J. and Thorpe, J. 2004. Effects of deforestation on ecosystem carbon densities in central Saskatchewan, Canada. Forest Ecology and management **188**(1):349-361.

Fuhlendorf, S. D., Engle, D. M., Elmore, R. D., Limb, R. F., and Bidwell, T. G. 2012. Conservation of pattern and process: Developing an alternative paradigm of rangeland management. Rangeland Ecology and Management **65**(6):579-589.

Gauthier, D.A. and Wiken, E. B. 2003. Monitoring the conservation of grassland habitats, prairie ecozone, Canada. Environmental Monitoring and Assessment **88**(1):343-364.

Gifford, M. and Otfinowski, R. 2013. Landscape disturbances impact affect the distribution of exotic grasses in northern fescue prairies. Invasive Plant Science and Management **6**(4):577-584.

Glass, G.V., Peckham, P.D. and Sanders, J.R. 1972. Consequences of failure to meet assumptions underlying the fixed effects analyses of variance and covariance. Review of Educational Research **42**(3): 237-288.

Gordon, I.J. 1989. Vegetation community selection by ungulates on the Isle of Rhum. II. Vegetation community selection. Journal of Applied Ecology **26**(1):53-64.

Gossner, M. M., Lewinsohn, T. M., Kahl, T., Grassein, F., Boch, S., Prati, …Allan, E. 2016. Land-use intensification causes multitrophic homogenization of grassland communities. Nature **540**(7632)266-269.

Government of Alberta. 2010. Rangeland health assessment for grassland, forest & tame pasture. Alberta Government.

Government of Alberta. 2013. Range plant communities and range health assessment guidelines for the Central Parkland subregion of Alberta. Alberta Government, Red Deer, AB.

Grekul, C.W. and Bork, E.W. 2004. Herbage yield losses in perennial pasture due to Canada thistle (*Cirsium arvense*). Weed Technology **18**(3):784-794.

Grekul, C. W. and Bork, E.W. 2007. Fertilization augments Canada thistle (*Cirsium arvense* L. Scop.) control in temperate pastures with herbicides. Crop Protection **26**(4):668-676.

Grekul, C.W., Cole, D.E. and Bork, E.W. 2005. Canada thistle (*Cirsium arvense*) and pasture forage responses to wiping with various herbicides. Weed Technology **19**(2):298-306.

Grime, J.P. 1979. Plant strategies and vegetation processes. John Wiley & Sons, New York, USA.

Gross, J. and Ligges, U. 2015. nortest: Tests for normality. R package version 1.0-4. https://CRAN.R-project.org/package=nortest

Hao, X. and Chang, C. 2003. Does long-term heavy cattle manure application increase salinity of a clay loam soil in semi-arid southern Alberta? Agriculture, Ecosystems & Environment **94**(1):89-103.

Henderson, D.C., Ellert, B.H., and Naeth, M.A. 2004. Grazing and soil carbon along a gradient of Alberta rangelands. Journal of Range Management **57**(4):402-410.

Hewins, D.B., Lyseng, M.P., Schoderbek, D.F., Alexander, M., Willms, W.D., Carlyle, C.N., Chang, S.X. and Bork, E.W. 2018. Grazing and climate effects on soil organic carbon concentration and particle-size association in northern grasslands. Scientific Reports **8**(1):1336.

Hilger, H. and Lamb, E.G. 2017. Quantifying optimal rates of litter retention to maximize annual net primary productivity on mixed-grass prairie. Rangeland Ecology and Management **70**(2):219-224.

Josephson, R.M. 1993. Economics of agricultural encroachment on wildlife habitat in southwest Manitoba. Canadian Journal of Agricultural Economics **41**(4):429-435.

Jungnitsch, P. F., Schoenau, J. J., Lardner, H. A. and Jefferson, P. G. 2011. Winter feeding beef cattle on the western Canadian prairies: Impacts on soil nitrogen and phosphorus cycling and forage growth. Agriculture, Ecosystems & Environment **141**(1):143-152.

Kjorlien, M.E. 1977. A review of historical information of fire history and vegetation description of Elk Island and the Beaver Hills. Parks Canada, Elk Island National Park, Fort Saskatchewan, Alberta.

Laird, A.S. 2014. Residual effect of herbicides used in pastures on clover establishment and productivity. M. Sc. Thesis, Louisiana State University and Agricultural and Mechanical College, Department of Plant, Environmental and Soil Science, Baton Rouge, Louisiana, USA.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2000. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on dry matter yield and forage quality. Canadian Journal of Plant Science **80**(4): 781-791.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2001. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on botanical composition. Canadian Journal of Plant Science **81**(4):673-683.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2002. Rejuvenation affects nutritive value of long-established tame forages. Canadian Journal of Animal Science **82**(4):621-626.

Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangelands: a viewpoint. Journal of Range Management **44**(5):427-433.

Lavkulich, L.M., and Wiens, J.H. 1970. Division S-3—Soil microbiology and biochemistry: comparison of organic matter destruction by hydrogen peroxide and sodium hypochlorite and its effects on selected mineral constituents. Soil Science Society of America Proceedings **34**:755-758.

Legendre, P. and Legendre, L. 1998. Numerical Ecology. 2nd ed. Elsevier, Amsterdam, Netherlands.

López-Mariño, A., Luis-Calabuig, E., Fillat, F., and Bermudez, F.F. 2000. Floristic composition of established vegetation and the soil seed bank in pasture communities under different traditional management regimes. Agriculture, Ecosystems & Environment **78**(3):273-282

Malhi, S.S., Heier, K., Nielsen, K., Davies, W.E., and Gill, K.S. 2000. Efficacy of pasture rejuvenation through mechanical aeration of N fertilization. Canadian Journal of Plant Science **80**(4):813-815.

Malo, J.E., and Suarez, F. 1995. Establishment of pasture species on cattle dung: the role of endozoochorous seeds. Journal of Vegetation Science **6**(2):169-174.

Mapfumo, E., Chanasyk, D. S., Baron, V. S., and Naeth, M. A. 2000. Impacts on selected soil parameters under short-term forage sequences. Journal of Range Management **53**(5):466-470.

Mapfumo, E., Chanasyk, D. S., Naeth, M. A. and Baron, V. S. 1999. Soil compaction under grazing of annual and perennial forages. Canadian Journal of Soil Science **79**(1):191-199.

Mapfumo, E., Naeth, M. A., Baron, V. S., Dick, A. C. and Chanasyk, D. S. 2002. Grazing impacts on litter and roots: perennial versus annual grasses. Journal of Range Management **55**(1):16-22.

McGeehan, S.L. and Naylor, D.V. 1988. Automated instrumental analysis of carbon and nitrogen in plant and soil samples. Communication in Soil Science and Plant Analysis **19**(4):493-505.

McLeod, E.M., Banerjee, S., Bork, E.W., Hall, L.M., and Hare, D.D. 2015. Structural equation modeling reveals complex relationships in mixed forage swards. Crop Protection **78**:106-113.

McNeill, J. 1977. The biology of Canadian weeds: 25. *Silene alba* (Miller) E. H. L. Krause. Canadian Journal of Plant Science **57**(4):1103-1114.

Menalled, F. D., Kohler, K. A., Buhler, D. D. and Liebman, M. 2005. Effects of composted swine manure on weed seedbank. Agriculture, Ecosystems & Environment **111**(1):63-69.

Mikutta, R., Kleber, M., Kaiser, K., and Jahn, R. 2005. Review: organic matter removal from soil using hydrogen peroxide, sodium hypochlorite, and disodium peroxodisulfate. Soil Science Society of America Journal **69**:120-135.

Milchunas, D.G., Sala, O.E. and Lauenroth, W.K. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. The American Naturalist **132**(1):87-106.

Miller, A.J. 2013. Recovery of legumes in northern temperate pastures following the application of broadleaf herbicides. M. Sc. Thesis, University of Alberta, Agricultural, Food and Nutritional Science. Edmonton, Alberta.

Miller, A.J., Bork, E.W., Hall, L.M., and Summers, B. 2015. Long-term forage dynamics in pastures sprayed with residual broadleaf herbicide: A test of legume recovery. *Canadian Journal of Plant Science* **95**(1):43-53.

Miller, J., Chanasyk, D., Curtis, T., Entz, T., and Willms, W. 2010. Influence of streambank fencing with a cattle crossing on riparian health and water quality of the Lower Little Bow River in Southern Alberta, Canada. Agricultural Water Management **97**(2):247-258.

Mills, A. and Sina Adl, M. 2006. The effects of land use intensification on soil biodiversity in the pasture. Canadian Journal of Plant Science **86**(Special Issue):1339-1343.

Moore, R.J. 1975. The biology of Canadian weeds. 13. *Cirsium arvense* (L.) Scop. Canadian Journal of Plant Science **55**:1033-1048.

Naeth, M.A. Bailey, A.W., Pluth, D.J., Chanasyk, D.S., and Hardin, R.T. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland. Journal of Range Management **44**(1):7-12.

Ojima, D. S., Schimel, D. S., Parton, W. J. and Owensby, C. E. 1994. Long-and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry **24**(2):67-84.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, K., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson. G.L., Solymos, P., M., Stevens, M.H., Szoecs, E., and Wagner, H. 2017. vegan: Community ecology package. R package version 2.4-4. https://CRAN.R-project.org/package=veg

Pleasant, J.M., and Schlather, K. J. 1994. Incidence of weed seed in cow (*Bos* sp.) manure and its importance as a weed source for cropland. Weed Technology **8**(2):304-310.

Ponomarenko, E. V., and Anderson, D. W. 2001. Importance of charred organic matter in Black Chernozem soils of Saskatchewan. Canadian Journal of Soil Science **81**:285-297.

Pyle, L., Hall, L.M. and Bork, E.W. 2017. Linking management practices with range health in northern temperate pastures. Canadian Journal of Plant Science.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.https://www.R-project.org/

Rashford, B.S., Walker, J.A., and Bastian, C.T. 2011. Economics of grassland conversion to cropland in the prairie pothole region. Conservation Biology **25**(2):276-284.

Ren, L. and Bai, Y. 2017. Burning modifies composition of emergent seedlings in fescue prairie. Rangeland Ecology & Management **70**(2):230-237.

Rook, A.J., Dumont, B., Isselstein, J., Osoro, K., WallisDeVries, M.F., Parente, G., and Mills, J. 2004. Matching type of livestock to desired biodiversity outcomes in pastures – a review. Biological Conservation **119**(2):137-150.

Rowan, R.C. 1994. Are small-acreage livestock producers real ranchers? Rangelands **16**(4):161-166.

Rowe, J.S. 1987. Status of the aspen parkland in the Prairie Provinces. Endangered Species in the Prairie Provinces, Prairie Conservation and Endangered Species Conference, pp 27-33.

Rozema J., Tosserams M., Nelissen H. J. M, Vanheerwaarden L., Broekman R. A., and Flierman, N. 1997. Stratospheric ozone reduction and ecosystem processes: enhanced UV-B radiation affects chemical quality and decomposition of leaves of the dune grassland species *Calamagrostis epigeios*. Plant Ecology **128**(17):284–94.

Sanderson, M.A., Skinner, R.H., Barker, D.J., Edwards, G.R., Tracy, B.F., and Wedin, D.A. 2004. Plant species diversity and management of temperate forage and grazing land ecosystems. Crop Science **44**:1132-1144.

Savory, A. and Parsons, D. S. 1980. The Savory grazing method. Rangelands **2**: 234–237.

Sayre, N.E. 2004. Viewpoint: The need for qualitative research to understand ranch management. Journal of Range Management **57**:668-674.

Schellberg, J., Möseler, B.M., Kühbauch, W., and Rademacher, I.F. 2001. Long-term effects of fertilizer on soil nutrient concentration, yield, forage quality and floristic composition of a hay meadow in the Eifel Mountains, Germany. Grass and Forage Science **54**(3):195-207.

Sharafatmandrad, M., Mesdaghi, M., Bahremand, A., and Barani, H. 2010. The role of litter in rainfall interception and maintenance of superficial soil water content in arid rangeland in Khabr National Park in south-eastern Iran. Arid Land Research and Management **24**:213-222.

Šimek, M., Brůček, P., Hynšt, J., Uhlířová, E. and Petersen, S. O. 2006. Effects of excretal returns and soil compaction on nitrous oxide emissions from a cattle overwintering area. Agriculture, Ecosystems & Environment **112**(2):186-191.

Sinkins, P.A., and Otfinowski, R. 2012. Invasion or retreat? The fate of exotic invaders on the northern prairies, 40 years after cattle grazing. Plant Ecology **213**(8):1251-1262.

Smoliak, S. 1974. Range vegetation and sheep production at three stocking rates on *Stipa-Bouteloua* prairie. Journal of Range Management **27**(1): 23-26.

Stephenson, G. R. and Veigel, A. 1987. Recovery of compacted soil on pastures used for winter cattle feeding. Journal of Range Management **40**(1):46-48.

Symstad, A.J. and Jonas, J.L. 2011. Incorporating biodiversity into rangeland health: Plant species - richness and diversity in the Great Plains grasslands. Rangeland Ecology & Management **64**(6):555-572.

Tannas, S. 2011. Mechanisms regulating *Poa pratensis* L. and *Festuca campestris* Rybd. Within the foothills fescue grasslands of southern Alberta. Ph.D. Thesis, University of Alberta, Department of Agriculture, Food and Nutritional Science. Edmonton, Alberta.

Tannas, S., Hewins, D.B., and Bork, E.W. 2015. Isolating the role of soil resources, defoliation, and interspecific competition on early establishment of late successional bunchgrass *Festuca campestris*. Restoration Ecology **23**(4):366-374.

Teague, R. Provenza, F., Kreuter, U., Steffens, T. and Barnes, M. 2013. Multi-paddock grazing on rangelands: Why the perceptual dichotomy between research results and rancher experience? Journals of Environmental Management **128**:699-717.

Thomas, A.G. 1985. Weed survey system used in Saskatchewan for cereal and oilseed crops. Weed Science **33**(1):34-43.

Trumble, J.T. and Kok, L.T. 1982. Integrated pest management techniques in thistle suppression in pastures of North America. Weed Research **22**:345-359.

Vujnovic, K., Wein, R., and Dale, M.R.T. 2000. Factors determining the centrifugal organization of remnant Festuca grassland communities in Alberta. Journal of Vegetation Science **11**:127-134.

Westoby, M., Walker, B., and Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management **42**(4):266-274.

Whicker, A. D. and Detling, J. K. 1988. Ecological consequences of prairie dog disturbances. BioScience **8**(11):778-785.

Willms, W.D., McGinn, S.M., and Dormaar, J.F. 1993. Influence of litter on herbage production in the Mixed Prairie. Journal of Range Management **46**(4):320-324.

Willms, W. D., and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Willms, W.D., Smoliak, S., and Dormaar, J.F. 1985. Effects of stocking rate on a rough fescue grassland vegetation. Journal of Range Management **38**(3):220-225.

Willms, W.D., Smoliak, S., and Bailey, A.W. 1986. Herbage production following litter removal on Alberta native grasslands. Journal of Range Management **39**(6):536-540.

[WWF] World Wildlife Fund. 2016. Plowprint report: Facts & Figures. Accessed February 22, 2017. https://c402277.ssl.cf1.rackcdn.com/publications/947/files/original/plowprint_AnnualReport_2016_Final_REV09192016.pdf

Yeomans, J.C. and Bremner, J.M. 1991. Carbon and nitrogen analysis of soils by automated combustion techniques. Communication in Soil Science and Plant Analysis **22**(9-10):843-850.

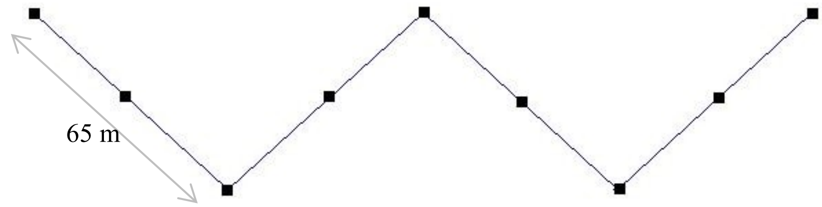


Figure 4.1. W-shaped transect used for vegetation and soil of vegetation in each pasture. Each segment of the 'W' is 65 m, totaling 260 m. Black squares represent points where foliar cover was measured, every 32.5 m, using a 50 cm x 50 cm quadrat.

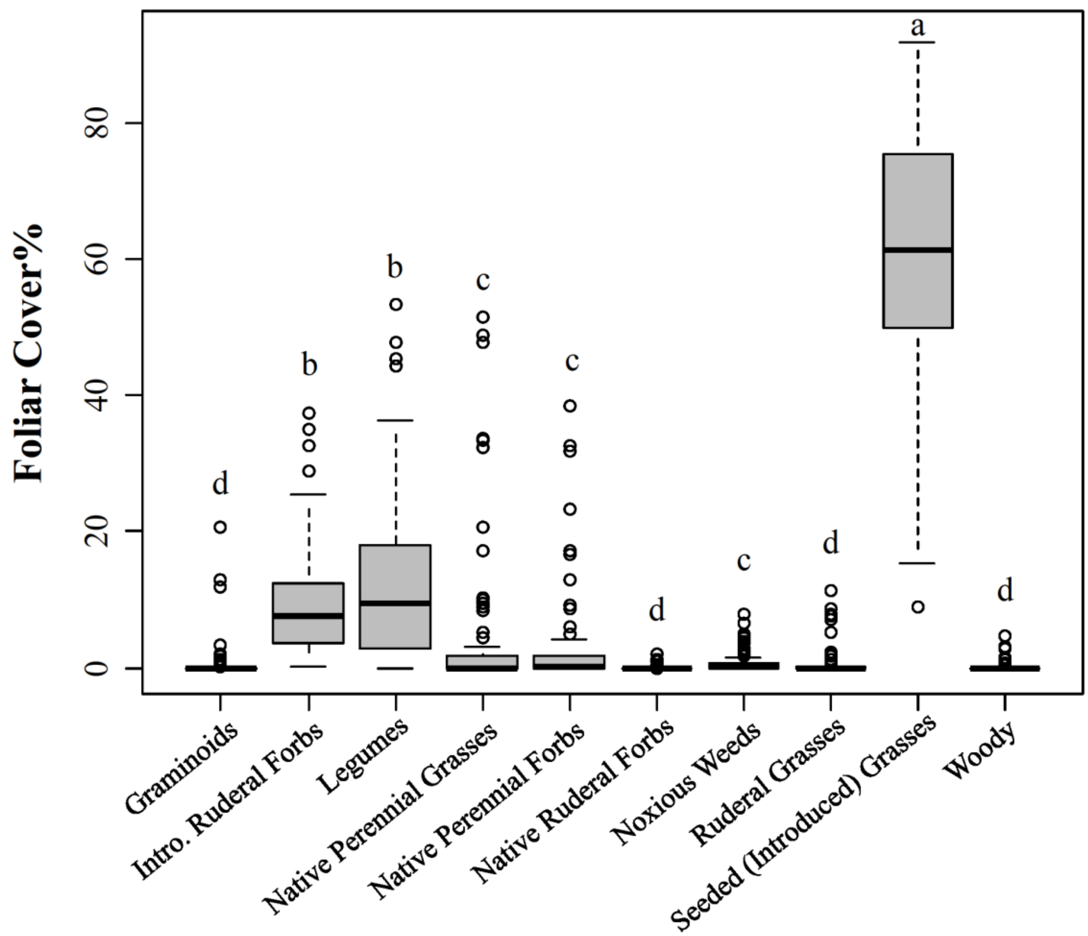


Figure 4.2. Median foliar cover (% ± IQR) of various functional plant groups present within parkland pastures of north central Alberta, Canada (Kruskal-Wallis, X^2 = 649.82, df=9, P < 0.001). Component medians with different letters differ at P < 0.05 following Bonferroni correction.

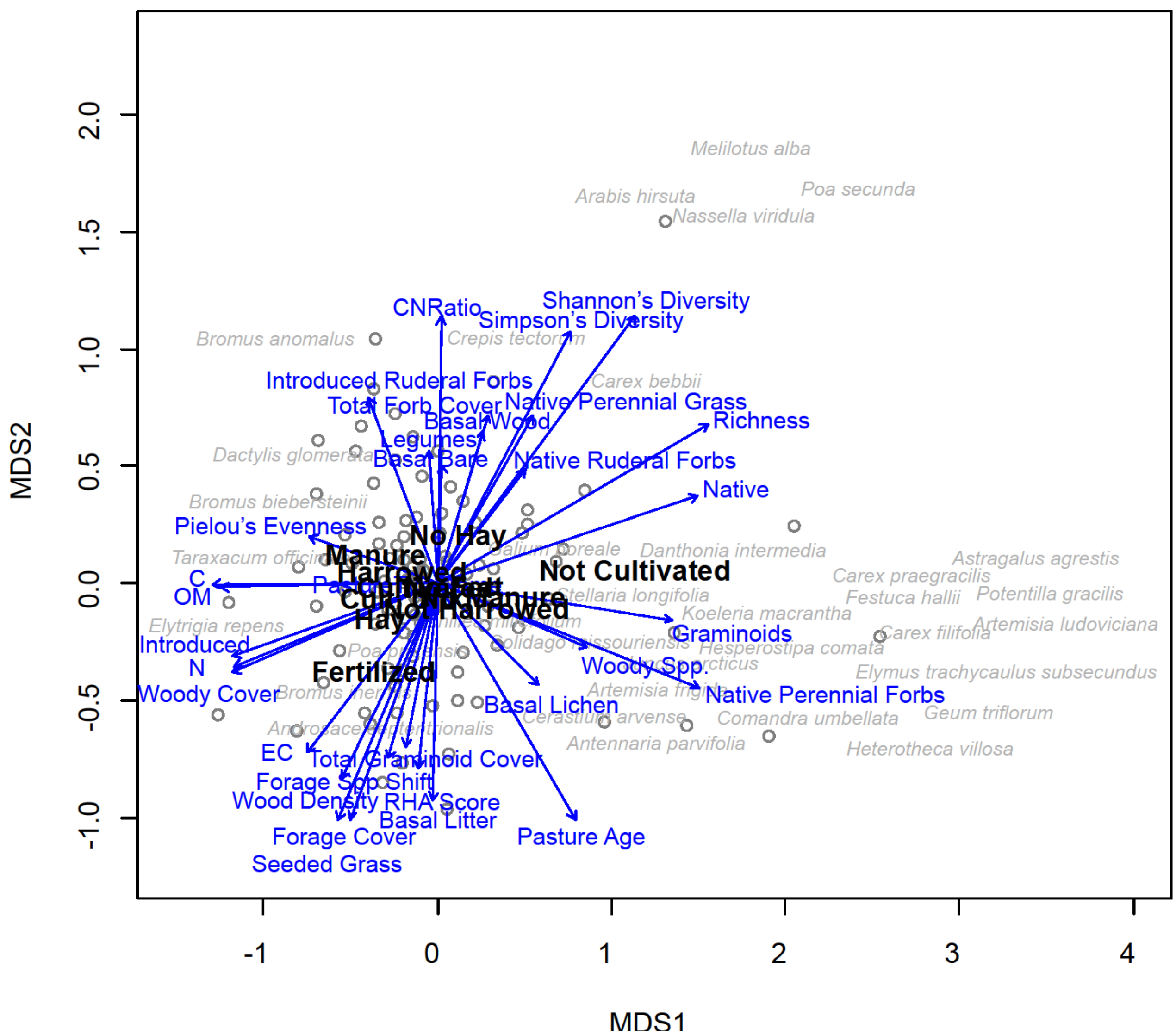


Figure 4.3. NMDS ordination of pasture plant community composition (distance = Bray-Curtis, dimensions = 2, stress = 0.23). Centroids of management factors (bolded) and the vectors for soil properties, RHA scores total and categorical), plant functional group cover, and other indices (blue text) plotted were all significant at $P < 0.05$, while listed plant species (grey text) were significant at $P < 0.01$. Vectors 'Forage Cover', 'Forage Spp. Shift', 'Woody Cover', and 'Wood Density' were derived from RHA scores, the vector indicates sites with high scores.

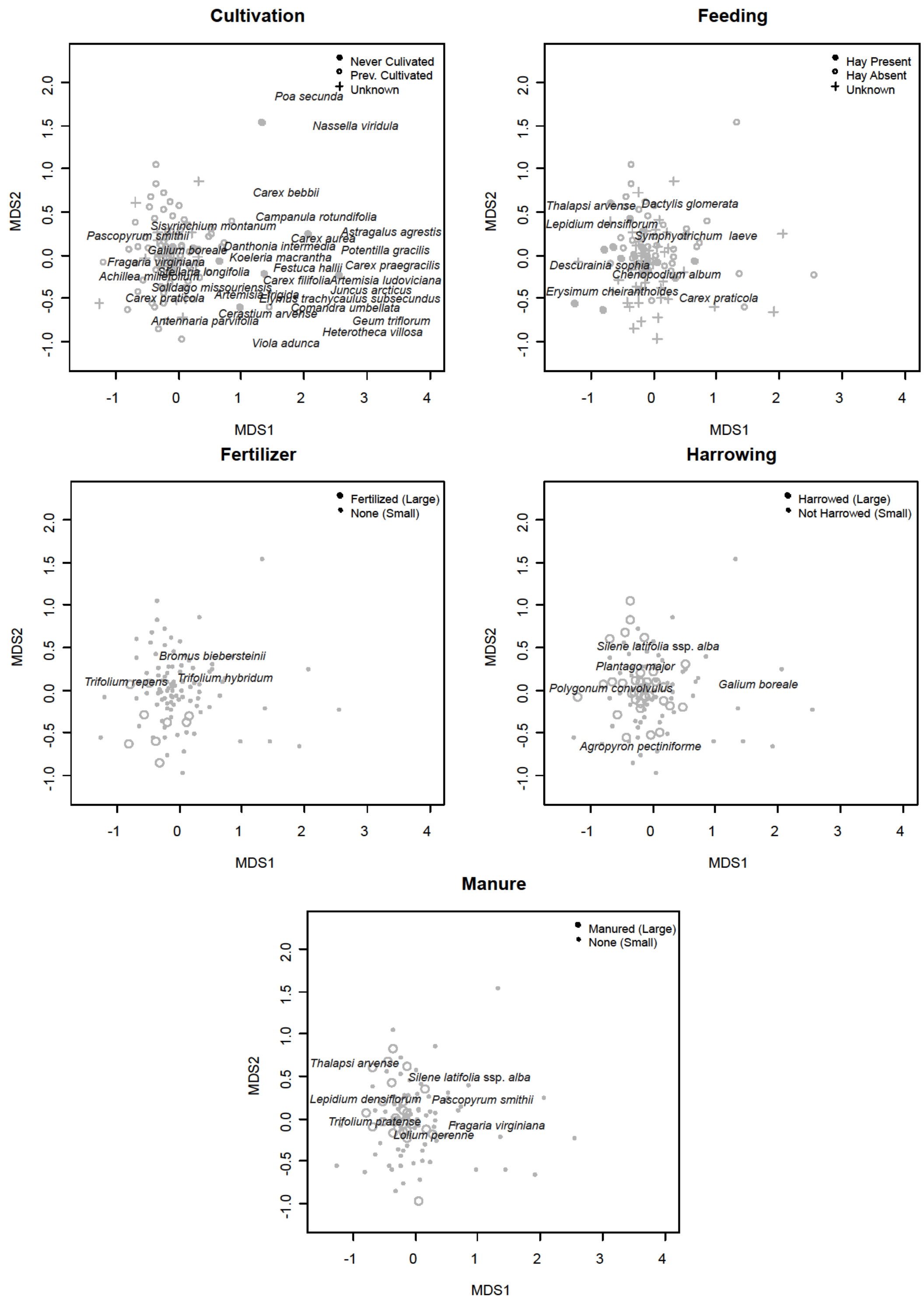


Figure 4.4. NMDS ordinations of plant community composition (distance = Bray-Curtis, dimensions = 2, stress = 0.23), using the same scores from Fig 4.3 and showing the relationship between significant management factors and their indicator plant species.

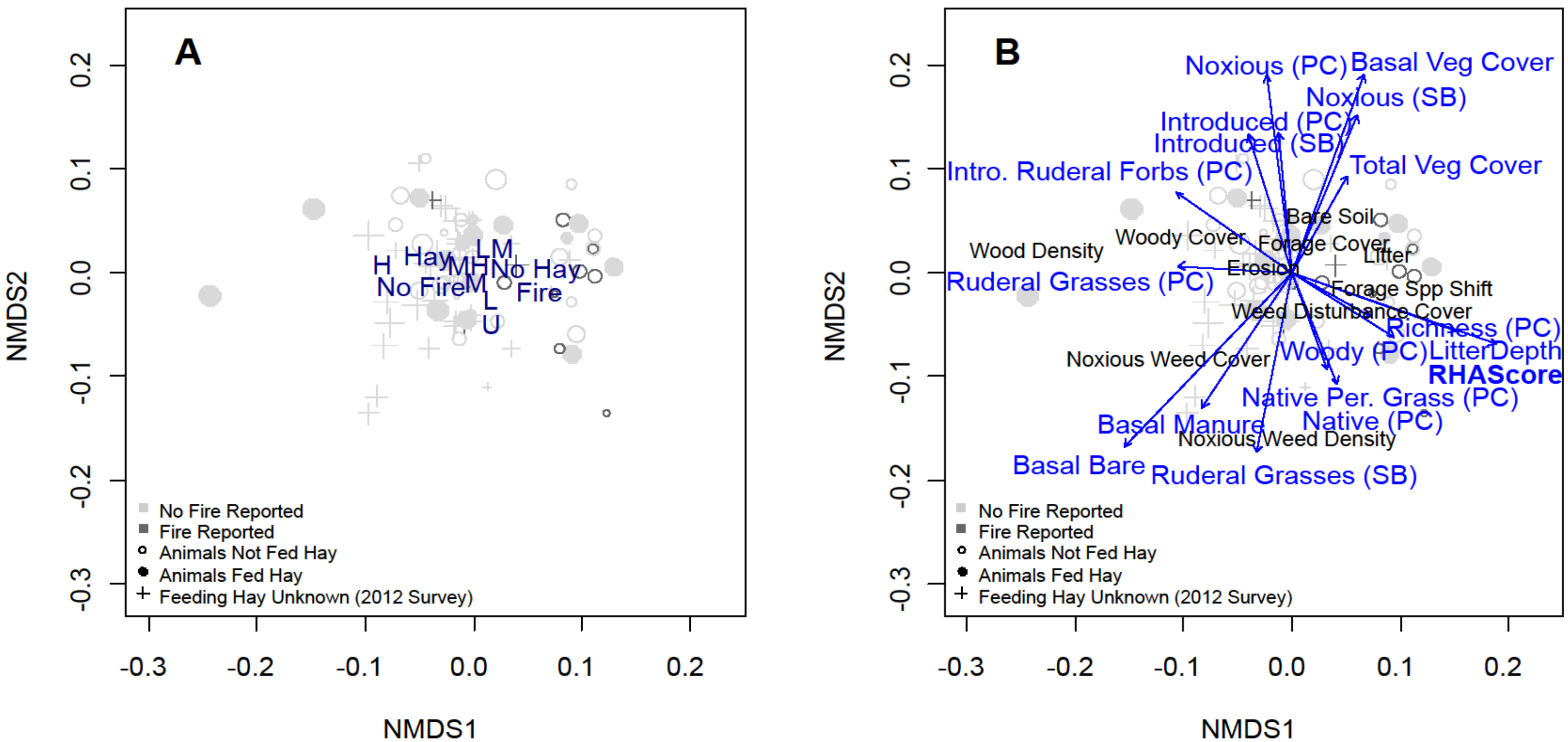


Figure 4.5. NMDS ordination of rangeland health scores (distance = Euclidean, dimensions = 2, stress = 0.14). A). Symbols of sites show significant management factors (P < 0.05) in the ordination (identified with *envfit::vegan* in R software). Larger symbols indicate higher grazing intensities, smaller symbols indicate a lower grazing intensity. Dark grey symbols indicate sites that were identified as recently burned in the interview, light grey represents no report of fire. A solid circle indicates a pasture where animals were being fed hay, an open symbol indicates no feed, and a cross represents sites surveyed in 2012 where these data were not collected. Centroids of the management factors were also plotted. B) Site scores and symbols are the same as on panel A, with rangeland health categories shown in black text. Note that their position indicates sites that had high scores in each category and should be interpreted as 'healthy' in that category. Significant gradients are plotted in blue, including total RHA score, basal properties of plots (basal bare ground, manure, and vegetation cover), litter depth, plant community (PC) characteristics and seed bank (SB) characteristics for which only noxious weed seed density responded (identified with *envfit::vegan*, P < 0.05). Soil properties were also fit to the ordination and no significant gradients found. Summary of significant management, plant community, seed bank, and environmental variables are in Appendix 4.2.

Table 4.1. Dominant plant species ranked by mean foliar cover (±SD) found across all pastures (n=102) from central Alberta during 2012 and 2013.

Scientific Name	**Common Name**	**Mean Foliar Cover (%)**
Poa pratensis L.	Kentucky Bluegrass	25.2 (±17.3)
Bromus inermis Leyss. subsp. *inermis*	Smooth Brome	13.7 (±14.9)
Taraxacum officinale F.H. Wigg.	Dandelion	8.6 (±7.6)
Elytrigia repens (L.) Gould.	Quack Grass	8.1 (±12.9)
Trifolium repens L.	White Clover	6.8 (±9.9)
Bromus biebersteinii Roem. & Schult.	Meadow Brome	5.0 (12.1)
Festuca rubra L.	Red Fescue	3.1 (±7.7)
Medicago sativa L.	Common Alfalfa	2.3 (±6.8)
Trifolium hybridum L.	Alsike Clover	1.95 (±3.1)
Dactylis glomerata L.	Orchardgrass	1.93 (±6.9)

Table 4.2. Summary of PerMANOVA assessment of vegetation composition responses to various producer management factors taking place in northern temperate pastures sampled in 2012 and 2013. Analysis was conducted using a Bray-Curtis distance metric, and 999 permutations. Significance was set at $P < 0.05$, with those values meeting this level shown in bold.

Management Factors	Mean Square	F Model	R^2	P Value
Owned or Rented	0.24	1.11	0.01	0.338
Previous Cultivation	0.41	1.91	0.04	**0.016**
Grazing System	0.23	1.06	0.02	0.339
Timing of Grazing	0.26	1.17	0.03	0.236
Gr. System * Timing of Gr.	0.26	1.21	0.05	0.176
Herbivore Type(s)	0.23	1.04	0.04	0.401
Herbicide	0.28	1.26	0.01	0.232
Fertilized	0.33	1.51	0.01	0.133
Manure Spreading	0.39	1.79	0.02	0.061
Harrowed	0.24	1.09	0.01	0.378
Aerated	0.17	0.79	0.01	0.622
Swathed or Mowed	0.26	1.17	0.01	0.288
Fed Hay in Pasture*	0.44	1.95	0.03	**0.033**
Burrowing Mammals	0.35	1.60	0.02	0.099
Fire (Survey)	0.66	3.07	0.03	**0.003**
Fire (Charcoal in Soil)	0.31	1.43	0.01	0.157
Grazing Intensity	0.28	1.27	0.06	0.116

Table 4.3. Significance of PerMANOVA contrasts evaluating cultivation effects on plant community composition in 102 pastures surveyed across north central Alberta during 2012 and 2013. Analysis was conducted using a Bray-Curtis distance metric, and 999 permutations. Significance was set at $P < 0.05$, with those values meeting this level shown in bold.

Management Factor	Contrast	Mean Square	F Model	R^2	P Value
Previous Cultivation	Cultivated vs Never	0.55	2.47	0.03	**0.012**
	Cultivated vs Unknown	0.26	1.22	0.01	0.264
	Never vs Unknown	0.51	2.50	0.10	**0.005**

Table 4.4. Summary of indicator species analysis for plant community species association with various management factors documented on 102 pastures of north central Alberta during 2012 and 2013. Analysis was run with 999 permutations, and only species significant at P < 0.01 are shown.

Management	Category	Species	A	B	P value
Cultivation	Never Cultivated	*Achillea millefolium*	0.81	0.88	0.002
		Antennaria parvifolia	0.91	0.38	0.005
		Artemisia frigida	0.83	0.38	0.010
		Artemisia ludoviciana	1.00	0.25	0.006
		Astragalus agrestis	1.00	0.25	0.006
		Campanula rotundifolia	1.00	0.38	0.001
		Carex aurea	1.00	0.25	0.008
		Carex bebbii	0.88	0.25	0.010
		Carex filifolia	0.88	0.50	0.001
		Carex praegracilis	1.00	0.38	0.001
		Cerastium arvense	0.95	0.38	0.005
		Danthonia intermedia	0.93	0.50	0.001
		Elymus trachycaulus ssp. *subsecundus*	1.00	0.25	0.005
		Festuca hallii	1.00	0.38	0.001
		Galium boreale	0.96	0.50	0.004
		Heterotheca villosa	0.95	0.25	0.005
		Juncus arcticus ssp. *balticus*	0.97	0.25	0.005
		Koeleria macrantha	0.98	0.25	0.010
		Nassella viridula	1.00	0.25	0.005
		Pascopyrum smithii	0.67	0.63	0.005
		Poa secunda	1.00	0.25	0.005
		Potentilla gracilis	1.00	0.25	0.006
		Sisyrinchium montanum	0.70	0.50	0.008
		Solidago missouriensis	0.84	0.63	0.001
		Stellaria longifolia	0.91	0.38	0.003
	Unknown	*Elytrigia repens*	0.62	0.94	0.008
Grazing System	None (Abandoned)	*Danthonia intermedia*	0.99	0.50	0.003
		Stellaria longipes	0.95	0.50	0.005
Timing of Grazing	Never (Abandoned)	*Stellaria longipes*	0.97	0.50	0.004
	Winter	*Pascopyrum smithii*	0.91	1.00	0.005
Gr. System x Timing	Never (Abandoned)	*Stellaria longipes*	0.97	0.50	0.004
	Winter	*Pascopyrum smithii*	0.91	1.00	0.005
Herbicide	Sprayed	*Festuca rubra*	0.78	0.63	0.007
Fertilization	Not Fertilized	*Trifolium hybridum*	0.98	0.84	0.001
		Trifolium repens	0.96	0.78	0.002
Harrowed	Harrowed	*Plantago major*	0.91	0.38	0.001
Aerated	Aerated	*Poa palustris*	0.89	0.75	0.006
Swathed or Mowed	Swath/Mowed	*Medicago sativa*	0.89	0.67	0.001
Hay Feeding (in pasture)	Hay	*Chenopodium album*	0.79	0.56	0.001
		Descurainia sophia	0.69	0.25	0.011
		Erysimum cheiranthoides	0.89	0.25	0.003
	No Hay	*Dactylis glomerata*	0.86	0.37	0.006
Burrowing Mammals	Absent	*Lathyrus ochroleucus*	0.99	0.17	0.004
Recent Fire	Fire (Survey)	*Fragaria virginiana*	0.88	0.53	0.001
		Galium boreale	0.67	0.40	0.010
		Lathyrus ochroleucus	0.99	0.40	0.001
		Rosa acicularis	0.95	0.27	0.002
		Sonchus arvensis	0.88	0.33	0.004
		Trifolium pratense	0.72	0.53	0.008
Historical Fire	Fire (Charcoal in Soil)	*Fragaria virginiana*	0.86	0.35	0.004
		Lathyrus ochroleucus	0.99	0.23	0.001
		Vicia americana	0.80	0.39	0.003
Grazing Intensity	U	*Danthonia intermedia*	0.97	0.50	0.001
		Stellaria longipes	0.90	0.50	0.005

A = Probability of occurring, B = Fidelity

Table 4.5. Indicator species analysis of plant community functional group association with various pasture management factors evaluated across 102 pastures in north central Alberta during 2012 and 2013. Analysis was run with 999 permutations, and results with P < 0.1 are shown, significant results (P < 0.05) are bolded.

Management Factors	Category	Species	A	B	P value
Ownership	Owned	Native Ruderal Forbs	0.72	0.40	0.073
Cultivation	Not Cultivated	Graminoids	0.90	0.75	**0.001**
		Native Perennial Grasses	0.82	0.75	**0.002**
		Native Perennial Forbs	0.85	1.00	0.079
		Native Species	0.79	1.00	**0.001**
	Unknown	Introduced Species	0.38	1.00	**0.001**
		Noxious Weeds	0.64	0.65	0.073
		Seeded (Introduced) Grasses	0.39	1.00	**0.013**
	Not Cultivated + Unknown	Ruderal Grasses	0.73	0.40	0.079
Grazing System	Continuous	Introduced Species	0.36	1.00	**0.039**
	Never (Abandoned)	Graminoids	0.74	0.50	0.051
		Native Species	0.61	1.00	**0.029**
Time of Grazing	Never (Abandoned)	Graminoids	0.83	0.50	0.079
Gr. System x Timing of Gr.	Continuous	Introduced Species	0.36	1.00	0.069
	Never (Abandoned)	Graminoids	0.74	0.50	0.065
		Native Species	0.60	1.00	**0.039**
Type of Herbivore	Multiple Species	Seeded (Introduced) Grasses	0.24	1.00	0.070
	No Livestock	Graminoids	0.74	0.50	0.065
		Native Species	0.50	1.00	0.058
Herbicide	Not Sprayed	Native Ruderal Forbs	1.00	0.27	0.065
Harrowing	Not Harrowed	Woody Species	0.89	0.15	0.097
Feeding Hay (in pasture)	Not Fed + Unkn. History	Native Perennial Grasses	0.95	0.43	0.069
Burrowing Mammals	Absent	Woody Species	0.90	0.19	**0.009**
Fire (Survey)	Fire	Woody Species	0.83	0.27	**0.019**
Grazing Intensity	U + L + LM	Native Perennial Grasses	0.83	0.57	**0.043**
	U + L + LM + M + MH	Native Perennial Forbs	1.00	0.68	**0.027**

A = Probability of occurring, B = Fidelity

Table 4.6. Significant one-way ANOVA and Kruskal-Wallis tests (univariate) for effects of management factors on the abundance of various primary vegetation cover groupings documented across 102 pastures of north central Alberta during 2012 and 2013.

	Graminoids		Broad Leaf		Native		Introduced	
Management	**F Value**	**P Value**	**F Value**	**P Value**	X^2	**P Value**	X^2	**P Value**
Owned or Rented	1.397	0.240	2.876	0.093	0.213	0.644	0.208	0.649
Previous Cultivation	1.111	0.333	1.071	0.347	**12.195**	**0.002**	**7.376**	**0.025**
Grazing System	0.204	0.816	0.113	0.893	2.768	0.251	3.942	0.139
Timing of Grazing	0.899	0.445	0.401	0.753	4.145	0.246	2.681	0.443
System x Timing	1.026	0.398	0.529	0.714	4.746	0.314	6.190	0.185
Herbivore Type(s)	2.408	0.055	1.106	0.358	3.730	0.444	4.868	0.301
Herbicide	**4.372**	**0.039**	2.125	0.148	1.073	0.300	**4.857**	**0.028**
Fertilized	3.442	0.067	**16.707**	**<0.001**	3.613	0.057	0.080	0.777
Manure Spreading	1.124	0.292	0.349	0.556	**9.820**	**0.002**	**4.762**	**0.029**
Harrowed	0.316	0.576	0.921	0.340	**6.577**	**0.010**	3.606	0.058
Aeration	1.566	0.214	0.628	0.430	0.107	0.743	0.178	0.673
Swathed or Mowed	2.422	0.123	2.307	0.132	0.713	0.399	0.007	0.934
*Fed Hay in Pasture Sampled	0.004	0.952	**4.137**	**0.047**	**6.107**	**0.013**	0.010	0.922
Burrowing Mammals	0.086	0.770	0.035	0.853	1.007	0.316	1.749	0.186
Fire (Survey)	0.307	0.581	0.019	0.892	**7.781**	**0.005**	1.463	0.226
Fire (Charcoal in Soil)	1.445	0.232	3.781	0.055	0.822	0.365	0.014	0.904
Grazing Intensity	2.255	0.055	0.810	0.545	10.263	0.068	1.719	0.886
Health	**13.905**	**<0.001**	**7.890**	**0.001**	0.378	0.828	2.531	0.282

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1

* Analysis based on 58 sites from the 2013 survey

Table 4.7. Summary LS means (±SE) for all significant management effects on the cover of primary vegetation groups. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups.

Management	Treatment	Graminoids	Broadleaf	Native	Introduced
Ownership	Owned		26.5 (±1.6)		
	Rented		19.1 (±4.8)		
Cultivation	Cultivated			6.7 (±1.4) b	85.7 (±1.7) a
	Never Cultivated			37.2 (±4.3) a	54.5 (±5.2) b
	Unknown			6.7 (±1.4) b	85.9 (±3.6) ab
Herbivore Type(s)	Cattle	67.0 (±1.7)			
	Horses	55.8 (±4.0)			
	Multiple Species	75.6 (±5.9)			
	Other	64.7 (±7.2)			
	Wildlife (None)	68.8 (±7.2)			
Herbicide	Sprayed in Last 3 Years	73.1 (±3.7) a			90.3 (±4.2) a
	Not Sprayed Recently	64.8 (±1.6) b			82.0 (±1.8) b
Fertilized	Fertilized	74.8 (±4.9)	10.6 (±4.8) b	4.1 (±3.7)	
	Not Fertilized	65.2 (±1.5)	27.2 (±1.5) a	9.3 (±1.6)	
Manure Spreading	Manured			3.1 (±2.9) b	88.4 (±3.3) a
	Not Manured			10.3 (±1.7) a	81.6 (±1.9) b
Harrowed	Harrowed			3.9 (±2.5) b	88.0 (±2.8)
	Not Harrowed			10.8 (±1.8) a	80.9 (±2.0)
Fed Hay in Pasture Sampled	Hay		22.1 (±3.6) b	1.9 (±3.8) b	
	No Hay		29.6 (±2.1) a	11.0 (±2.3) a	
Fire (Survey)	Reported			14.6 (±3.8) a	
	Not Reported			7.5 (±1.6) b	
Grazing Intensity	U	68.8 (±7.2)		23.1 (±7.3)	
	L	61.0 (±4.8)		9.6 (±4.9)	
	LM	72.9 (±2.9)		11.1 (±3.0)	
	M	64.5 (±2.5)		8.1 (±2.5)	
	MH	66.5 (±3.0)		5.4 (±3.0)	
	H	55.5 (±5.1)		2.6 (±5.2)	
Health	Healthy	71.0 (±1.6) a	21.6 (±1.7) b		
	Problems	57.4 (±2.4) b	34.1 (±2.5) a		
	Unhealthy	50.7 (±6.6) b	30.7 (±7.1) ab		

Black: P < 0.05, Grey: P < 0.1

Table 4.8. Significant ANOVA and Kruskal-Wallis tests of management factors on the cover of specific plant functional groups, as sampled across 102 pastures across north central Alberta during 2012 and 2013.

	Native & Introduced				Introduced						Native									
	Legumes		Ruderal Grasses		**Noxious Weeds		Ruderal Forbs		Seeded Graminoids		Ruderal Forbs		Perennial Forbs		Perennial Grasses		Graminoids		Woody Spp.	
Management	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**
Owned or Rented	2.378	0.126	0.032	0.858	2.528	0.112	1.973	0.163	0.354	0.553	1.833	0.176	0.070	0.791	0.403	0.526	1.734	0.188	0.028	0.867
Previous Cultivation	1.393	0.253	4.794	0.091	3.443	0.179	1.568	0.214	**8.981**	**<0.001**	1.701	0.427	**12.180**	**0.002**	**7.461**	**0.024**	**17.771**	**<0.001**	3.668	0.160
Grazing System	0.653	0.523	1.240	0.538	2.205	0.332	0.115	0.892	1.779	0.174	1.321	0.517	0.637	0.727	1.551	0.460	3.934	0.140	0.672	0.715
Timing of Grazing	0.416	0.742	2.989	0.393	3.267	0.352	1.341	0.266	0.343	0.795	3.982	0.263	2.842	0.417	2.647	0.449	5.169	0.160	0.911	0.823
Gr. System x Timing of Gr.	0.581	0.677	3.223	0.521	5.116	0.276	1.067	0.377	1.401	0.239	5.191	0.268	3.004	0.557	2.890	0.576	9.254	0.055	1.271	0.866
Herbivore Type(s)	0.816	0.518	1.224	0.874	2.054	0.726	0.829	0.510	1.581	0.185	4.774	0.311	5.236	0.264	1.593	0.810	3.199	0.525	3.609	0.462
Herbicide	2.081	0.152	0.288	0.592	2.889	0.089	0.212	0.646	**4.052**	**0.047**	**5.357**	**0.021**	3.119	0.077	0.760	0.383	0.055	0.814	0.491	0.483
Fertilized	**10.752**	**0.001**	3.092	0.079	0.058	0.810	**4.842**	**0.030**	5.409	0.221	0.211	0.646	0.842	0.359	1.842	0.175	0.006	0.937	0.000	1.000
Manure Spreading	0.001	0.990	0.042	0.837	0.058	0.810	1.479	0.227	2.557	0.113	0.031	0.861	**9.474**	**0.002**	2.536	0.111	1.301	0.254	**3.947**	**0.047**
Harrowed	2.145	0.146	0.064	0.800	0.876	0.349	3.603	0.061	0.092	0.763	1.058	0.304	**4.199**	**0.040**	2.290	0.130	0.210	0.647	3.286	0.070
Aerated	0.886	0.349	**6.899**	**0.009**	0.111	0.739	0.215	0.644	0.900	0.345	0.020	0.888	0.858	0.354	0.363	0.547	1.239	0.266	0.496	0.481
Swathed or Mowed	**7.685**	**0.007**	0.022	0.882	0.010	0.921	0.526	0.470	1.397	0.240	0.292	0.589	2.446	0.118	0.094	0.760	2.937	0.087	1.176	0.278
*Fed Hay in Pasture Sampled	3.521	0.066	1.863	0.172	2.546	0.111	1.275	0.264	0.531	0.469	1.134	0.287	3.601	0.058	1.713	0.191	2.080	0.149	0.985	0.321
Burrowing Mammals	0.609	0.437	0.061	0.804	0.217	0.641	**4.175**	**0.044**	0.521	0.472	0.068	0.795	0.030	0.862	0.000	0.988	0.005	0.945	**5.370**	**0.020**
Fire (Survey)	1.595	0.210	0.509	0.476	0.045	0.832	**6.122**	**0.015**	0.008	0.928	0.096	0.757	**4.446**	**0.035**	0.437	0.509	0.089	0.765	**5.085**	**0.024**
Fire (Charcoal in Soil)	**5.574**	**0.020**	2.104	0.147	0.011	0.915	0.131	0.718	0.668	0.416	1.009	0.315	1.429	0.232	0.029	0.863	0.430	0.512	1.479	0.224
Grazing Intensity	2.110	0.071	5.499	0.358	3.188	0.671	1.535	0.186	1.437	0.218	4.877	0.431	10.678	0.058	7.403	0.192	4.994	0.417	5.118	0.402
Health	1.716	0.185	**7.195**	**0.027**	2.205	0.332	**8.902**	**<0.001**	**9.721**	**<0.001**	5.035	0.081	4.321	0.115	2.123	0.346	0.095	0.954	1.614	0.446

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$
* Analysis based on 58 sites from the 2013 survey
**Note noxious weeds include 1 graminoid species

Table 4.9. Summary of LS mean (±SE) cover values of various plant functional groups with significant treatment responses to various management factors. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups.

		Native & Introduced		Introduced			Native				
Management Factors	**Treatment**	**Legumes**	**Ruderal Grasses**	**Noxious Weeds**	**Ruderal Forbs**	**Seeded Grasses**	**Ruderal Forbs**	**Perennial Forbs**	**Perennial Grasses**	**Graminoids**	**Woody Spp.**
Cultivation	Cultivated		0.6 (±0.2)			63.1 (±1.9) a		2.0 (±0.7) b	2.7 (±0.9) b	0.3 (±0.3) b	
	Never Cultivated		0.5 (±0.7)			37.6 (±5.9) b		15.0 (±2.0) a	15.6 (±2.9) a	4.9 (±0.8) a	
	Unknown		1.1 (±0.5)			64.7 (±4.0) a		0.6 (±1.4) b	2.7 (±0.9) b	0.2 (±0.6) b	
Gr. System x Timing of Gr.	Abandoned									3.1 (±1.3)	
	All Year									0.0 (±0.9)	
	Growing Season (Cont.)									4.6 (±0.5)	
	Growing Season (Rot.)									7.3 (±0.4)	
	Winter									0.0 (±1.5)	
Herbicide	Sprayed in Last 3 Years			1.3 (±0.3)		69.5 (±4.4) a	0.00 (±0.08) b	0.5 (±1.7)			
	Not Sprayed Recently			0.6 (±0.1)		59.8 (±1.9) b	0.12 (±0.03) a	3.2 (±0.7)			
Fertilized	Fertilized	3.1 (±3.9) b	0.0 (±0.7)		5.4 (±2.6) b						
	Not Fertilized	13.4 (±1.2) a	7.4 (±0.2)		9.9 (±0.8) a						
Manure Spreading	Manured							0.4 (±1.3) b			0.0 (±0.1) b
	Not Manured							3.5 (±0.8) a			2.4 (±0.1) a
Harrowed	Harrowed				11.0 (±1.3)			1.0 (±1.1) b			0.0 (±0.1)
	Not Harrowed				8.8 (±1.0)			3.6 (±0.8) a			0.3 (±0.1)
Aerated	Aerated		3.4 (±1.0) a								
	Not Aerated		0.6 (±0.2) b								
Swathed or Mowed	Swath/Mow	23.4 (±3.8) a								0.0 (±0.9)	
	No Swath/Mow	11.4 (±1.2) b								7.2 (±0.3)	
Fed Hay in Pasture Sampled	Hay	8.4 (±2.9)						0.4 (±2.0)			
	No Hay	15.0 (±1.8)						4.3 (±1.2)			
Burrowing Mammals	Present				10.5 (±1.0) a						0.0 (±0.1) b
	Absent				8.1 (±1.2) b						0.4 (±0.1) a
Fire (Survey)	Reported				5.4 (±2.0) b			3.4 (±1.7) a			0.6 (±0.2) a
	Not Reported				10.2 (±0.8) a			2.6 (±0.7) b			0.1 (±0.1) b
Fire (Charcoal in Soil)	Present	15.8 (±2.1) a									
	Absent	11.0 (±1.4) b									
Grazing Intensity	U	6.9 (±5.8)						8.6 (±3.4)			
	L	20.4 (±3.8)						0.9 (±2.2)			
	LM	8.5 (±2.4)						2.9 (±1.4)			
	M	15.4 (±2.0)						3.1 (±1.1)			
	MH	9.6 (±2.4)						2.6 (±1.4)			
	H	13.8 (±4.1)						8.6 (±3.4)			
Health	Healthy		0.3 (±0.2) b		7.4 (±0.9) c	65.8 (±2.0) a	0.04 (±0.04)				
	Healthy with Problems		1.1 (±0.3) ab		12.3 (±1.3) b	55.2 (±3.0) b	0.19 (±0.05)				
	Unhealthy		3.4 (±0.9) a		23.2 (±3.6) a	34.9 (±8.2) c	0.47 (±0.15)				

Black: P < 0.05, Grey: P < 0.1

Table 4.10. Significant ANOVA and Kruskal-Wallis tests of management factors on plant richness, diversity, and evenness within 102 parkland pastures sampled across north central Alberta during 2012 and 2013.

	Richness		Shannon's Diversity		Simpson's Diversity		Pielou's Evenness	
Management Factors	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Owned or Rented	0.345	0.557	0.037	0.847	0.4631	0.4978	0.027	0.870
Previous Cultivation	**13.607**	**0.001**	**10.436**	**0.001**	**5.252**	**0.007**	**3.936**	**0.023**
Grazing System	0.178	0.915	0.386	0.681	0.310	0.734	0.225	0.799
Timing of Grazing	4.749	0.191	0.503	0.681	0.388	0.762	1.158	0.330
Gr. System x Timing of Gr.	4.749	0.314	0.595	0.667	0.493	0.741	0.928	0.451
Herbivore Type(s)	1.752	0.781	0.180	0.948	0.132	0.970	0.478	0.752
Herbicide	3.383	0.066	**5.054**	**0.027**	3.419	0.067	1.188	0.278
Fertilized	**5.617**	**0.018**	**12.369**	**0.001**	**13.693**	**0.000**	0.212	0.646
Manure Spreading	2.158	0.142	1.697	0.196	1.230	0.270	0.541	0.464
Harrowed	0.346	0.556	0.028	0.867	0.154	0.696	3.657	0.059
Aerated	0.735	0.391	1.611	0.207	2.100	0.151	3.706	0.057
Swathed or Mowed	2.655	0.103	0.139	0.711	0.021	0.884	3.646	0.059
*Fed Hay in Pasture Sampled	**6.132**	**0.013**	**8.911**	**0.004**	**6.319**	**0.015**	0.010	0.920
Burrowing Mammals	1.783	0.182	0.740	0.392	0.246	0.621	0.522	0.472
Fire (Survey)	**5.231**	**0.022**	**5.607**	**0.020**	**5.191**	**0.025**	0.055	0.816
Fire (Charcoal in Soil)	0.995	0.319	1.462	0.229	2.035	0.157	0.208	0.649
Grazing Intensity	1.487	0.915	0.298	0.913	0.476	0.793	0.446	0.815
Health	**7.033**	**0.030**	2.209	0.115	2.629	0.077	0.170	0.844

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1

*Analysis based on 58 sites from the 2013 survey

Table 4.11. Summary LS mean (±SE) values of plant richness, diversity, and evenness, for pastures sampled in relation to the management factors. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups.

Management Factors	Treatment	Richness	Shannon's Diversity	Simpson's Diversity	Pielou's Evenness
Cultivation	Cultivated	13.5 (±0.6) b	1.62 (±0.05) b	0.71 (±0.01) b	0.126 (±0.003) a
	Never Cultivated	23.9 (±1.7) a	2.27 (±0.14) a	0.83 (±0.0) a	0.100 (±0.010) b
	Unknown	13.5 (±1.2) b	1.54 (±0.10) b	0.68 (±0.03) b	0.115 (±0.007) ab
Herbicide	Sprayed in Last 3 Years	11.6 (±1.4)	1.44 (±0.11) b	0.67 (±0.03)	
	Not Sprayed Recently	14.8 (±0.6)	1.70 (±0.05) a	0.72 (±0.01)	
Fertilized	Fertilized	10.2 (±1.8) b	1.19 (±0.14) b	0.57 (±0.04) b	
	Not Fertilized	14.7 (±0.6) a	1.70 (±0.04) a	0.73 (±0.01) a	
Harrowed	Harrowed				0.129 (±0.005)
	Not Harrowed				0.119 (±0.003)
Aerated	Aerated				0.151 (±0.014)
	Not Aerated				0.121 (±0.003)
Swathed or Mowed	Swath/Mow				0.141 (±0.009)
	No Swath/Mow				0.120 (±0.003)
Fed Hay (in pasture)	Hay	12.8 (±2.5) b	1.50 (±0.10) b	0.78 (±0.05) a	
	No Hay	15.4 (±0.8) a	1.84 (±0.06) a	0.74 (±0.02) b	
Fire (Survey)	Reported	16.8 (±1.4) a	1.90 (±0.11) a	0.78 (±0.03) a	
	Not Reported	13.9 (±0.6) b	1.62 (±0.05) b	0.70 (±0.01) b	
Health	Healthy	13.7 (±0.7) b		0.51 (±0.02)	
	Healthy with Problems	15.3 (±1.0) a		0.56 (±0.03)	
	Unhealthy	18.0 (±2.8) a		0.66 (±0.08)	

Black: P < 0.05, Grey: P < 0.1

Table 4.12. Significant ANOVA and Kruskal-Wallis tests effects of management factors on various soil properties found across 102 pastures surveyed across north central, Alberta during 2012 and 2013.

	C (%)		N (%)		C:N		OM (%)		pH		EC (µS/cm)		Compaction (kg/cm^2)	
Management Factor	**X^2**	**P Value**	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Owned or Rented	0.790	0.374	0.894	0.345	1.296	0.258	1.516	0.221	0.190	0.664	3.000	0.086	**5.589**	**0.023**
Previous Cultivation	4.071	0.131	5.879	0.053	1.719	0.185	2.468	0.090	1.867	0.160	2.481	0.089	2.269	0.116
Grazing System	1.861	0.394	2.063	0.356	1.416	0.248	0.847	0.432	0.835	0.437	1.570	0.189	0.033	0.856
Timing of Grazing	3.322	0.345	2.612	0.455	0.315	0.815	0.904	0.442	0.812	0.490	0.892	0.448	0.386	0.682
Gr. System x Timing of Gr.	3.744	0.442	3.360	0.499	0.721	0.580	0.798	0.529	0.713	0.585	0.553	0.577	0.254	0.858
Herbivore Type(s)	3.665	0.453	2.735	0.603	0.679	0.608	0.441	0.779	0.762	0.553	1.492	0.211	0.231	0.874
Herbicide	3.489	0.062	3.524	0.061	0.545	0.462	2.127	0.148	1.078	0.302	0.021	0.885	0.000	0.984
Fertilized	0.010	0.920	0.012	0.911	0.009	0.923	0.011	0.915	0.412	0.522	3.088	0.082	0.104	0.749
Manure Spreading	**6.491**	**0.011**	**7.997**	**0.005**	0.920	0.340	**5.382**	**0.022**	0.752	0.388	**5.166**	**0.025**	2.672	0.109
Harrowed	**8.225**	**0.004**	**7.315**	**0.007**	0.282	0.596	**7.353**	**0.008**	0.253	0.616	1.116	0.293	1.349	0.252
Aerated	0.966	0.326	1.456	0.228	0.414	0.522	0.828	0.365	2.962	0.088	0.010	0.920	0.455	0.504
Swathed or Mowed	0.006	0.939	0.025	0.873	0.103	0.749	0.002	0.965	1.424	0.236	2.538	0.114	0.119	0.732
*Fed Hay (in pasture)	0.299	0.587	0.641	0.427	1.171	0.284	0.274	0.603	0.089	0.766	3.009	0.088	0.983	0.327
Burrowing Mammals	0.172	0.678	0.666	0.415	**8.923**	**0.004**	0.443	0.507	0.445	0.506	1.920	0.169	1.362	0.250
Fire (Survey)	0.383	0.536	1.013	0.314	**7.698**	**0.007**	0.037	0.847	0.074	0.787	0.107	0.744	2.092	0.155
Fire (Charcoal in Soil)	1.215	0.270	3.402	0.065	**5.411**	**0.022**	0.752	0.388	2.048	0.156	0.867	0.354	1.227	0.274
Grazing Intensity	10.466	0.063	10.663	0.058	1.298	0.271	1.664	0.151	0.981	0.434	**2.378**	**0.044**	2.371	0.068
Health	1.028	0.362	0.723	0.697	1.522	0.223	0.907	0.407	1.737	0.181	0.810	0.448	**3.783**	**0.031**

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1

*Includes only 58 sites from the 2013 survey

Table 4.13. Effect of significant management factors on the LS means (±SE) of various soil properties as sampled across 102 pastures of north central Alberta sampled during 2012 and 2013. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups.

Management Factor	**Treatment**	**C (%)**	**N (%)**	**C:N**	**OM (%)**	**pH**	**EC (µS/cm)**	**Compaction (kg/cm^2)**
Ownership	Owned						482.5 (±51.0)	2.1 (±0.1) a
	Rented						355.2 (±154.7)	1.3 (±0.4) b
Cultivation	Cultivated		0.39 (±0.04)		7.8 (±0.6)		458.9 (±55.9)	
	Never Cultivated		0.30 (±0.11)		5.8 (±2.0)		366.7 (±173.4)	
	Unknown		0.50 (±0.08)		9.2 (±1.4)		569.2 (±118.9)	
Herbicide	Sprayed in Last 3 Years	5.7 (±0.9)	0.50 (±0.08)					
	Not Sprayed Recently	4.6 (±0.4)	0.38 (±0.03)					
Fertilizing	Fertilized						606.2 (±162.9)	
	Not Fertilized						456.9 (±50.7)	
Manure Spreading	Manured	6.1 (±0.7) a	0.50 (±0.06) a		9.8 (±1.1) a		582.4 (±97.3) a	
	Not Manured	4.4 (±0.4) b	0.37 (±0.04) b		7.2 (±0.6) b		433.6 (±55.4) b	
Harrowed	Harrowed	5.6 (±0.6) a	0.45 (±0.06) a		9.2 (±1.0) a			
	Not Harrowed	4.4 (±0.5) b	0.37 (±0.04) b		7.2 (±0.7) b			
Aerated	Aerated					5.7 (±0.3)		
	Not Aerated					6.2 (±0.1)		
Fed Hay (in pasture)	Hay						581.7 (±89.7)	
	No Hay						391.9 (±53.0)	
Burrowing Mammals	Present			11.9 (±0.2) b				
	Absent			12.9 (±0.3) a				
Fire (Survey)	Reported			13.4 (±0.4) a				
	Not Reported			12.1 (±0.2) b				
Fire (Charcoal in Soil)	Present		0.33 (±0.06)	12.9 (±0.3) a				
	Absent		0.43 (±0.04)	12.0 (±0.2) b				
Grazing Intensity	U	3.6 (±1.9)	0.33 (±0.16)				304.7 (±238.6) ab	
	L	2.9 (±1.3)	0.23 (±0.11)				287.1 (±159.1) b	
	LM	5.3 (±0.8)	0.43 (±0.07)				575.5 (±97.4) ab	
	M	4.6 (±0.7)	0.37 (±0.06)				350.5 (±81.8) b	
	MH	5.0 (±0.8)	0.46 (±0.07)				514.3 (±99.5) ab	
	H	6.2 (±1.3)	0.48 (±0.12)				822.9 (±168.7) a	
Health	Healthy							2.07 (±0.14) a
	Problems							2.03 (±0.19) a
	Unhealthy							1.10 (±0.43) b

Black: P < 0.05, Grey: P < 0.1

Table 4.14. Summary of significant ANOVA and Kruskal-Wallis tests of management factors on various ground cover characteristics found in 102 pastures of north central Alberta during 2012 and 2013.

	Basal Veg Cover (%)		Litter Cover (%)		Litter Depth (cm)		Bare Ground (%)		Manure Cover (%)	
Management Factor	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**
Owned or Rented	0.697	0.406	2.155	0.145	3.246	0.075	1.305	0.253	1.573	0.210
Previous Cultivation	0.362	0.697	1.572	0.213	0.420	0.658	0.044	0.978	**6.409**	**0.041**
Grazing System	2.048	0.135	3.055	0.052	**3.926**	**0.023**	**7.516**	**0.023**	0.611	0.737
Timing of Grazing	1.725	0.167	**3.777**	**0.013**	**2.934**	**0.037**	5.629	0.131	5.703	0.127
Gr. System x Timing of Gr	1.767	0.142	**2.808**	**0.030**	2.223	0.072	9.006	0.061	5.719	0.221
Herbivore Type(s)	1.504	0.207	2.367	0.058	2.364	0.058	4.098	0.393	6.793	0.147
Herbicide	0.001	0.976	0.721	0.398	0.095	0.759	1.016	0.314	0.127	0.722
Fertilized	**4.503**	**0.036**	**7.430**	**0.008**	1.979	0.163	0.064	0.800	0.001	0.971
Manure Spreading	3.055	0.084	1.682	0.198	0.704	0.404	0.524	0.469	**7.021**	**0.008**
Harrowed	0.020	0.888	0.740	0.392	**7.806**	**0.006**	**5.211**	**0.022**	**4.360**	**0.037**
Aerated	1.413	0.237	2.638	0.108	0.173	0.678	0.007	0.931	1.358	0.248
Swathed or Mowed	0.504	0.479	0.005	0.941	1.518	0.221	0.535	0.464	0.161	0.688
*Fed Hay (in pasture)	1.155	0.287	0.110	0.742	2.916	0.093	0.444	0.505	**6.552**	**0.010**
Burrowing Mammals	0.937	0.335	0.325	0.570	**6.507**	**0.012**	1.221	0.269	0.703	0.402
Fire (Survey)	2.027	0.158	0.180	0.672	**7.175**	**0.009**	1.318	0.251	**7.589**	**0.006**
Fire (Charcoal in Soil)	0.187	0.667	0.586	0.446	2.344	0.129	0.700	0.403	**4.548**	**0.033**
Grazing Intensity	0.816	0.541	**3.439**	**0.007**	**9.552**	**<0.001**	6.894	0.229	**19.942**	**0.001**
Health	1.407	0.250	**18.831**	**0.000**	**12.865**	**<0.001**	**15.036**	**0.001**	**6.740**	**0.034**

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

*Includes only 58 sites from the 2013 survey

Note: Only trace amounts of rock, moss, and lichen were recorded.

Basal Veg. Cover = area of soil surface occupied by shoots, stems, and crowns of plants.

Table 4.15. Summary LS mean (±SE) responses of various ground cover characteristics in relation to different management factors. Within a column and management factor, treatment means with different letters differ, P < 0.05, Bonferroni corrected within groups.

Management Factors	Treatment	Basal Veg Cover (%)	Litter Cover (%)	Litter Depth (cm)	Bare Soil (%)	Manure Basal Cover (%)
Ownership	Owned			1.2 (±0.1)		
	Rented			2.1 (±0.4)		
Cultivation	Cultivated					0.8 (±0.2) b
	Never Cultivated					0.4 (±0.5) b
	Unknown					1.7 (±0.3) a
Grazing System	Abandoned (None)		67.1 (±8.1)	3.8 (±0.6) a	3.1 (±5.6) b	
	Continuous		46.2 (±2.6)	1.2 (±0.2) b	14.1 (±1.8) a	
	Rotational		48.7 (±2.1)	1.2 (±0.2) b	7.4 (±1.5) b	
Timing of Grazing	Abandoned (None)		67.1 (±7.9) a	3.8 (±0.6) a		
	All Year		41.9 (±5.6) b	1.0 (±0.4) b		
	Growing Season		47.5 (±1.7) ab	1.2 (±0.1) b		
	Winter		67.1 (±9.2) a	1.8 (±0.7) ab		
Gr. System x Timing of Gr.	Abandoned (None)		67.1 (±8.0) a	3.8 (±0.6)	3.1 (±5.6)	
	All Year		41.9 (±5.6) b	1.0 (±0.4)	18.2 (±4.0)	
	Growing Season (Continuous)		47.3 (±2.8) ab	1.2 (±0.2)	13.0 (±2.0)	
	Growing Season (Rotational)		47.7 (±2.1) ab	1.2 (±0.2)	7.6 (±1.5)	
	Winter		67.1 (±9.2) a	1.8 (±0.7)	2.8 (±6.5)	
Animals	Cattle		49.2 (±1.9)	1.2 (±0.1)		
	Horses		44.9 (±4.5)	1.4 (±0.3)		
	Multiple		41.8 (±6.6)	1.3 (±0.5)		
	Sheep/Alpaca		36.4 (±8.0)	0.7 (±0.6)		
	No Livestock		67.1 (±8.0)	3.8 (±0.6)		
Fertilization	Fertilized	29.4 (±5.3) b	62.3 (±5.3) a			
	Not Fertilized	41.7 (±1.6) a	47.1 (±1.7) b			
Manure Spreading	Manured	46.2 (±3.2)				1.3 (±0.3) a
	Not Manured	38.7 (±1.8)				0.9 (±0.2) b
Harrowed	Harrowed			0.8 (±0.2) b	11.7 (±2.0) a	1.2 (±0.2) a
	Not Harrowed			1.5 (±0.1) a	8.9 (±1.4) b	0.9 (±0.2) b
Fed Hay (in pasture)	Hay			0.9 (±0.3)		1.6 (±0.3) a
	No Hay			1.4 (±0.2)		0.6 (±0.2) b
Burrowing Mammals	Present			1.0 (±0.2) b		
	Absent			1.7 (±0.2) a		
Fire (Survey)	Reported			1.9 (±0.3) a		0.3 (±0.4) b
	Not Reported			1.2 (±0.1) b		1.1 (±0.2) a
Fire (Charcoal in Soil)	Present					0.6 (±0.3) b
	Absent					1.1 (±0.2) a
Grazing Intensity	U		67.1 (±7.8) a	3.8 (±0.5) a		2.5 (±0.7) bc
	L		48.7 (±5.2) ab	2.0 (±0.3) a		0.9 (±0.5) abc
	LM		55.1 (±3.2) a	2.0 (±0.2) a		0.5 (±0.3) c
	M		45.8 (±2.7) ab	0.9 (±0.2) b		0.5 (±0.2) bc
	MH		46.7 (±3.2) ab	0.7 (±0.2) b		1.5 (±0.3) ab
	H		35.2 (±5.5) b	0.8 (±0.4) b		1.8 (±0.5) a
Health	Healthy		54.4 (±1.7) a	1.6 (±0.1) a	6.1 (±1.3) b	0.9 (±0.2) b
	Problems		38.5 (±2.5) b	0.8 (±0.2) b	15.7 (±1.8) a	0.9 (±0.3) ab
	Unhealthy		25.3 (±7.1) b	0.4 (±0.6) b	26.6 (±5.1) a	2.1 (±0.7) a

Black: P < 0.05, Grey: P < 0.1

Table 4.16. Significant ANOVA effects on total RHA score found for 102 pastures of north central Alberta during 2012 and 2013.

Management Factor	F Value	P Value
Owned or Rented	0.181	0.671
Previous Cultivation	**3.438**	**0.036**
Grazing System	0.511	0.601
Timing of Grazing	**3.678**	**0.015**
System x Timing	**2.784**	**0.031**
Herbivore Type(s)	1.633	0.172
Herbicide	0.456	0.501
Fertilized	3.122	0.080
Manure Spreading	0.036	0.849
Harrowed	1.232	0.270
Swathed or Mowed	0.088	0.768
Fed Hay (in pasture)*	0.594	0.444
Burrowing Mammals	0.001	0.983
Fire (Survey)	1.514	0.221
Fire (Charcoal in Soil)	0.501	0.481
Grazing Intensity	**7.281**	**<0.001**

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1
*Analysis includes 58 sites from the 2013 survey

Table 4.17. Summary of LS means (±SE) for the total RHA scores for various management factors (P < 0.05).

Management Factor	Treatment	RHA Score
Cultivation	Cultivated	81.8 (±1.5) a
	Never Cultivated	74.1 (±4.7) ab
	Unknown	73.6 (±3.2) b
Timing of Grazing	Abandoned (None)	87.0 (±6.4) ab
	All Year	65.6 (±4.6) b
	Growing Season	80.35(±1.4) ab
	Winter	89.3 (±7.4) a
Gr. System x Timing of Gr.	Abandoned (None)	87.0 (±6.4) abc
	All Year (Continuous)	65.6(±4.6) c
	Growing Season (Continuous)	81.6 (±2.3) ab
	Growing Season (Rotational)	79.8 (±1.7) abc
	Winter (Rotational)	89.3 (±7.5) a
Fertilized	Fertilized	86.9 (±4.5)
	Not Fertilized	79.2 (±1.4)
Grazing Intensity	U	87.0 (±5.8) a
	L	87.3 (±3.9) a
	LM	85.7 (±2.4) a
	M	81.0 (±2.0) a
	MH	75.0 (±2.4) ab
	H	59.5 (±4.1) b
Health	Unhealthy	43.3 (±3.1) c
	Problems	67.2 (±1.1) b
	Healthy	87.9 (±0.8) a

Black: P < 0.05, Grey: P < 0.1

Table 4.18. Results of the PerMANOVA analysis assessing the impact of management factors on rangeland health scores. Analysis was conducted using both a Euclidean and Bray-Curtis distance metric, and 999 permutations. Significance was set at P < 0.05, with those values meeting this level shown in bold.

	Euclidean				Bray-Curtis			
Management Factors	**Mean Square**	**F Model**	**R^2**	**P Value**	**Mean Square**	**F Model**	**R^2**	**P Value**
Owned or Rented	0.005	0.299	0.003	0.783	0.005	0.299	0.003	0.786
Previous Cultivation	0.027	1.749	0.034	0.097	0.027	1.749	0.034	0.090
Grazing System	0.043	2.838	0.054	**0.032**	0.043	2.838	0.054	**0.039**
Timing of Grazing	0.042	2.866	0.081	**0.014**	0.042	2.866	0.081	**0.021**
Herbivore Type(s)	0.030	2.025	0.077	**0.046**	0.030	2.051	0.077	**0.049**
Herbicide	0.014	0.870	0.009	0.486	0.014	0.870	0.009	0.455
Fertilized	0.036	2.333	0.023	0.080	0.036	2.333	0.023	0.085
Manure Spreading	0.020	1.286	0.013	0.305	0.020	1.286	0.013	0.291
Harrowed	0.029	1.859	0.018	0.162	0.029	1.859	0.018	0.152
Aerated	0.021	1.355	0.013	0.237	0.021	1.355	0.013	0.251
Swathed or Mowed	0.014	0.910	0.009	0.447	0.014	0.910	0.009	0.439
Fed Hay (in pasture)*	0.025	1.476	0.026	0.235	0.025	1.476	0.026	0.214
Burrowing Mammals	0.016	1.052	0.010	0.398	0.016	1.052	0.010	0.366
Fire (Survey)	0.044	2.877	0.028	**0.041**	0.044	2.877	0.029	**0.042**
Fire (Charcoal in Soil)	0.000	0.025	0.000	0.905	0.000	0.025	0.000	0.891
Grazing Intensity	0.079	6.451	0.252	**0.001**	0.079	6.451	0.252	**0.001**
Significant Interactions								
Cultivation * Gr. System	0.035	2.716	0.088	**0.029**	0.035	2.716	0.088	**0.028**
Herbicide * Timing of Gr.	0.053	4.167	0.034	**0.017**	0.053	4.167	0.034	**0.021**
Gr. System * Timing of Gr.	0.035	2.388	0.090	**0.030**	0.035	2.388	0.090	**0.021**

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1

*Analysis based on 58 sites from the summer of 2013

Table 4.19. Indicator analysis of inversed RHA scores to detect which management actions are associated with deteriorating RHA scores. Analysis was run with 999 permutations, and results with P < 0.1 are shown, significant results (P < 0.05) are bolded.

Management	Treatment	RHA Category	A	B	P Value
Ownership	Rented	Noxious Weed Density	0.58	1.00	**0.022**
Cultivation	Never Cultivated	Forage Cover	0.59	1.00	**0.001**
		Cover of Tall Productive Forages	0.55	0.50	**0.043**
Grazing System	Abandoned (None)	Cover of Tall Productive Forages	0.58	0.50	**0.049**
	Continuous + Rotational	Hydraulic Function & Litter	1.00	0.58	**0.038**
Timing of Grazing	All Year	Anthropogenic Bare Soil	0.62	0.75	**0.039**
		Erosion	0.59	1.00	**0.002**
		Hydraulic Function & Litter	0.59	0.88	**0.011**
		Noxious Weed Density	0.38	1.00	**0.034**
	All Year + Winter	Anthropogenic Bare Soil	0.84	0.64	**0.023**
Gr. System x Timing of Gr.	All Year (Continuous)	Anthropogenic Bare Soil	0.53	0.75	**0.046**
		Erosion	0.48	1.00	**0.001**
		Hydraulic Function & Litter	0.45	0.88	**0.024**
		Noxious Weed Density	0.28	1.00	0.099
	All Year + Winter	Anthropogenic Bare Soil	0.71	0.67	**0.024**
	All Year + Continuous + Rotational +Winter	Hydraulic Function & Litter	1.00	0.58	0.061
Herbivores	Horses + Sheep/Alpaca + None	Woody Spp Density	0.87	0.52	**0.023**
	Sheep/Alpaca	Noxious Weed Density	0.29	1.00	0.082
Fed Hay in Pasture	Hay	Anthropogenic Bare Soil	0.52	0.44	**0.027**
	No Hay	Woody Spp Cover	0.58	0.23	0.090
		Woody Spp Density	0.51	0.42	0.082
Aerated	Aerated	Anthropogenic Bare Soil	0.74	0.75	**0.029**
Burrowing Mammals	Absent	Woody Spp Cover	0.72	0.21	0.089
Fire (Survey)	Reported	Woody Spp Cover	0.81	0.40	**0.003**
		Woody Spp Density	0.77	0.67	**0.001**
Fire (Charcoal in Soil)	Present	Woody Spp Cover	0.70	0.23	0.091
Grazing Intensity	MH	Erosion	0.32	0.87	**0.044**
	H	Anthropogenic Bare Soil	0.52	0.88	**0.002**
		Hydraulic Function & Litter	0.39	0.88	**0.007**
		Cover of Tall Productive Forages	0.45	0.63	**0.021**
		Weedy & Ruderal Cover	0.44	0.38	0.098

A = Probability of occurring, B = Fidelity

Chapter 5

Linking pasture seed bank composition and legume recovery potential to management history

5.1 Abstract

Northern temperate grasslands and their corresponding persistent seed banks are influenced by producer management and disturbance legacies. This study examined the seed bank composition across 102 pastures in north central Alberta, and interpreted these data using surveys of recent and historical pasture management. Seed banks were strongly shaped by legacy effects of cultivation and fire, with additional responses to grazing intensity and timing, herbicide use, and manure spreading, among others. Seed banks were dominated by introduced ruderal forbs, followed by introduced (seeded forages), with relatively little representation of native vegetation. Higher densities of introduced ruderal forbs occurred in pastures more recently cultivated, subject to greater livestock stocking, particularly during the growing season, or exposed to supplemental feeding and manuring. A history of cultivation negatively impacted native species in the seed bank. Seed banks abundant in desirable forages (including seeded forage grasses) were associated with higher rangeland health scores. Legumes like clovers formed a persistent seed bank, and overall legume densities were not significantly reduced by herbicide use. Overall, this study indicates that management practices have a strong influence on seed bank composition, and in turn, may help explain long-term vegetation dynamics in northern temperate pastures.

5.2 Introduction

Seed banks are an important component of grasslands, facilitating the entry of individuals spatially and temporally into established communities. In doing so, healthy seed banks are valuable for maintaining grassland productivity, rangeland health, and associated biodiversity (Zhan et al. 2007). As the seed bank often has unique floristic diversity that is dissimilar from aboveground vegetation (Eriksson and Eriksson 1997; Tracey and Sanderson 2000; Williams 1984; Hopfensperger 2007), it partly serves as a reservoir of desirable species (i.e. forage grasses, legumes, etc.), as well as the potential for the establishment of weedy plants. Seed banks often contain an abundance of dormant ruderal species that include introduced or invasive species, as well as noxious weeds (D'Antonio and Meyerson 2002; Eschtruth and Battles 2009).

Seed bank composition is shaped by historical and contemporary disturbances, such as severe retrogression (i.e. fire or cultivation), modified seed input (i.e. mowing, manure addition, etc.), or reduced reproduction of late seral species (i.e. grazing) (Kinucan and Smeins 1992; Wellstein et al. 2007). Existing grasslands are often managed primarily for livestock grazing, which modifies seed banks through the timing and intensity of defoliation (Kinucan and Smeins 1992). Removal of plant biomass and associated flowering parts directly reduces seed production (Sanderson et al. 2007). Additionally, grazing modifies microenvironment at the soil surface by altering litter accumulation, soil compaction, intactness of biological soil crusts, and the formation of bare ground, all of which can change recruitment from the seed bank (Clements et al. 2007; Li et al, 2005; Willms and Quinton 1995).

Disturbances that cause rapid and marked retrogression (i.e. fire and cultivation) together with chronic perturbations (i.e. long term heavy grazing) can both degrade rangeland condition, in turn, affecting seed banks. Previous research has identified that grazing intensity (Clements et al. 2007; Jacquemyn et al. 2011; Ma et al. 2010; Sanderson et al. 2007; Wellstein et al., 2007; Willms and Quinton 1995; Zhan et al. 2007), grazing systems (Kinucan and Smeins 1992), disturbance intensity (Renne and Tracy 2007), manure application (López-Mariño et al. 2000), herbicide exposure (Mayor and Dessaint 1998), previous cultivation (Levassor et al. 1990; Sanderson et al. 2007), and repeated tillage (Goslee et al. 2009), can all influence grassland seed banks. In general, both the richness and diversity of plant communities benefit from low to moderate levels of grazing (Milchunas et al. 1988), in turn, likely resulting in more diverse seed inputs to the soil. However, disturbances like cultivation and manure application are often associated with increases in annual plant species within the seed bank (López-Mariño et al. 2000; Sanderson et al. 2007). Grasses that propagate vegetatively, particularly in the case of long-lived perennials, are known to be relatively rare in the seed bank (Coffin and Laurenroth 1989; Ma et al. 2010; Sanderson et al. 2014), with the exception of the invasive grass Kentucky bluegrass (*Poa* *pratensis*) (Ren and Bai 2016a; Sanderson et al. 2007; Tracy and Sanderson 2000). The seed density of grasses and other forages typically increases when grazing is deferred or removed (Kinucan and Smeins 1992; Tracey and Sanderson 2000), likely due to improved grass phenological development through seed production and dispersal.

In North Central Alberta, plains rough fescue (*Festuca hallii*) grassland has been markedly altered through cultivation and fire, both of which were used to improve land for annual crop production and livestock grazing following European settlement (Bailey et al. 2010; Coupland and Brayshaw 1953; McCartney 1993). Further modification of the existing agricultural landscape is driven by agricultural commodity prices and soil quality; although rates vary, the prairie pothole region loses about 1.33% of remaining native grassland annually (Rashford et al. 2011). Alberta and Manitoba contain the greatest proportion of intact fescue grassland at 11.5 to 12%, while as little as 5.9% remains in Saskatchewan (Gauthier and Wiken 2003). In many cases arable land in the Parkland has been reseeded to introduced forage grasses (colloquially called tame pasture) to support livestock (mostly cow-calf) operations, with pastures in peri-urban areas supporting horses or other companion animals as well. Native grassland patches that remain are often semi-natural, containing introduced forages like Kentucky bluegrass (*Poa pratensis*) and smooth brome (*Bromus inermis*), both of which invade and increase under grazing pressure and favorable moisture (De Keyser et al. 2015). Demand for arable land has also extended into the neighboring boreal natural subregions (Dry Mixedwood and Central Mixedwood) (Young et al. 2006), likely causing a similar pattern of impacts on vegetation, including any remaining pockets of grassland. Legumes are an important component of pastures due to their ability to fix nitrogen (N) and reduce input costs, as well as increase forage productivity and quality, particularly crude protein (Ledgard and Steele, 1992). Within the Parkland, legumes such as white clover (*Trifolium repens*) and alfalfa (*Medicago sativa*) are particularly widespread in seeded pasture, and white clover has become naturalized in the region (Barret and Silander 1992). Native legumes like American vetch (*Vicia americana*) and peavines (*Lathyrus* spp.) can also occur, even in tame pasture. Land use management practices that reduce legume abundance are likely to decrease production efficiency. Where noxious broadleaf weeds are common in

Alberta, land owners are mandated to control weeds through the *Weed Control Act* (Province of Alberta, 2010). Herbicides can be an effective tool for reducing weeds (Grekul et al. 2005; Grekul and Bork, 2007), restoring forage production (Bork et al. 2007), and meeting local municipal guidelines for weed control. However, one undesirable side effect of herbicides is that those with the greatest efficacy on perennial weeds are deleterious to legumes, frequently eliminating them from the forage sward (Grekul and Bork 2007; Bork et al. 2007). Aside from pasture plow-down and legume reseeding, a costly process that temporarily removes land from production, legume re-establishment must occur through other means. For example, volunteer legume re-establishment could occur from the existing seed bank, or pasture over-seeding could be used to reintroduce legumes. Both these processes may be negatively impacted by the ongoing presence of herbicide residue (Miller et al. 2015). Palatable legumes like alfalfa could also decrease with grazing pressure due to repeated selection (Smith et al. 1988), particularly under continuous grazing (Walton et al. 1981). Legume recruitment can be limited by their seed ecology, with a thick, indurate seed coat limiting contributing to dormancy (Baskin et al. 2000; Tracy and Sanderson 2000). As a result, the potential for legume re-establishment from the seed bank is of significant interest in this research, in addition to the presence of other desirable forages and problematic weeds.

Although several studies have been done on the seed banks of pasture, including native grasslands, in western Canada (Clements et al. 2007; Harker et al. 2000; Johnston et al. 1969; Otfinowski et al. 2008; Ren and Bai 2017; Ren and Bai 2016a; Ren and Bai 2016b; Ren and Bai 2007; Romo and Gross 2011; White et al. 2012; Willms and Quinton 1995), these have generally been limited to a small number of sites at select locations. In addition, many of these studies focus on only one or two aspects of management, greatly limiting them in scope and their ability to explain seed bank characteristics. Consequently, the full extent to which seed bank composition of northern temperate pastures is altered by management remains poorly understood, including how it is shaped by both contemporary and historical management.

In this study, seed banks were sampled across a large area of north central Alberta to assess the role of environment (soils) and divergent management history (grazing and other disturbances) in altering seed bank composition. Our study is unique because it allows direct linkage of many aspects of management (herbivore type, grazing system, intensity and associated range health, and inputs such as herbicide, fertilizer, manure, fire, etc.) with the seed bank. We predict that seed bank density, composition, diversity and abundance, as tested by plant recruitment from the soil seed bank of pastures, will be associated with past and current management factors, and therefore be a product of past perturbation events combined with current management regimes, with increasing disturbance leading to more disturbance tolerant ruderal plant species, and a decline in desirable forages, including legumes (Willms and Quinton 1995). Our specific objectives were to: 1) evaluate the relative importance of biophysical and management factors in regulating pasture seed bank composition and diversity, 2) relate rangeland health to belowground seed bank composition, and 3) quantify the similarity between aboveground (foliar) and belowground (seed bank) communities, including how this relationship varies in relation to environment and management history. Ultimately, this information should help identify the suite of management factors and environmental indicators that promote healthy seed banks (abundant in forages, including legumes) and factors associated with undesirable seed banks such as ruderal, unpalatable or noxious weeds.

5.3 Methods

5.3.1 Study Site Selection and Vegetation/Soil Assessment

We surveyed a total of 102 pastures during 2012 (N=44) and 2013 (N=58) between May 24 and July 6, distributed across 4 counties (Leduc, Parkland, Strathcona and Sturgeon County) immediately surrounding the city of Edmonton, Alberta (Figure 3.1). This sampling area is located in north central Alberta's Central Parkland natural subregion, which is characterized by Black Chernozemic soils (i.e. organic matter of 4-10%), and receives 445 mm of precipitation annually, with 77% falling during the growing season (April through September) (Fig. 3.2). About half the pastures sampled were in the Central Parkland (N=50), while the remainder occurred within the neighboring boreal natural subregions: Dry Mixedwood (N=50) and Central Mixedwood (N=2). Although precipitation levels are similar, soils in the latter regions are lower in organic matter, resulting in soils varying from Eluviated Black Chernozems to

Gray Luvisols. The previously cultivated and seeded pastures within the boreal zone resemble Parkland pastures in composition (Donkor et al. 2002). The large sample size ensured a wide range of pasture types were represented, including old growth pastures (often *Trifolium* spp. dominated) and high-performance pastures containing *Medicago* spp., with a corresponding wide range of management activities.

Pastures were selected at random by driving rural roads and approaching landowners with suitable landscapes. Pastures were separated by at least 800 m and had to be large enough (≥ 4 ha) to accommodate a 260 m long transect (Fig. 5.1), with suitable buffer zones from wetlands, forests, and fence lines (outlined in Chapter 4.3.3 describing the plant community survey). If a producer owned or rented multiple pastures, duplicate pastures were only sampled if they were separated spatially, although select exceptions (N=2) were made if management was divergent (i.e. a previous cultivated vs. non-cultivated field; or pastures seeded with different forage mixtures). Acquisition of sites was further constrained by the willingness of landowners to grant access to their land. Finally, during the selection of grasslands, efforts were made to avoid hay fields. Further information on pasture sampling is provided in Chapter 3.3.1.

5.3.2 Determining Producer Management

Producer management information was acquired for all 102 pastures through a retrospective, in-person interview, described in detail in Chapter 3.3.2. The interview (see Appendix 3.1), approved by the Research Ethics Office at the University of Alberta, was designed to identify all historical and current land use practices on each pasture. Surveys were intended to identify all key management activities that may influence the soil, plant community and associated seed bank composition. We identified whether the pasture had been previously cultivated, and if so, the date of last cultivation if known (the latter was often unknown for grazing leases or when the land was cultivated before the occupant's possession). Other data on management collected included grazing history (number of animals, type of herbivore and timing of use), whether the land had been previously seeded to introduced forages, fertilized (chemical or manure), or sprayed with herbicide in the last 3 years, and whether the pasture had been otherwise disturbed (burned, pest control, oil and gas disturbance, etc.).

5.3.3 Plant Community, Rangeland Health, and Soil Properties

Plant community composition was assessed at 9 equidistant locations along the W-shaped transect using a 50 x 50 cm (0.25 m^2) quadrat (Fig. 5.1). Foliar cover by individual plant species, together with ground cover (litter, bare soil, manure, rock, moss and lichen), was visually estimated to the nearest 1 %; cover <1% was recorded as trace (0.1%). In addition, litter depth was measured at 5 points in each frame (4 corners and centre). Rangeland health was assessed using the *Tame Pasture Assessment Form* developed by Alberta Environment and Parks, formerly Alberta Environment and Sustainable Resource Development (Adams et al. 2005) and was described in Chapter 3.3.3. For reference the RHA form used in the assessment of pastures is provided in Appendix A.2, and resultant RHA scores for all pastures summarized in Chapter 3. When classifying pastures as tame or modified-tame, we classified pastures as modified-tame based on the guideline that specified pasture composition had to be comprised of more than 50% native cover. This was modified further where native grass cover was present, especially of plains rough fescue (*F. hallii*) or intermediate oatgrass (*Danthonia intermedia*), or high native forb cover (which was usually over 50%), each of which led to assignment of plant communities to the modified-tame category.

One additional amendment made while assessing range health was to include all seeded forages like creeping red fescue (*Festuca rubra*), Kentucky bluegrass (*Poa pratensis*), and white clover (*Trifolium repens*) as desirable forages, as these species contribute to favorable scoring in sections 1A, 2.1, and 2.2 (see Appendix A.2). This was done as many species described as grazing induced forages were seeded by producers and therefore should not be discounted in the RHA. Additionally, native grasses were included in the forage cover for pastures classified as tame, because although guidelines suggest otherwise, native species can still contribute to the agro-ecological function of these pastures. Under the tame pasture assessment however, native forbs contributed negatively to the RHA score.

Soil cores (n = 10) were plunged randomly across each field, and after the surface LFH was removed, the 0-15 cm mineral soil layers combined to produce one composite sample for each field. Samples were dried at 55ºC, run through a 2 mm sieve, and later assessed for soil physical properties, including % organic matter (OM), total nitrogen (N), total carbon (C), pH, electrical conductivity (EC; µS/m), and texture (% sand, silt and clay). Levels of OM were quantified by burning 10 g of soil in a furnace at 450ºC for 4 hours and measuring the subsequent mass loss. EC and pH were measured in a soil solution that was one-part soil and two-parts water. Soils were shaken for at least 30 minutes before measuring pH, and the soil solution filtered before measuring EC. Total carbon and nitrogen were measured using a LECO TruSpec CN elemental analyzer (LECOCorporation, St. Joseph, MI, USA), with samples that were ground to a powder with a ball mill to ~ 0.1 mm (Retsch MM400 Mixer Mill, Retsch, Haan, Germany) and fumigated with HCl beforehand to remove inorganic C present as carbonate in alkaline soils (note that all soils were treated similarly). Soils from north Central Alberta typically had OM exceeding 5% for the majority of sites, thus all soil samples were pre-treated before texturing. OM was removed by applying small volumes of hydrogen peroxide to ~60 g of soil until soils achieved a color change and the reaction ceased (Lavkulich and Wiens 1970; Mikutta et al. 2005). Texturing was then performed on pre-treated soils using the hydrometer method (Bouyoucos 1927), where 40 g of soil and 4 g of sodium hexametaphosphate (Calgon) were suspend in 1 L sedimentation tubes, and the proportion sand, silt and clay subsequently quantified. In 2013, soil compaction was measured at 45 sites using a soil surface penetrometer.

5.3.4 Seed Bank Sampling

The soil seed bank was sampled in each pasture between May 24 and July 6 of either 2012 or 2013. This sampling window coincided with the period prior to the majority of current year vegetation casting seed (particularly weedy annuals and biennials) and was intended to capture the density and diversity of seeds in the persistent seed bank (i.e. those seeds remaining after the winter dormant season). We noted that dandelion (*Taraxacum officinale*), trembling aspen (*Populus tremuloides*), and willows (*Salix* spp.) went into seed early in the year, around early to mid-June. Grasses did not produce seed by the end of the sampling window.

To initiate sampling, a randomly selected point in the pasture was located that met our criteria, and was relatively uniform in ecosite conditions (aspect, slope, elevation, drainage, soils, etc.), and remained distant from disturbances (roads, well sites, feeding areas, etc.) in a representative area of the pasture. During sampling, areas that may have been under the influence of edge effects such as pasture margins (<10 m from fences), wetlands (<30 m), and areas influenced by nearby forest (<10 m) were avoided. From that point a 260 m long 'W-transect' was formed (Fig. 5.1), as adapted from Thomas (1985). Along each of the four linear sides of the 'W', soil cores were taken at 5 m intervals, totaling 53 cores. The soil surface remained non-agitated before coring (i.e. litter was not removed) in order to avoid loss of seed. Since soil cores were considered subsamples of our experimental units (i.e. pastures), seed bank cores were bulked in plastic freezer bags. Soil seed bank samples were promptly frozen until further processed in the greenhouse. To assess the spatial heterogeneity of seed banks, we kept the 53 cores separate from one another for a subset of 11 of the 102 pastures; these cores were subsequently observed individually for seedling emergence. For these individual cores, relative elevation in the landscape was recorded as either 'upland', 'mid-slope', 'lowland', or 'depression' (mesic patches with hydrophytic vegetation) and aspect was also recorded (i.e. north vs south-facing slopes). Our relatively high sampling intensity of 53 cores per pasture demonstrated a reduction in the standard error for both seed bank richness (Figure C.1.1) and density (seed abundance) (Figure C.1.2) of seeds recruited.

Finally, during the 2013 sampling period, where producers indicated that they had spread manure on their pasture (n = 8), the manure pile within their winter feeding area was haphazardlysampled by hand, filling a 3 L bag, for subsequent testing of germinable seeds. These results are presented in Appendix C.5.

5.3.5 Characterising the Germinal Seed Bank

Shortly after removing samples from the field, they were placed in cold storage (below 0^{o}C) to prevent pre-mature germination. This period of freezing temperatures lasted a minimum of seven days and provided cold stratification to improve germination of persistent seed (Acharya 2006; Baskin et al. 2000). After thawing, roots, rocks and rhizomes were removed, and trays (28 cm x 54 cm in size, and 6 cm deep) with holes for drainage prepared to assess seed bank composition. Trays were first filled with 2 cm of sand sterilized in an autoclave to provide additional rooting depth. Germinable seed bank samples (soil or manure) were then spread out on trays (one tray per pasture) to a depth of about 2 cm. To verify that the sand was sterilized (and free of germinable seed) four replicate trays of sand without pasture topsoil were observed for germination over the trial, from which no seedlings emerged. All trays were watered as required to prevent desiccation and promote seedling emergence.

Plant species composition of the seed bank was identified by allowing seeds to germinate under greenhouse conditions. All trays started and ended their germination period at the same time (from each year of sampling) and were grown under similar conditions (16:8 hr day and night; 20ºC). Soil was stirred every 3 months to encourage further germination after germination had slowed. Seedlings were counted as they germinated and removed after identification. Unidentifiable seedlings were transplanted into pots and grown out until mature enough for identification, which occurred to the species level following the taxonomy and nomenclature of *Flora of Alberta* (Moss 2010). Nomenclature was verified using the USDA Plants Database to ensure the most accurate description of species. Each greenhouse trial was terminated 1 year after the start date.

Seed abundance (by species, and functional group) was converted to the number of seeds/m^2 (seed density) based on the collective area of soil sampled (i.e. 53 cores = 0.0604 m^2 area). For soil cores stored individually, each core was prepared individually and placed in square 5 cm x 5 cm deep pots over 2 cm of sterilized sand. For analysis of pasture seed bank characteristics, the germination from individual cores was pooled. Seed densities were totaled for both primary vegetation categories (introduced, native, broadleaf and graminoid) and secondary plant groupings [legumes (including both native and tame species), noxious weeds, introduced ruderal forbs, seeded/introduced grasses, ruderal grasses, native ruderal forbs, native perennial forbs, native perennial grasses, graminoids (sedges and rushes), and woody species]. Similarity in seed bank richness was compared to the aboveground plant community using the Sørenson's index of community similarity, as follows:

$$S=2(A\cap B)/A+B$$

A similar procedure was followed for all manure samples removed from manure piles. Trays were lined with 2 cm of sterilized sand, and then 2 L of compact manure was measured out and distributed across the tray. Seedling emergence was then assessed similar to that from the soil samples, with agitation used periodically to stimulate germination.

5.4 Statistical Analysis

5.4.1 Linking Management to Seed Bank Characteristics

Seed bank composition, measured as the seed density (seeds/m^2) of all primary and secondary plant functional groups listed above, indices of total species richness and diversity, and similarity to the aboveground vegetation, were analyzed with both univariate and multivariate statistical methods in R software (R Core Team 2017). Seed densities and indices from all sites were initially tested for normality, with residuals examined using the Kolmogorov-Smirnov test using the *lillie.test* function in the *nortest* package in R (Gross and Ligges 2015), and homogeneity of variances using Levene's test. The only variables that met assumptions without transformation included similarity, richness, and Simpson's diversity. A square root transform (total seed density, total introduced, and Pielou's evenness) and logarithmic transform (total graminoids, total broadleaf, total native, native ruderal forbs, introduced ruderal forbs, seeded/introduced grasses, and ruderal grasses) were used for many variables. A box-cox transformation was used for positively skewed data, while a x^2 transform was used for data with a negative skew (Shannon's diversity), prior to ANOVA.

Shifts in aggregate seed bank characteristics within vegetation groups relative to management factors were tested using univariate analysis of variance (ANOVA), and in the case of the density of legumes, noxious weeds, perennial native forbs, perennial native grasses, graminoids, and woody species, which could not be normalized, a Kruskal-Wallis test was done for non-parametric data using the *agricolae* package (De Mendiburu 2017). One-way ANOVAs were done using Type III SS (sums of squares) and LS (least-squared) means because we had unequal sample sizes among levels of each management factor. LS means and contrasts were derived from the *lsmeans* package (Lenth 2016). Post-hoc contrasts were Bonferroni corrected when three or more management factors were compared. Non-normal data were assessed with a Kruskal-Wallis test in R with *kruskal* in the *agricolae* package, which also provided Bonferroni adjusted mean ranks for contrasts.

To assess species level seed bank compositional differences, all 102 pastures were analysed for the impact of management factors on seed bank composition using permutational multivariate analysis of variance (perMANOVA) with a Bray-Curtis distance and the *adonis* function in R package *vegan*, set to run 999 permutations (Oksanen et al. 2017). Due to our unbalanced experimental design of management factors (i.e. it was impossible to know survey responses in advance of the producer interview) and differences in multivariate spread among factors (Anderson 2005), we tested each management factor individually – comparable to a one-way ANOVA. Once significant primary management factors and interactions were identified, contrasts were performed on the inherent treatment levels within management factors (e.g. cultivated vs not cultivated vs pastures with unknown cultivation history). When testing for differences in seed bank composition among pastures where animals were given supplemental feed, we only analysed pastures sampled in 2013 due to the omission of this question from surveys the previous year (N=58). Tests of perMANOVA were followed by an indicator species analysis (ISA) on the species matrix and plant species functional groups. All ISA were run in R using the *indicspecies* package's *multipatt* function with 999 permutations (De Caceres and Legendre 2009). When testing for indicator species arising from the supplemental feeding of hay, data were again subset for pastures sampled in 2013.

Non-linear multidimensional scaling (NMDS) ordination was used to graphically explain the relationships between seed bank species composition, pasture management factors obtained from the producer surveys, ancillary environmental (i.e. soil) variables, and rangeland health. Ordination was performed in R software with the *metaMDS* function in the *vegan* package using the Bray-Curtis distance measure and 999 permutations (Oksanen et al. 2017). Given the large number of variables analysed, assessment of ordinations was limited to the first two dimensions. For the resultant NMDS, individual management factors (as centroids), seed bank characteristics, plant species, rangeland health, and ancillary environmental variables were tested for significance using the function *envfit* in R's *vegan* package (Oksanen et al. 2017); only significant factors ($P < 0.05$) were plotted. Additional panes of the same ordination were included for significant management factors (centroids), which also included their respective indicator species.

5.4.2 Seed Bank Relationship to the Plant Community and Environment

Correlation matrices were used to assess the relationship between seed bank characteristics, vegetation cover, and soil properties. Spearman's correlations were run in R software using the package *corrplot*, with only those significant ($P < 0.05$) reported. Seed bank and plant community composition were also linked using a canonical correspondence analysis (CCA), where variance in seed bank composition within the resulting ordination was constrained by the overlying plant community composition. The CCA model was generated in R with the *vegan* package's *cca* function, with constraining variables (plant community species) selected using a stepwise selection ($P < 0.05$) (Oksanen et al. 2017). To simplify the model and reduce run time, aboveground plant species that occurred in three or less pastures were considered rare and excluded for the CCA.

The composition of individual seed bank cores among variable topographic positions and aspects were described using ISA and perMANOVA. Cores that produced no seedlings were included as a dummy variable in the matrix.

5.4.3 Rangeland Health Assessment Criteria

Seed bank composition, seed densities, and all complex indices were tested directly against rangeland health criteria using univariate tests (ANOVA and Kruskal-Wallis) and perMANOVA. ISA (indicator species analysis) was used to identify specific plant species in the seed bank responsive to each question in the RH assessment, total scores for functional groups were also analysed with ISA. Results were included in Appendix C.7 with a brief discussion, some key results from this section may be pulled into the discussion to support discussions around rangeland health and community shifts with management. Finally, regression analysis was used to relate seed density to total RHA scores using GLMs (generalized linear models) set to a Poisson distribution, which is suited to count data.

5.4.4 Stockpiled Manure

Stockpiled manure samples were examined for seed bank composition from a total of 8 sites in 2013. These data were assessed with NMDS using a similar procedure outlined for the soil seed bank. Due to the limited information on manure pile history (i.e. age, salinity, etc.), we were limited in our ability to use other variables or perMANOVA to further explain manure seed bank composition. Seed bank densities from various functional groups (legumes, graminoids, noxious weeds, and weedy forbs) were directly compared from manure samples using a Kruskal-Wallis test ($P < 0.05$).

5.5 Results

5.5.1 Seed Bank Composition

A total of 165 different plant species emerged from the soil seed bank. Aboveground surveys of vegetation revealed 159 species, with 100 common to both and a mean similarity of 34.0% (Sørenson's similarity index). Seed banks contained an average of 5976 ±3756 (1 SD) seeds/m^2 and ranged from 810 to 17826 seeds/m^2. Remaining dissimilarity was accounted for by abundant introduced ruderal forbs in the seed bank (Fig. 5.2), while aboveground vegetation was mostly productive, perennial forage grasses (see Fig. 4.2). Seeds of woody species and native perennial grasses were particularly uncommon (Fig. 5.2). Legume seeds were present in 80.4% of pastures, with clovers like *Trifolium repens* and *T. hybridum* being the most frequent in the seed bank (Table 5.1). Species common within the seed bank were often poorly represented within the overlying vegetation and vice versa (Table 5.1). Native and introduced ruderal species that were rare aboveground, but common and dominating the seed bank like stinkweed (*Thlaspi arvense*; rank 52 above and 3 belowground), lambsquarters (*Chenopodium album*; rank 44 above and 4 belowground), marsh cudweed (*Gnaphalium uliginosum*; rank 87 aboveground and 5 belowground) likely formed persistent seed banks. Two species that occured with high rank dominance in both the seed bank and aboveground, were dandelion (*Taraxacum officinale*) and Kentucky bluegrass (*Poa pratensis*). Species much more common in the seedbank than aboveground were generally forbs such as lambsquarters, plantain, and stinkweed (Table 5.1).

5.5.2. Seed Bank Responses to Management

Both seed density and seed bank composition were affected by aspects of pasture management and disturbance history. The NMDS ordination of seed bank composition (distance = Bray-Curtis, dimensions = 2, stress = 0.31) illustrated important gradients in seed bank communities, that in turn, related to management (Figure 5.3). Across all pastures cultivation history, evidence of fire (based on charcoal presence), hay feeding, herbivore type, manure spreading, and cutting, all had an impact on seed bank composition in the 2-dimensional NMDS solution ($P \leq 0.05$), with cultivation history explaining the most variation ($R^2 = 0.12$; Table C.2.1). These management factors are decomposed further in Fig. 5.4 using the same scores from Fig. 5.3 and include their indicator species (Table 5.4). Sandy soils were associated with abundant native grasses and forbs. Vectors indicative of high soil fertility (organic matter, total carbon, total nitrogen) and salinity (EC) were associated with pastures of greater species richness, seed bank diversity, and native ruderal forb seed density. Vectors for similarity, Pielou's evenness, pasture age, litter cover, high RH scores for plant community composition (forage spp. shift), and proportion of silt were associated seeded forage species like common timothy (*Phleum pratense*), smooth brome (*Bromus inermis*), and *Poa pratensis*. Pastures with high densities of forage grasses were associated with the presence of charcoal in the topsoil and the absence of livestock. High RH scores for woody cover (indicating the lack of shrub encroachment in tame pasture) were associated with alternative livestock like sheep and alpacas and corresponded with weedier seed banks dominated by introduced ruderal forbs. These weedy seed banks were also associated with the following management factors: cultivation history confirmed and unknown, manure spreading, and feeding hay in pasture. The influence of specific management factors on seed banks is examined in detail in the following sections.

5.5.3. Ownership

Across all pastures, rented and owned land had similar seed bank composition ($P = 0.102$; Table 5.2, and see Table 5.5), with limited differences in seed densities among plant functional groups. Rented pastures were associated with higher densities of native perennial grasses (functional group ISA, $P = 0.007$; Table 5.5; Table 5.8 and 5.9) in the seed bank. The ISA analysis revealed that rented pasture had a greater abundance of slender wheatgrass in the seed bank, together with the noxious weed Canada thistle (Table C.6.1).

5.5.4. Cultivation

At least 5 pastures recorded as previously cultivated were clustered with communities classified as non-cultivated (Figure 5.3 and 5.4) in the ordination. Cultivation history affected seed bank composition ($P = 0.025$; Table 5.2). Pastures with an unknown and a known cultivation history were similar in composition ($P = 0.179$), while fields reported as never having been cultivated were unique from the former categories ($P < 0.028$; Table 5.3). Plant species in the seed bank indicative of the absence of cultivation included a variety of native forbs, grasses, and graminoids, such as common yarrow (*Achillea millefolium*) and harebell (*Campanula rotundifolia*) ($P < 0.01$; Table 5.4), with many other species (primarily perennial forbs) associated as well [e.g., fringed sage (*Artemisia frigida*), slender blue beard tongue (*Penstemon procerus*), and Pennsylvania cinquefoil (*Potentilla pensylvanica*) (Table C.6.1.1]. This trend was also supported by an indicator analysis of plant functional groups in the seed bank (Table 5.5); functional groups associated with pastures with an unknown cultivation history included introduced ruderal forbs, all introduced plant species, native ruderal forbs, and total forbs ($P \leq 0.059$; Table 5.5).

Cultivated fields generally had seed banks with more introduced ruderal forbs compared to non-cultivated fields (Tables 5.6, 5.7). However, pastures with an unknown cultivation history had more introduced ruderal forbs and greater total introduced seeds relative to both other groups. Native seed densities were marginally reduced with cultivation (Tables 5.6, 5.7), while native perennial forb seed densities were markedly reduced ($P = 0.002$; Table 5.8 and 5.9). Cultivation history did not influence indices of seed bank diversity, richness, and evenness (Table 5.10). The approximate year of last cultivation was known for 71 pastures in this study. Through NMDS ordination (distance = Bray-Curtis, dimensions = 2, stress = 0.30) we found that pasture age effectively described seed bank community gradients in these pastures (Figure 5.5). Older pastures generally had an abundance of forage grasses, primarily Kentucky bluegrass (*Poa pratensis*), but also had a trend for more native perennial forb and grass seeds (Figure 5.5). In contrast, younger pastures had greater seed bank richness, diversity, and abundance of introduced plant species, all forbs, and total seed density.

5.5.5. Grazing History

Seed bank composition was affected by the timing of grazing (P = 0.048; Table 5.2), with pastures grazed all winter differing from those grazed year-round and throughout the growing season (P ≤ 0.069) (Table 5.3). When grazing systems and timing were combined, differences in seed bank composition among pastures were more apparent (P = 0.032; Table 5.2); in particular, seed banks differed in composition between continuously grazed pastures used year-round and only during the growing season (P = 0.022; Table 5.3). Additionally, the seed bank of pastures rotationally grazed during the growing season differed from pastures used over winter (P = 0.044; Table 5.3). Pastures grazed only during winter had seed banks abundant in tame forages, including *Festuca rubra*, *Phleum pratense*, *Astragalus cicer*, and *Festuca ovina var. arundinacea* (P < 0.05) (Tables 5.4, C.6.1). In contrast, abandoned pastures, grazed solely by free-ranging wildlife, contained the native grass *Danthonia intermedia*, together with legumes such as *Medicago sativa* and *Trifolium pratense* in the seed bank (Table 5.4). Total graminoid seed density was generally greatest for those pastures winter grazed or not receiving any livestock use (Tables 5.5, 5.6, 5.7), and remained lowest in areas with year-round continuous grazing (Table 5.7). Winter grazing generally favored the accumulation of legumes and seeded forage grasses in the seed bank (Table 5.9). Areas winter grazed or remaining non-grazed had lower Simpson's diversity within the seed bank (Tables 5.10, 5.11). Grazing intensity, as quantified during the rangeland health assessment (RHA), was not associated with a significant difference in seed bank composition (P = 0.422; Table 5.2). Legume seed bank density was affected by grazing intensity (categorized during the RHA) (P = 0.04; Table 5.8), demonstrating a bi-modal response. Legume seed density peaked in the absence of grazing and at the highest intensity, legume density was lowest at a low-moderate grazing intensity (Table 5.9). Grazing intensity also marginally influenced noxious weed seed density (P = 0.084; Table 5.8), with weed density increasing under increasing grazing pressure, as reflected by lower RHA scores (Table 5.9).

The type of herbivore grazed on pasture did not affect seed bank composition (P = 0.291; Table 5.2), nor seed densities of plant functional groups. A small number of ruderal indicator species were detected for pastures grazed by alternative livestock (Tables 5.4, C6.1) including the noxious weed perennial sow thistle (*Sonchus arvensis*). Both Shannon's and Simpson's diversity in the seed bank were altered by herbivore type (P ≤ 0.016; Table 5.10), being greater in pastures grazed by mixed livestock herds, usually both cattle and horses, relative to pastures lacking livestock (Table 5.11). Finally, while pocket gophers and ground squirrels were observed to be common pests of the pastures surveyed, their presence was not associated with significant differences in seed banks (P = 0.403; Table 5.2).

Rangeland health score, which were linked to timing of grazing, grazing systems x timing, and grazing intensity in Chapter 4, were associated with shifts in seed densities and similarity to the aboveground plant community. Across all species, total forb density declined and graminoid seed density increased with greater range health scores (P < 0.001; Figure 5.7). Additionally, similarity between the seed bank and aboveground vegetation declined with higher range health (Figure 5.8).

5.5.6. Feeding Hay on Pasture

Where animals were fed supplemental hay on pasture, the seed bank differed from those pastures experiencing grazing alone (P = 0.016; Table 5.2). Positive indicator species for pastures where animals were fed hay included slough grass (*Bekmannia syzigachne*), lambsquarters (*Chenopodium album*), green foxtail (*Setaria viridis*), and stinging nettle (*Urtica dioica*) (P < 0.027; Tables 5.4, C.6.1).

Plant functional groups in the seed bank of pastures associated with supplemental feeding were introduced ruderal forbs, total introduced species, and total forbs, while graminoids and native species were associated with pastures where no feeding occurred (P ≤ 0.084; Table 5.5). There were also trends for lower seed density of graminoids (sedges and rushes) where hay was fed on pasture (P = 0.065; Tables 5.8, 5.9), while woody species were greater (P < 0.01; Table 5.8).

5.5.7. Herbicide

Herbicide application within the last 3 years was associated with a shift in seed bank composition (P = 0.032; Table 5.2). Herbicide treated pastures had marginal reductions in total forbs (P = 0.092) and total native seeds (P = 0.057) (Tables 5.6, 5.7), as well as fewer native ruderal forbs (P = 0.082; Tables 5.8, 5.9). There were no plant functional groups in the seed bank indicative of herbicide treatment, with only stinging nettle (*Urtica dioica*) indicative of sprayed pastures (P = 0.049) and tickle hair grass (*Agrostis scabra*) (P = 0.04) indicative of non-sprayed pastures (Appendix C.6.1). Herbicide treatment had a strong effect on seed bank richness (P = 0.011) and both Shannon's and Simpson's diversity (P < 0.001) (Table 5.10), all of which demonstrated a loss in diversity with herbicide exposure (Table 5.11). Of note is that total legume seed density was not affected by herbicide application (P = 0.155).

5.5.8. Fertilizer and Manure

Application of fertilizers had little effect on seed bank composition (P = 0.327; Table 5.2), with the lone indicator species positively associated with fertilization being quackgrass (*Elytrigia repens*) (P = 0.024; Appendix C.6.1). However, seed density responses revealed many functional plant groups had divergent responses to fertilizer. Seed densities of both forbs and native plant species declined by more than 50% under fertilization (P ≤ 0.01; Tables 5.6, 5.7), with native ruderal forbs and native graminoids in particular, both lower in fertilized pastures (P < 0.023; Tables 5.8, 5.9). Legume seed densities in fertilized pastures were much lower at 39.7 (±66.2) seeds/m^2 compared to 168.9 (±20.6) seeds/m^2 (P = 0.032; Tables 5.8 and 5.9), a reduction of 76.5%. A similar reduction in noxious weed seed density was evident under fertilization, where fertilized pastures had 18.5 (±156.3) seeds/m^2 and non-treated pastures had 186.0 (±48.6) seeds/m^2 (P = 0.026; Tables 5.8 and 5.9). Measures of seed bank diversity were not affected (Table 5.10).

Manure application altered seed bank composition (P = 0.037; Table 5.2), with a trend for manured pastures to have reduced total graminoids, grasses and grasslikes (P ≤ 0.079; Tables 5.6, 5.7), increased native ruderal forb seed densities (P = 0.076; Tables 5.8, 5.9), but reductions in native perennial forbs and native perennial grasses (P ≤ 0.041; Tables 5.8, 5.9). Where manure was applied, seed banks were associated with ruderal species like rocky mountain goosefoot (*Chenopodium salinum*) (P = 0.001), common chickweed (*Stellaria media*) (P = 0.021), wormseed wallflower (*Erysimum cheiranthoides*) (P = 0.024), and pineapple weed (*Matricaria matricarioides*) (P = 0.049), along with disturbance adapted grasses like foxtail barley (*Hordeum jubatum*) (P = 0.025) and Canada bluegrass (*Poa compressa*) (P = 0.036) (Appendix C.6.1). Contributions from ruderal species to the seed bank caused a small but significant increase in Simpson's diversity (P = 0.049; Tables 5.10, 5.11).

Stockpiled manure collected from farms (n = 8 piles) contained a germinable seed bank that was dominated by weedy forbs (Figure C.5.1), primarily *Chenopodium* spp. Manure also included the seed of some forage grasses and an early seral sedge (*Carex sychnocephala*), along with all three common clovers (*T. hybridum*, *T. pratense*, and *T. repens*). Noxious weeds were present in trace amounts within manure piles. Manure piles were primarily derived from cattle manure, with a single case of sheep manure. NMDS ordination of stockpiled manure seed banks showed 2 general types of seed bank communities: four sites had manure with greater representation of weedy forbs including noxious weeds, while manure rich with graminoids, legumes, and overall species richness appeared to represent a more desirable seed bank at the other four locations. We were unable to link seed bank composition of stockpiled manure to the age of the manure pile.

5.5.9. Mechanical Pasture Maintenance: Harrowing, Aeration, Swathing/Mowing

Harrowing was not associated with distinct shifts in seed bank composition (P = 0.108; Table 5.2), with no responses in any plant functional groups within the seed bank. A few indicator plant species were evident for harrowing, including those comparable to manure treatment, such as *Chenopodium salinum* and common chickweed (*S. media*), but also included unique species of concern like the noxious weed white cockle (*Silene latifolia* sbsp. *alba*) and pale smartweed (*Polygonum lapathifolium*) (P < 0.05; Appendix C.6.1). Seed bank biodiversity metrics was also unaffected by harrowing.

Mowing or swathing of pastures was not linked to pasture seed bank composition (P = 0.159; Table 5.2). Plant species indicators for pastures that were mowed/swathed included the legumes cicer milkvetch (*Astragalus cicer*) and red clover (*Trifolium pretense*), and the weeds pale smartweed and corn spury (*Spergula arvensis*). Seed bank functional group abundance and diversity indices were again not responsive.

Aeration was reported in only a few pastures (n=4), and thus changes in seed bank in relation to this practice should be interpreted cautiously. Aerated pastures were generally not associated with significant differences in overall seed bank composition ($P = 0.200$; Table 5.2), although those exposed to aeration did contain more legumes ($P = 0.017$, Table 5.5; and $P = 0.036$, Tables 5.8 and 5.9), while non-aerated pastures were associated with more seeds of introduced forage grasses ($P = 0.033$; Table 5.5). Indicator species analysis showed six legume species favored aerated pastures, in addition to some disturbance adapted species that also occurred in manured and harrowed pastures like Canada bluegrass, white cockle, and Polish canola (*Brassica napus*) ($P < 0.05$; Appendix C.6.1). Finally, Kentucky bluegrass was an indicator of pastures that had not been aerated ($P = 0.042$, Table C.6.1), occurring 100% of the time. Both Shannon's and Simpson's diversity were greater within aerated pastures ($P < 0.046$; Tables 5.10, 5.11).

5.5.10. Fire

History of fire influenced seed bank composition, but only based on direct evidence of fire within the soil in the form of charred woody debris ($P = 0.007$), rather than on producer responses to the question of whether fire had occurred ($P = 0.130$; Table 5.2). Moreover, this pattern was paralleled by responses within the seed bank functional group abundances. Pastures containing charcoal had lower densities of total forbs, native plant species and introduced plant species, which combined, translated into an overall reduction in seed density ($P < 0.027$; Tables 5.6, 5.7). Introduced ruderal forbs were 51.9% less abundant (1563 ± 502 seeds/m^2) in pastures with charcoal than those lacking charcoal (3252 ±331 seeds/m^2) ($P = 0.012$; Tables 5.8, 5.9). Native ruderal forbs were similarly reduced ($P = 0.008$; Tables 5.8, 5.9). Seed bank biodiversity was not affected by fire history.

For pastures that had been reported as burned by the manager, native graminoids (like nodding brome - *Bromus anomalus*), native forbs, and the noxious weed perennial sow thistle, were indicators in the seed bank (Tables 5.4, C6.1). In contrast, toad rush (*Juncus bufonius*) occurred in pastures where no fire was reported (Table C.6.1). Pastures containing charcoal in the soil had nodding brome and

Bicknell's cranesbill as indicator species, while lambsquarters and marsh cudweed were indicative of pastures lacking charcoal ($P < 0.05$; Table C.6.1).

5.5.11. Similarity between Seed Banks and Plant Communities

Overall seed densities of individual plant functional groups were correlated with plant community cover aboveground (Figure 5.11). Abundance of native perennial forbs in the seed bank correlated closely with numerous cover variables, including as expected, the cover of native perennial forbs ($r = 0.73$). Native perennial forb seed density also was positively correlated with the cover of native plant species, all graminoids (sedges and rushes), and species rich communities, but was negatively associated with the cover of introduced species. In contrast to native forbs, the density of most other functional groups were not strongly correlated with their expression aboveground. Introduced forage grasses in the seed bank were negatively correlated with aboveground vegetation diversity (Shannon's $r = -0.30$; Simpson's $r = -0.32$) and the cover of introduced ruderal forbs ($r = -0.23$). Introduced forage grass cover was also associated with lower seed bank diversity, driven largely by the under representation of native species. Legume seed bank density was positively associated with legume cover aboveground ($r = 0.26$) but was negatively associated with native cover. Noxious weed seed bank density correlated weakly with most cover variables, but surprisingly not with noxious weed cover ($r = -0.22$). Interestingly, noxious weed seed density was positively correlated with total vegetation diversity aboveground, while noxious cover was negatively associated with total seed bank diversity.

Ordination using CCA revealed that plant community composition aboveground explained 56.0% of the variation expressed in the seed bank (distance = Euclidean, dimensions = 2, axes = 27; Figure 5.6), with 17/27 axes significant ($P < 0.05$); for simplification, only the first 2 axes describing the most variation (24.8%) will be discussed. Seed bank composition diverged in two ways. First, pastures that were never cultivated and not grazed diverged from those exposed to livestock grazing on the primary (first) axis (CCA1), which explained 13.2% of the variance. Low disturbance pastures were associated with a suite of native forbs, native graminoids, and native perennial grasses like plains rough fescue (*Festuca hallii*) and tickle hairgrass (*Agrostis scabra*) (Figure 5.6). Second, pastures with seed banks containing early seral native forbs like long-leaved bluets (*Houstonia longifolia*), fringed sage (*Artemisia frigida*), and pygmy flower (*Androsace septentrionalis*), and which were more likely to be winter grazed by livestock, diverged from those pastures with year-round grazing and more intensive disturbance (e.g. hay feeding) along the second axis (CCA2). The former pastures coincided with seed banks high in early seral native forbs like common yarrow (*Achillea millefolium*) and fringed sage, in combination with Kentucky bluegrass (*Poa pratensis*) and white clover (*Trifolium repens*) (Figure 5.6). In contrast, seed banks with the greatest seed densities were associated with high densities of introduced ruderal forbs, total introduced species, and total forbs (upper right corner of Figure 5.6); seed banks of the latter were associated with stinging nettle (*Urtica dioica*), lambsquarters (*Chenopodium album*), cleavers (*Gallium aparine*), common groundsel (*Senecio vulgaris*), meadow foxtail (*Alopecurus pratensis)*, and perennial ryegrass (*Lollium perenne*). These pastures were also associated with an unknown cultivation history. Pastures that clustered in the center represented the modal seed bank community of cultivated pastures subjected to growing season rotational and continuous grazing.

5.5.12. Seed Bank Characteristics, Ground Cover and Soil Properties

Ground cover variables (i.e. bare ground, litter, etc.) were correlated with various seed bank characteristics. Litter cover was negatively correlated with seed bank diversity (Shannon's $r = -0.26$; Simpson's $r = -0.28$), native ruderal forbs ($r = -0.42$), total broadleaf ($r = -0.28$), introduced ruderal forbs ($r = -0.17$), noxious weeds ($r = -0.13$), and total native seed densities ($r = -0.22$); with only seeded/introduced grasses ($r = 0.23$) and native perennial grasses ($r = 0.13$) responding positively ($P < 0.05$; Figure 5.10). Relationships between seed bank and litter depth were similar, however similarity ($r = -0.19$) was more negative correlated with litter depth ($P < 0.05$; Figure 5.10). Bare ground was positively associated with similarity ($r = 0.16$), ruderal grasses ($r = 0.15$), and native ruderal forb ($r = 0.13$) seed density ($P < 0.05$; Figure 5.10). Where manure was detected there was a positive association with higher densities of legumes ($r = 0.24$), introduced plant species ($r = 0.20$), ruderal grasses ($r = 0.16$), total species richness ($r = 0.11$), and overall seed density ($r = 0.15$), however manure was negatively associated with seed bank evenness ($r = -0.22$) and Simpson's diversity ($r = -0.15$) ($P < 0.05$; Figure 5.10). Ground cover of stems emerging from the soil were positively associated with diversity (Shannon's r = 0.24; Simpson's r = 0.25), native ruderal forbs (r = 0.33), native (r = 0.18), and overall seed density (r = 0.13) ($P < 0.05$; Figure 5.10). Both lichens (r = -0.12) and mosses (r = -0.17) were negatively correlated with similarity, while lichens were associated with greater Shannon's diversity (r = 0.15) ($P < 0.05$; Figure 5.10). When the soil surface was rocky the associated seed bank was more even (r = 0.37) ($P < 0.05$; Figure 5.10).

Soil properties were associated with seed bank characteristics, however these relationships were typically weaker that their relationships with ground cover. Similarity between the seed bank and plant community were negatively correlated with all properties, primarily soil salinity/electrical conductivity (EC) (r = -0.26), except for the proportion of clay (r = 0.03) and silt (r = 0.19) ($P < 0.05$; Figure 5.10). Siltier soils were also positively correlated with legumes (r = 0.25) ($P < 0.05$; Figure 5.10). Soils rich in clay were positively correlated with noxious weed seeds (r = 0.26) and ruderal grasses (r = 0.22) ($P < 0.05$; Figure 5.10). Sandier soils were positively associated with bank diversity (Shannon's r = 0.17; Simpson's r = 0.14) and native perennial grasses (r = 0.22), however legumes were negatively associated with sand (r = -0.22) ($P < 0.05$; Figure 5.10). Carbon-nitrogen (C:N) ratio was positively associated with seed bank diversity (Shannon's r = 0.17; Simpson's r = 0.21), evenness (r = 0.17), and native perennial grasses (r = 0.24); however it was negatively associated with introduced seed (r = -0.20), total graminoids (grasses and grass-likes) (r = -0.25), graminoids (sedges and rushes) (r = -0.24) ($P < 0.05$; Figure 5.10).

5.6 Discussion

This study illustrates that seed banks in northern temperate pastures are significantly altered by ongoing management regimes. Moreover, resultant changes in RHA scores appear to be capable of detecting shifts in seed bank composition. Some studies suggest that seed banks hold a record of the community's 'ecological legacy' (Renne and Tracy 2007) or 'memory' of previous states (Bakker et al. 1996) as they are shaped by their disturbance history. There is evidence for this in the current study, with cultivation and fire being two historical events that sharply altered seed banks. Contemporary management practices that intuitively influence the addition or removal of seeds (Sanderson et al. 2007) through disturbances like the timing of grazing, herbicide application, manure spreading on pasture, and feeding of hay on pasture, were also linked to divergent seed banks. Aspects of management that were linked to plant community shifts did not necessarily affect the seed bank in a similar way (vegetation responses were covered in Chapter 4). Overall, the seed bank was dominated by small-seeded introduced weedy species and had high dissimilarity in species richness (66%) from the aboveground plant community. These small, hard-seeded, abundant species comprised the persistent seed bank of pastures, but were rare in the vegetation (Kinucan and Smeins 1992), and demonstrated linkages to long-term disturbance regimes.

5.6.1 Cultivation History

The majority of pastures sampled in this study had been cultivated and seeded with improved forage mixes (Chapter 3). Cultivation had a profound effect on seed banks (this chapter) and plant communities (Chapter 4), resulting in the loss of numerous native plant species from both above and belowground. In contrast, non-cultivated pastures had lower seed densities than those cultivated but tended to have greater diversity of native plant species. Native graminoids like plains rough fescue (*Festuca hallii*), which tend to decrease with disturbance (McLean and Wikeem 1985), were eliminated from the seed bank of cultivated pastures, and were relatively rare in soil from non-cultivated pastures. A handful of native perennial forb species were retained in the seed bank of previously cultivated pastures and appeared to accumulate in the seed bank over an extended period of time following cultivation. Additionally, very few (n=2) modified-tame communities were cultivated historically and retained an abundance of native species (Chapter 3), with ordination showing a greater number (~5) of these pastures bearing similarity to pastures known to be non-cultivated. Seed banks of non-cultivated fields were associated with coarse textured soils, suggesting these pastures may be less suitable for annual cropping or conversion into improved pasture.

Within the Parkland region of north central Alberta, natural regeneration of native grasses like plains rough fescue (*Festuca hallii*) and western porcupine grass (*Hesperostipa comata*) from the seed bank is highly unlikely for several reasons. First, late seral native graminoids may form transient seed banks (Kinucan and Smeins 1992), suggesting that within the context of these highly cultivated landscapes with limited native grass cover, seed inputs to facilitate recolonization is unlikely. Second, in the case of rough fescue, flowering can be highly variable and intermittent between years (Toynbee 1987), with growing season conditions in combination with precipitation from the previous year regulating flowering and seed production (Biligetu et al. 2013; Palit et al. 2017). Third, field trials have repeatedly demonstrated that establishing rough fescue through seed is challenging (Desserud and Naeth 2013; Elsinger 2009), yet it readily germinates under greenhouse conditions (Romo et al. 1991). Early seral native graminoids were more abundant here in the seed banks of modified-tame pastures (i.e. tickle hairgrasss (*Agrostis scabra*), *Carex* spp., intermediate oatgrass (*Danthonia intermedia*)).

Previously cultivated pastures had higher densities of introduced ruderal forbs rather than native ruderal forbs, suggesting cultivation is a primary factor facilitating the build-up of undesirable ruderal forbs in seed banks. Disturbed habitats are characterized as having seed banks with long-lived (persistent) seeds and these species tend to be annuals or biennials (Bakker et al. 1996; Harper 1977). While actively cultivated fields can have seed densities lower than non-cultivated areas (Froud-Williams et al. 1983) as cultivation can alter the vertical stratification of seeds in the soil (Froud-Williams et al. 1983; Hoffman et al. 1998), this was not the case here. Instead, pastures with previous cultivation had abundant agronomic weeds. Surprisingly, overall richness and diversity were unaffected by cultivation because native diversity was largely replaced by introduced agronomic and weedy species. However, younger pastures (more recently cultivated) had greater richness and diversity, which reflected a greater seed density of introduced species. Recovery of native species has been observed in abandoned agricultural fields elsewhere in Australis and Europe (Cramer et al. 2008; Ruprecht 2006), and we detected a similar trend within tame pastures, even those actively grazed. Pastures with unknown cultivation history in the survey appeared to have been previously cultivated based on their seed bank composition, with changes in ownership preventing their classification. Seed banks of these pastures were not significantly different from pastures with known cultivation history, however they were typically associated with higher densities of introduced species, primarily ruderal forbs.

5.6.2 Grazing Management

In our survey area, grazed pastures are known to be highly productive, even exceeding that of native grasslands (Kupsch et al. 2013). The most abundant species in the seed bank of these pastures, namely Kentucky bluegrass (*Poa pratensis*), dandelion (*Taraxacum officinale*), and stinkweed (*Thlaspi arvense*), are indicative of pastures that have had a history of long-term heavy grazing and disturbance, as shown on public lands across this region of western Canada (Kupsch et al. 2013; Moisey et al. 2012). This is further supported by other studies in the region (Harker et al., Willms et al. 1985). While these species are known to be favored by intensive grazing (Bork 1993; Willms et al. 1985; Kupsch et al. 2013; Vujnovic at al. 2000), they nevertheless provide abundant forage to support livestock grazing (Kupsch et al. 2013). Other studies have also confirmed that these species, Kentucky bluegrass in particular, dominate the seed bank of grazed pastures (Sanderson et al. 2007; Tracy and Sanderson 2000; Travnicek et al. 2005; Willms and Quinton 1995). Despite its prevalence aboveground (2nd highest cover; Chapter 4), smooth brome (*Bromus inermis*) exhibited low seed abundance based on emergence in the greenhouse, thereby emphasizing the importance of asexual plant recruitment and the need to manage the bud bank of this species (Klimes 2007; Ott et al. 2016).

Presence of introduced forage grasses like Kentucky bluegrass (*Poa pratensis*), smooth brome (*Bromus inermis*), and common timothy (*Phleum pratense*) within the seed bank were associated with higher similarity to the aboveground vegetation and greater evenness. These pastures also had greater litter cover and higher scores for rangeland health in general. These findings suggest seed banks abundant in forage species could be managed for by ensuring pastures have at least 75% relative cover contributed by tall-statured introduced and native forage species, and higher litter cover could aid in the capture and retention of transient grass seeds. In order to manage for these guidelines, low to moderate grazing pressures are likely required (Willms and Quinton 1995).

Within the study area, proximity to the city of Edmonton has resulted in a highly fragmented landscape. Smaller farms and acreages can lead to simplistic grazing systems (i.e. single pastures, switchbacks, etc.) which can result in overutilization. There was no significant difference in the seed banks of pastures grazed continuously and rotationally, potentially because they were grazed at similar stocking rates (Chapter 3). Stocking rates are widely recognized as the primary management factor altering plant communities (Smoliak 1974; Willms et al. 1985) and seed bank composition (Kinucan and Smeins 1992; Willms and Quinton 1995). In the current study, although 60% of pastures were subject to rotational grazing, any benefits of rotational grazing may have been lost due to higher stocking densities, which substantially exceeded that recommended for the region (Chapter 3), particularly during the growing season.

Pastures that were solely used during winter had greater densities of desirable forages in the seed bank, with creeping red fescue (*Festuca rubra*), common timothy (*Phleum pratense*), cicer milkvetch (*Astragalus cicer*), and hard fescue (*Festuca ovina var. arundinacea*) emerging as indicators of growing season rest, presumably allowing these productive forages to set seed and form a seed bank (Tracey and Sanderson 2000). Common legumes included alfalfa (*Medicago sativa*), red clover (*Trifolium pratense*) and white clover, all of which were indicators for dormant season grazing and non-grazed pastures, which presumably benefited these palatable species by allowing them to grow, mature, and disseminate seed. Non-grazed pastures were also associated with several native forbs, native grasses, and graminoids in general, with intermediate oatgrass (*Danthonia intermedia*) being the strongest indicator species in the seed bank. Presence of the latter species appears to be closely tied to limited grazing, likely similar to that of other native grasses such as plains rough fescue.

The increased presence of weedier seed banks (introduced annuals and the noxious weed perennial sowthistle) in pastures with alternative livestock (sheep and alpacas) may be related to these areas being subject to more intensive grazing. Sheep in particular, are known to graze very closely, and may open up the canopy of pastures to the point of favoring ruderal, disturbance adapted species.

Supplemental hay feeding was relatively widespread and likely occurred on pastures to reduce grazing pressure, particularly in the absence of more pasture to accommodate the high demand for forage. These changes under supplemental feeding may have been partly induced by higher herbivore pressure as these pastures were associated with higher stocking rates (Chapter 3). Seed banks of these pastures were still impacted by this disturbance, with higher densities of introduced ruderal forbs, total introduced species, and total forbs. Feeding animals hay also has the potential to introduce seeds from forages and weedy bycatch, particularly if brought in from off-site, potentially spreading introduced or invasive species (Dutt et al. 1982). Germinable seed banks of these pastures were associated with disturbance adapted species like slough grass (*Bekmannia syzigachne*), lambsquarters (*Chenopodium album*), stinging nettle (*Urtica dioica*), and green foxtail (*Setaria viridis*), the latter of which is a noxious weed.

5.6.3 Fertilizer and Manure

Reductions in the seed bank density of forbs were likely reflective of changes in competitive dynamics within the pasture sward resulting from nutrient addition. Although we did not ask what nutrients were applied, N is frequently limiting of growth in grasslands, and therefore was likely applied. Fertilization with N would favor grasses, which in turn, would reduce legume abundance via heightened interspecific competition, thereby limiting the input of legume seeds (Aydin and Uzun 2005; Schellberg et al. 2001). Vigorous grass growth under fertilization may also explain the reduction in native ruderal forbs and noxious weeds in seed banks, and parallels previous studies on fertilization in the region (Grekul and Bork 2007; Schellberg et al. 2001). As producers were not asked about their motivation for using fertilization, our results can not be used to rule out the possibility that pasture managers were more likely to fertilize pastures lacking legumes due to their inability to maintain productivity in the absence of N-fixing legumes. Both real and perceived reductions in production of legume-impoverished swards could encourage fertilization to be used. Increased presence of quackgrass (*Elytrigia repens*) in the seed bank of these pastures may also indicate a higher intensity of pasture use, as this species is well-adapted to disturbance (Werner and Rioux 1977). Although no changes in species diversity were found under fertilization, species richness has been documented to return to seed banks in the long-term following the cessation of fertilization in hayfields (Bekker et al. 2000).

Manure addition was expected to increase the abundance and diversity of ruderal species through the introduction of endozoochorus seeds (Malo and Suárez 1995) that survive digestion and become stored in stockpiled manure (Pleasant and Schlather 1994). Manure piles can store high densities of seeds (up to 75,000 seeds/kg) (Pleasant and Schlather 1994), and this can increase the density of weedy annuals when applied to pasture (Dastgheib 1989; López-Mariño et al. 2000). However, the weed bank in stockpiled manure can also be managed, as extended periods of storage at higher temperatures during composting can reduce or eliminate weeds, with efficacy influenced by a species' biology (Larney and Blackshaw 2003; Wiese et al. 1998). Thus, sufficiently composted manure can minimize the risk of increasing the density of weed seeds in soil (Menalled et al. 2005). Unfortunatley, we did not determine the age of manure piles sampled. Stockpile age and winter forage sources likely contributed to divergence in seed bank composition of manure piles, which ranged from predominantly weedy forbs to piles containing desirable forages like legumes. We found an increase in Simpson's diversity in manured pasture soil, potentially due to the introduction of novel species in feed or colonizing manure piles, while native perennial grass and forb seed density declined. Reduced native seed was likely a result of pasture management history, as tame pastures were more likely to receive manure amendments. Treatment with manure was also associated with ruderal and halophytic species like *Chenopodium salinum,* common chickweed (*Stellaria media*), pineapple weed (*Matricaria matricarioides*) or foxtail barley (*Hordeum jubatum*). In Chapter 4, soil of manure treated pasture had higher salinity (electrical conductivity), which can be derived from manure inputs (Hao and Chang 2003). Similarly, the seed bank of stockpiled manure was dominated by weedy forbs, mainly goosefoot species (*Chenopodium* spp.). Shifts toward greater soil salinity of pastures receiving manure inputs could also result in the increased recruitment of halophytic plants, reproducing and creating seed rain *in-situ*. Forages were present in manure as well, with all three naturalized clover species represented. Legume seeds can exhibit high dormancy, aided by a thick seed coat, often requiring scarification or stratification to enable imbibition of the embryo (Acharya 2006; Baskin et al. 2000). Manure (likely deposited *in-situ*) cover in pastures was positively associated with legume seed.

5.6.4 Pasture Maintenance: Harrowing, Aeration, Swathing/Mowing

Harrowed pastures were not associated with large shifts in seed bank composition, plant functional groups, or measures of diversity. We previously identified harrowing as a management factor that accompanied manure amendments, resulting in similar plant community responses (Chapters 3 and

4). A few weedy indicator species were shared between manured and harrowed pastures [e.g. *Chenopodium salinum* and common chickweed (*Stellaria media*)]. *Chenopodium* species were common in sampled manure and common chickweed is known to survive herbivore digestion and therefore become more abundant in manure (Pleasant and Schlather 1994). Presence of the weeds white cockle (*Silene latifolia* ssp. *alba*) and pale smartweed (*Polygonum lapathifolium*) were unique to harrowing and may have been spread from annual cropland as soil bound to harrows during scarification of the soil surface. White cockle also emerged as an indicator of harrowing for the plant community (Chapter 4). Harrowing has been demonstrated elsewhere as an effective tool for reducing nuisance weeds in cultivated systems (Kurstjens and Kropff 2001; Wilson et al. 1993) but had limited testing in pasture.

Other mechanical forms of pasture management like mowing/swathing had no effects on seed bank composition, although a few indicator species were associated with this disturbance, including white clover. White clover is well adapted to mowing, in part due to the removal of overstory vegetation and the maintenance of high light levels (Kunelius and Campbell 1984), potentially benefiting this species. Aeration occurred infrequently and again was not associated with marked shifts in seed bank composition. Similar results have been found in Parkland pasture aboveground vegetation (Lardner et al. 2001; Malhi et al. 2000). However, aerated pasture seed banks did contain higher densities of legume seeds, with six introduced legume indicator species, as well as higher species diversity. Of the legumes, both black medic (*Medicago lupulina*) and yellow sweet clover (*Melilotus officinalis*) are potentially invasive and weedy species propagating mainly through seed (Turkington et al. 1978); additionally, the noxious weed white cockle (*Silene latifolia* ssp. *Alba*) was also linked to aeration. Feedback from producers suggested those who aerated were typically motivated to reduce soil compaction, increase porosity and improve water infiltration, which collectively should improve community productivity by improving root growth (Burgess et al. 2000). In theory, aeration could alter seed banks by altering ground cover characteristics and reducing limitations of seeds entering the seed bank, and perhaps improving their longevity. Working in a Parkland environment, Lardner et al. (2001) found aeration treatments coincided with greater soil exposure, decreased forage production in the year of treatment, and increased annual weeds, the latter of which were likely recruited from the seed bank following disturbance. In contrast, non-aerated fields had greater seed densities of introduced forage grasses, with Kentucky bluegrass (*Poa pratensis*) as the only indicator species. Overall, our results indicate that higher seed bank diversity in aerated fields is a product of the combination of increased legumes and annual weeds.

5.6.5 Herbicide and Noxious Weeds

Our original hypotheses predicted that seed banks of broadleaf plants would respond to herbicide, with legumes of special concern given their role in maintaining forage productivity and quality (Miller et al. 2015). Legume abundance in the seed bank did not respond as predicted, with a weak and non-significant decline in relation to recent herbicide exposure (within 3 years). Instead, total broadleaf and native ruderal forb density declined modestly in the seed bank. These results indicate that pastures sprayed with herbicide were able to retain legumes in the seed bank, in turn making natural regeneration of this important forage component possible. This contrasts with the conclusion of a small plot study by Miller et al. (2015), where emergence of broadcast seeded *Trifolium* and *Medicago* in old growth hayfields exposed to broadleaf herbicide declined, a response that persisted up to 15 to 24 months after treatment in Parkland soils. In our survey, we did not specifically ask how producers applied herbicide products, but based on our supplemental survey information, we found some producers had spot and/or broadcast sprayed affected areas. These contrasting methods could differentially alter plant communities, with pastures spot treated theoretically retaining more legume cover, and which in turn, could contribute to greater legume seed densities. Herbicide use was also associated with strong reductions in richness and diversity and may represent the fact that most herbicides used on pasture will be broadleaf-specific (Grekul and Bork 2007), in turn eliminating the growth and therefore seed input of this large group of plant species for the years following application. Moreover, this effect will be greater with herbicides having residual properties (e.g., Grekul and Bork 2007; Bork et al. 2007; Miller et al. 2015). Surprisingly, noxious weed species did not emerge as indicators of herbicide or a lack thereof given the obvious potential of the latter to control them; however, stinging nettle (*Urtica dioica*), which is a native nuisance weed, emerged as an indicator of herbicide treatment. The overall lack of noxious weed indicators in relation to the pastures examined here may also reflect the relatively high frequency of pastures that contain weeds (Chapter 3) and relatively lower proportion of them treated with herbicide.

Noxious weeds of concern, mainly Canada thistle (*Cirsium arvense*), and to a lesser extent common tansy (*Tanacetum vulgare*) and white cockle (*Silene latifolia* ssp. *alba*), were detected both above and belowground in the current study. Of note is that Canada thistle was relatively common aboveground and led to declined in range health (Chapter 3) but comprised a relatively small fraction of total seeds. Also of note is that the other noxious weeds encountered were relatively rare aboveground, but given their presence in the seed bank, these species may have potential to be more abundant than currently manifested in pasture vegetation. All these weeds are long-lived perennials, often with anti-herbivory mechanisms [aromatic terpenoids in *Tanacetum* (Kleine and Muller 2011) and spines in *Cirsium* spp. (Moore 1975)], making them difficult to eliminate without the use of herbicides.

Long-term presence of invasive species can have a legacy effect on seed banks as they contribute seeds over their lifetime and disrupt the input of native or other desirable species into the seed bank (D'Antonio and Meyerson 2002). Managing the seed bank of Canada thistle in pastures and native grasslands is of concern to producers as it is resilient to disturbance once established, and seedlings with two leaves can survive defoliation (Wilson 1979) and the plant can regenerate from very small root fragments (Gabruck et al. 2013). In native grasslands seed banks of Canada thistle infested areas were dominated by Kentucky bluegrass (*Poa pratensis*), followed by Canada thistle (Travnicek et al. 2005). Similarly, we found dominance of Kentucky bluegrass, which could be a more desirable species in tame pasture following thistle control (Grekul and Bork 2007), and in turn, could exacerbate dominance by this grass. Moreover, fertilization increased grasses and reduced noxious weed seeds, representative of intraspecific competitive shifts. Other studies have confirmed the formation of a persistent seed bank for white cockle (Peroni and Armstrong 2001), while the seed bank of common tansy is not well understood (Hogenbirk et al. 1992). There were also more noxious weeds in the seed bank when pastures were grazed continuously relative to non-grazed areas, and in areas with more grazing pressure.

5.6.6 Fire

Pastures that retained charred woody debris in the top 15 cm of mineral soil were likely subjected to fire some time ago and were associated with markedly divergent seed banks despite the lack of visible indicators of fire. Notably, seed banks did not differ with fire in pastures where this disturbance was known to have occurred (reported) by survey respondents. Charcoal in soil could date back to settlement or pre-settlement times, such as a historical fire which burned much of the study area in 1895 (Kjorlien 1977). In Chapter 4 we found an opposite response, where the aboveground community differed among pastures with variable reports of fire from pasture managers but did not differ in pastures containing charcoal. A seed bank study by Romo and Gross (2011) found that fescue grasslands with divergent fire histories (burned twice within 13 to 14 years vs >90 years pre-study) had seed bank compositions that were shaped by their burn histories. Recent intense disturbances, like a fire treatment, were also shown to overshadow long-term effects on seed bank composition (Romo and Gross 2011).

Seed banks of pastures containing charcoal had lower total seed densities, including that of forbs, native plants, and introduced seed. Other studies have reported reductions in seed density post-fire for grasses (Ren and Bai 2017), non-native plant species (Cox and Allen 2008; Ren and Bai 2017), total seed bank (Ferrandis et al. 2001), and both richness and diversity (Romo and Gross 2011). Reductions in seed density could be attributed to severe fires damaging seeds in the transient layer (i.e., litter and top 1 or 2 cm of soil) (Ferrandis et al. 2001), potentially eliminating non-native species, and preparing a relatively weed free seed bed. Burned pastures can also be susceptible to emergence from small, indurate annual seeds that survive fire and seed rain (Gonzales and Ghermandi 2008). Pastures with charcoal were associated with nodding brome (*Bromus anomalus*) and Bicknell's cranesbill (*Geranium bicknellii*), with the latter commonly expressed aboveground in burned areas (Tannas 2004). In contrast, pastures lacking charcoal were associated with weedy annuals, which are typically associated with cultivated land and disturbance. Native forbs, cattails (*Typha latifolia*), and noxious weed perennial sowthistle (*Sonchus arvensis*) were also more abundant.

Ren and Bai (2017) found increased seedling density *in-situ* and germinable seed bank emergence from a suite of native forbs (primarily *Artemisia* spp.) following burning in fescue prairie. Smoke was found to affect the germination of fescue prairie plants, with some species responding to smoke produced by specific species, including the legume alfalfa (*Medicago sativa*) (Ren and Bai 2016a). Interestingly, the noxious weed Canada thistle (*Cirsium arvense*), which is relatively common among Parkland pastures, exhibited reduced germination and embryo development with smoke (Ren and Bai 2016a). In germinable seed bank trials, Ren and Bai (2016b) found beneficial effects of ash addition (improved by the addition of smoke) on the recruitment of native prairie forbs. Thus, it is possible that charcoal presence in soils (much like ash) could be influencing seed germination and community assembly and warrants further investigation. Smoke and ash has been described as a germination cue in a number of other studies (Abu et al. 2016; Staden et al. 2000).

5.6.7 Rangeland Health

The rangeland health assessment (RHA) employed here observed soil and plant community characteristics to assess ecological function, and adds clarification to the results of Chapter 4 where rangeland health (RH) scores were affected by producer management. Lower RH scores were associated with higher stocking rates and higher intensities of pasture use (continuous grazing, small holdings, horses, year-long pasture use and supplemental feeding). We found that similarity between the seed bank and plant community was greater when RH scores were low, this is likely attributed to an abundance of weeds in the plant community, compromised soil cover (reduced litter and productive vegetation), and increased soil erosion and bare soil in pastures with low RH scores (Adams at al. 2005). Graminoids (Poaceae and Cyperaceae) were more abundant when RH scores were high, thus if producers want to manage for a seed bank with higher potential to recruit forage plants (which are typically grasses) maintaining tame and modified-tame pastures in healthy condition is prescribed. Our results also show that desirable, healthy seed banks are also associated with older pastures, overtime ruderal persistent species likely degrade. Seed densities and communities also responded to specific RH criteria, these were described in a brief summary in Appendix C.7.

5.6.8 Relationship between Seed Banks, Soils and Ground Cover

Seed densities and seed bank characteristics were associated with many soil properties and ground cover variables. For example, many species groupings in the seed bank (e.g. native, total forbs, native ruderal forbs, introduced ruderal forbs, and noxious weeds) were negatively related to litter cover. Litter in Parkland pastures provides a barrier to seed entry, including of weed seeds (Williams 1984), and may can prevent weedy forbs from establishing (Deák et al. 2011). Both native and introduced grass seed densities were positively associated with litter cover and could reflect increased inflorescences/seeds directly within standing grasses entering into the litter pool. As grasses senesce, fertile stems become incorporated into fallen litter and may still bare viable caryopses.

In Chapter 4, litter was reduced under intensive pasture management, causing increased bare soil. Soil exposure was associated with more similarity between the seed bank, which is dominated by weedy forbs, and the aboveground plant community; this pattern may reflect either increased inputs of ruderal grass and native ruderal forb seeds (Clements et al. 2007; Sanderson et al. 2014), or direct recruitment of ruderals into the aboveground community, both of which would increase similarity. Higher bare ground was also associated with fewer legume seeds and total graminoids in the seed bank, both of which are desirable species. Interestingly, the ground cover of plant stems (not differentiated by species) was positively associated with diversity and the density of native ruderal forb seeds (which was also reflected in native and overall seed density). In the current study, we suspect there is greater representation of native ruderal forbs within available niches. Although lichens and mosses were limited in cover among these predominantly tame pastures, their presence was associated with less similarity between seed banks and aboveground vegetation, possibly limiting the entry of introduced species into the seed bank. Lichen cover also corresponded with greater seed bank diversity, with lichens and mosses are important components of biological soil crusts and known to play a role in seed bank composition, primarily from research conducted in arid ecosystems (Clements et al 2007; Li et al. 2005).

Soil properties were weakly correlated with seed bank characteristics. Seed bank similarity was negatively related to most soil properties (e.g. OM, pH, N, C, C:N, sand), with the strongest relationship occurring with soil salinity. For the edaphic factors of soil OM, N, and C, seed banks could be exhibiting more similarity to vegetation when soil nutrients are lower. Ratios of C:N were also negatively associated with overall seed density and graminoids (grasses and sedges), with more diverse and even seed banks. This suggests that seed banks are perhaps more strongly influenced by management and the microenvironment of the soil surface, which regulates seed entry and seedling recruitment (Clements et al. 2007; Deák et al. 2011; Facelli and Pickett 1991; Li et al. 2005; Williams 1984).

5.6.9 Implications for the Management of Legume Seed Banks

Legumes were a common component of the seed bank, and it is evident that management had a role in determining the legumes that persist in the community. This research was initially motivated, in part, to determine whether legumes could regenerate from a seed bank under management conditions that reduced legume productivity, fecundity, and persistence, partly through chemical weed control (Miller et al. 2015), but also grazing management (Smith et al. 1988). This research showed that legume populations in northern temperate pastures were relatively unaffected when herbicides were used to control problematic broadleaf weeds. Instead, we found that both growing season grazing and fertilization were likely to reduce legumes in the seed bank, a response that was paralleled by lower aboveground legume foliar cover (Chapter 4). Legume-grass populations are susceptible to natural population oscillations (Schwinning and Parsons 1996), with the soil nitrogen regulating legume persistence. In the case of summer grazed pastures experiencing fertilization, the natural recovery of N-fixing legumes could be limited.

Native legumes like cream peavine (*Lathyrus ochroleucus*), American vetch (*Vicia americana*), and buffalo bean (*Thermopsis rhombifolia*), often common in native grasslands of the Parkland and lower Boreal, were underrepresented in the seed bank. However, these native legume species tend to produce fewer, larger seeds, and often spread vegetatively among suitable prairie ecosites. Naturalized white clover (*Trifolium repens*) followed by alsike clover (*Trifolium hybridum*) had the highest legume seed densities, and readily occupied niches created by grazing (Barret and Silander 1992; Tracy and Sanderson 2000).

5.6.10 Similarity

Seed densities of functional groups were surprisingly poorly correlated with cover of their corresponding group aboveground, with the exception of native functional groupings. Where the seed bank was abundant in desirable introduced forage grass seeds, the bank was formed under non-diverse stands of vegetation that expressed limited recruitment from introduced ruderal forbs. Introduced cover, primarily contributed by introduced forage grasses (Chapter 4), was negatively correlated with native perennial forb seed densities as competitive forage plants and introduced weeds are likely limiting the potential for native forbs establishment and seed set (Booth and Swanton 2002).

Many seed bank studies use the Sørenson's similarity index (Hopfensperger 2007; Tracy and Sanderson 2000) to compare seed bank richness to aboveground richness, or other indices (White et al. 2012). The information provided from this index showed high dissimilarity, but it exhibited limited responses and did not respond to pasture management in our study. High dissimilarity (66%) can be explained from a few perspectives. First, seed bank diversity is not apparent when sampling soil, as the cover and frequencies of individuals aboveground will not directly translate into their representation in the seed bank. Grassland cover tends to be dominated by a few competitive graminoids, making expression of perennial and weedy forbs relatively rare unless the ground is significantly disturbed, or the competitive nature of grasses is limited through defoliation (Grime 1979; Dyksterhuis 1949). However, an opposite relationship is generally expressed belowground as weedier annuals tend to dominate (Wellstein et al. 2007; Harker et al. 2000: Willms and Quinton 1995). Furthermore, aboveground disturbances can alter vegetation structure and diversity in the short-term (i.e. cultivation, fire, etc.), and alter seed bank inputs through seed-rain and recruitment of early seral or ruderal species, which can persist in the seed bank after aboveground recovery (Renne and Tracy 2007). It is also difficult to scale the intensity of sampling of belowground diversity in a way that similarly represents apparent aboveground diversity. Richness tallied from cover plots also may not scale with richness present across the entire pasture (Baltanás 1992; Hamer and Hill 2000). Although seed banks are typically comprised of propagules that disperse short distances from the parent plant, some plant families and genera have dispersal mechanisms that can overcome these limitations – i.e. seed dispersal features like a plumose pappus (e.g. many Asteraceae genera, *Epilobium*, etc.). Research has shown that seed banks are more accurately described when soil is sampled in higher volumes distributed across a large area (Benoit et al. 1989), requiring numerous smaller soil cores. It addition, some species that were expressed aboveground like grasses, shrubs, and some perennial forbs may have been missed during sampling due to rarity or other factors limiting their seedbank and may have been rare in the seed bank due to prolonged dormancy (discussed in detail below).

5.6.11 Limitations of the Study

We quantified the seed bank by measuring seedling recruitment from the germinable seed bank near the soil surface. It is important to note that total seed density and species richness could have been underestimated over the one-year germination period, as it is possible that some seed remained dormant, or seedlings may have been lost due to pre-emergence mortality. Seeds of certain species, legumes for example, are also known to have long dormancy periods which could have affected seed bank species richness.

This study also did not examine the composition and abundance of the vegetation bud bank. Compared to the seed bank, bud banks are severely understudied, particularly in northern temperate grasslands, despite their important role of facilitating revegetation and sustained production under ongoing disturbance. For some species that recruit slowly from seed banks, asexual propagation may be a more important mechanism for conserving the species (Klimes 2007). Interest in prairie bud banks has gained traction in recent years, particularly regarding invasive cool-season grasses that propagate vegetatively (Sprinkle 2010). Recruitment from the bud bank is important under disturbances like fire, which can negatively impact shallow seed densities, and native bud banks can make communities more resilient during periods of drought (Klimes 2007). Species like smooth brome have been shown to overwhelm the bud banks of native rhizomatous grasses like western wheatgrass (*Pascopyrum smithii*) under variable environments and grazing (Ott et al. 2016). Kentucky bluegrass, which is also aggressively rhizomatous, proved to be the most abundant species in our seed banks. In contrast, seeds of smooth brome were rarely detected.

While not specifically quantified, we detected an obvious bud bank during the trial, particularly for *Poa pratensis*. Where plants were recruited from buds they were excluded from the seed bank data set analyzed. We also attempted to remove vegetative propagules during the greenhouse trial set-up, in order to avoid confounding seed bank assessment with that from the bud bank. It is well known that most long-lived perennial grass species do not establish an effective seed bank and rely instead on vegetative propagules (i.e. smooth brome) (Otfinowski and Kenkel 2008). As result, simultaneously measuring both the seed and bud bank would have provided a more comprehensive understanding of plant recruitment potential.

Germinable seed banks provided insight into the maximum potential recruitment of species when and where these pastures were severely disturbed, as samples were grown out in the greenhouse in the absence of competing vegetation (i.e. seedlings were removed as they were identified). However, it is unlikely that these conditions duplicate conditions in the field, with the exception of perhaps cultivation. Without monitoring of *in-situ* seedling recruitment and their establishment in the natural environment (Ren and Bai 2017), our understanding of which species would be selected through environmental stresses and competition remains only an estimate, as we attempted to measure recruitment potential. Additional work examining the emergence and survival of select legumes in the field are assessed in the population demography study in Chapter 7.

5.7 Conclusions and Management Implications

Seed banks are reflective of long-term inputs from the aboveground plant community and ongoing persistence of seeds. Shifts in aboveground vegetation and ground cover resulting in degradation (i.e. increased erosion, increased bare ground, etc.) are reflected in the seed bank. Management practices that influence the ongoing addition or removal of seeds (Sanderson et al. 2007) through disturbances like the timing of grazing, herbicide application, manure spreading on pasture, and feeding of hay on pasture, were linked to divergent seed banks, often subtly, while severe disturbance legacies (i.e. cultivation and fire) also had strong influences on seed banks. Cultivation significantly altered the plant community and seed bank, eliminating a suite of native perennial grasses and forbs; however, native species also appeared to accumulate in the soil as pastures age. In contrast, younger pastures were associated with higher densities of undesirable plant species. Forage grasses and legumes seed densities responded positively to deferment of grazing into winter, and decreaser legumes were more abundant in abandoned pastures. Legumes were common above and belowground, and legumes appeared to specialize in certain disturbance regimes, with the same true of noxious weeds, suggesting both these vegetation groups can be manipulated by ongoing pasture management practices.

5.8 Literature Cited

Abu, Y., Romo, J. T., Bai, Y. and Coulman, B. 2016. Priming seeds in aqueous smoke solutions to improve seed germination and biomass production of perennial forage species. Canadian Journal of Plant Science **96**(4):551-563.

Acharya, S.N., Kastelic, J.P., Beauchemin, K.A., and Messenger, D.F. 2006. A review of research progress on cicer milkvetch (*Astragalus cicer* L.). Canadian Journal of Plant Science **86**(1):49-62.

Adams, B. W., Ehlert, G., Stone, C., Lawrence, D., Alexander, M., Willoughby, M., Hincz, C., Moisey, D., Burkinshaw, A., Carlson, J. and France, K. 2005. Rangeland health assessment for grassland, forest and tame pasture. Public Lands and Forests Division, Alberta Sustainable Resource Development.

Albrecht, H. and Auerswald, K. 2003. Arable weed seedbanks and their relation to soil properties. Aspects of Applied Biology **69**:11-20.

Anderson, M. J. 2005. Permutational multivariate analysis of variance. Department of Statistics, University of Auckland **26**:32-46.

Aydin, I., and Uzun, F. 2005. Nitrogen and Phosphorus fertilization of rangelands affects yield, forage quality and the botanical composition. European Journal of Agronomy **23**(1):8-14.

Bailey, A.W., McCartney, D. and Schellenberg, M.P. 2010. Management of prairie rangeland. Agriculture and Agri-Food Canada.

Bakker, J.P., Poschlod, P., Strykstra, R.J., Bekker, R.M., Thompson, K. 1996. Seed banks and seed dispersal: important topics in restoration ecology. Acta Botanica Neerlandica **45**(4):461-490.

Baltanás, A. 1992. On the use of some methods for the estimation of species richness. Oikos **65**(3):484-492.

Barret, J. P. and Silander Jr., J.A. 1992. Seedling recruitment limitation in white clover (*Trifolium repens*; Leguminosae). American Journal of Botany **79**(6):643-649.

Baskin. J.M., Baskin, C.C., and Li, X. 2000. Taxonomy, anatomy and evolution of physical dormancy in seeds. Plant Species Biology **15**:139-152.

Bekker, R. M., Verweij, G. L., Bakker, J. P. and Fresco, L. F. 2000. Soil seed bank dynamics in hayfield succession. Journal of Ecology **88**(4):594-607.

Benoit, D. L., N. C. Kenkel, and Cavers, P.B. 1989. Factors influencing the precision of soil seed bank estimates. Canadian Journal of Botany **67**:2833-2840.

Benvenuti, S. 2007. Natural weed seed burial: effect of soil texture, rain and seed characteristics. Seed Science Research **17**(3):211-219.

Booth, B.D., and Swanton, C.J. 2002. 50^{th} anniversary—invited article: Assembly theory applied to weed communities. Weed Science **50**:2-13.

Bork, E. 1993. Interaction of burning and herbivory in aspen communities in Elk Island National Park, Alberta. M.Sc. Thesis, University of Alberta, Edmonton.

Bork, E.W., Grekul, C.W., and De Bruijn, S.L. 2007. Extended pasture forage sward responses to Canada thistle (*Cirsium arvense*) control using herbicides and fertilization. Crop Protection **26**(10):1546-1555.

Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, K.M., Gillen, R.L., Ash., A.J. and Willms, W.D. 2008. Rotational grazing on rangelands: Reconciliation of perception and experimental evidence. Rangeland Ecology and Management **61**:3-17.

Burgess, C. P., Chapman, R., Singleton, P. L. and Thom, E. R. 2000. Shallow mechanical loosening of a soil under dairy cattle grazing: Effects on soil and pasture. New Zealand Journal of Agricultural Research **43**(2):279-290.

Clements, D.R., Krannitz, P.G., and Gillespie, S.M. 2007. Seed bank responses to grazing history by invasive and native plant species in a semi-desert shrub-steppe environment. Northwest Science **81**(1):37-49.

Coupland, R.T., and Brayshaw, T.C. 1953. The fescue grassland in Saskatchewan. Ecology **34**(2):386-405.

Cox, R. D. and Allen, E. B. 2008. Composition of soil seed banks in southern California coastal sage scrub and adjacent exotic grassland. Plant Ecology **198**(1):37-46.

Cramer, V. A., Hobbs, R. J. and Standish, R. J. 2008. What's new about old fields? Land abandonment and ecosystem assembly. Trends in Ecology and Evolution **23**(2):104-112.

D'Antonio, C. and Meyerson, L.A. 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restoration Ecology **10**(4):703-713.

Dastgheib, F. 1989. Relative importance of crop seed, manure and irrigation water as sources of weed infestation. Weed Research **29**(2):113-116.

De Caceres, M., Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. Ecology, URL http://sites.google.com/site/miqueldecaceres/

De Keyser, E.S., Dennhardt, L.A., and Hendrickson, J. 2015. Kentucky bluegrass (*Poa pratensis*) invasion in the Northern Great Plains: a story of rapid dominance in an endangered ecosystem. Weed Science **64**(3):409-420.

De Mendiburu, F. 2017. agricolae: Statistical Procedures for Agricultural Research. R package version 1.2-6. https://CRAN.R-project.org/package=agricolae

Deák, B., Valkó, O., Kelemen, A., Török, P., Miglécz, T., Ölvedi, T., Lengyel, S., and Tóthmérész, B. 2011. Litter and graminoid biomass accumulation suppresses weedy forbs in grassland restoration. Plant Biosystems - An International Journal Dealing with all Aspects of Plant Biology **145**(3):730-737.

Desserud, P. A. and Naeth, M. A.,2013. Promising results in central Alberta with rough fescue (*Festuca hallii*) seeding following disturbance. Native Plants Journal **14**(1):25-32.

Donkor, N.T., Gedir, J.V., Hudson, R.J., Bork, E.W., Chanasyk, D.S., and Naeth, M.A. 2002. Impacts of grazing systems on soil compaction and pasture production in Alberta. Canadian Journal of Plant Science **82**(1):1-8.

Dutt, T. E., Harvey, R. G. and Fawcett, R. S. 1982. Feed quality of hay containing perennial broadleaf weeds. Agronomy Journal **74**(4):673-676.

Dyksterhuis, E.J. 1949. Condition and management of rangeland based on quantitative ecology. Journal of Range Management **2**(3):104-115.

Eriksson, A. and Eriksson, O. 1997. Seedling recruitment in semi-natural pastures: the effects of disturbance, seed size, phenology and seed bank. Nordic Journal of Botany **17**(5):469-480.

Eschtruth, A.K. and Battles, J.J. 2009. Assessing the relative importance of disturbance, herbivory, diversity, and propagule pressure in exotic plant invasion. Ecological Monographs **79**:265-280.

Facelli, J.M., and Pickett, S.T.A. 1991. Plant litter: its dynamics and effects on plant community structure. Botanical Review **57**:2-32.

Ferrandis, P., Herranz, J., Martínez, J. and Martínez-Sánchez, J. 2001. Response to fire of a predominantly transient seed bank in a Mediterranean weedy pasture (eastern-central Spain). Ecoscience **8**(2):211-219.

Froud-Williams, R. J., Chancellor, R. J. and Drennan, D. S. H. 1983. Influence of cultivation regime upon buried weed seeds in arable cropping systems. Journal of Applied Ecology **20**(1):199-208.

Gabruck, D. T., Bork, E. W., Hall, L. M., King, J. R. and Hare, D. D. 2013. Interspecific relationships between white clover, Kentucky bluegrass, and Canada thistle during establishment. Agronomy Journal **105**(6):1467-1474.

Gauthier, D.A., and Wiken, E.B. 2003. Monitoring the conservation of grassland habitats, prairie ecozone, Canada. Environmental Monitoring and Assessment **88**:343-364.

Gonzalez, S. and Ghermandi, L. 2008. Postfire seed bank dynamics in semiarid grasslands. Plant Ecology **199**(2):175-185.

Grekul, C.W. and Bork, E.W. 2007. Fertilization augments Canada thistle (*Cirsium arvense* L. Scop.) control in temperate pastures with herbicides. Crop Protection **26**(4):668-676.

Grekul, C.W., Cole, D.E. and Bork, E.W. 2005. Canada thistle (*Cirsium arvense*) and pasture forage responses to wiping with various herbicides. Weed Technology **19**(2):298-306.

Grime, J.P. 1979. Plant strategies and vegetation processes. John Wiley & Sons, New York, USA.

Gross, J. and Ligges, U. 2015. nortest: Tests for normality. R package version 1.0-4. https://CRAN.R-project.org/package=nortest

Hamer, K.C. and Hill, J.K. 2000. Scale-dependent effects of habitat disturbance on species richness in tropical forests. Conservation Biology **14**(5):1435-1440.

Hao, X. and Chang, C. 2003. Does long-term heavy cattle manure application increase salinity of a clay loam soil in semi-arid southern Alberta? Agriculture, Ecosystems and Environment **94**(1):89-103.

Harker, K.N., Baron, V.S., Chanasyk, D.S., Naeth, M.A., and Stevenson, F.C. 2000. Grazing intensity effects on weed populations in annual and perennial pasture systems. Weed Science **48**(2):231-238.

Harper, J.L. 1977. Population biology of plants. Academic Press, London.

Hoffman, M. L., Owen, M. D. and Buhler, D. D. 1998. Effects of crop and weed management on density and vertical distribution of weed seeds in soil. Agronomy Journal **90**(6):793-799.

Hogenbirk, J. C. and Wein, R. W. 1992. Temperature effects on seedling emergence from boreal wetland soils: implications for climate change. Aquatic Botany **42**(4):361-373.

Hopfensperger, K.N. 2007. A review of similarity between seed bank and standing vegetation across ecosystems. Oikos **116**(9):1438-1448.

Jacquemyn, H., Mechelen, C.V., Brys, R., and Honnay, O. 2011. Management effects on the vegetation and soil seed bank of calcareous grasslands: An 11-year experiment. Biological Conversation **144**(1):416-422.

Johnston, A., Smoliak, S. and Stringer, P.W. 1969. Viable seed populations in Alberta prairie topsoils. Canadian Journal of Plant Science **49**(1):75-82.

Kinucan, R.J. and Smeins, F.E. 1992. Soil seed bank of a semiarid Texas grassland under three long-term (36-years) grazing regimes. The American Midland Naturalist **128**(1):11-21.

Kjorlien, M.E. 1977. A review of historical information of fire history and vegetation description of Elk Island and the Beaver Hills. Parks Canada, Elk Island National Park, Fort Saskatchewan, Alberta.

Kleine, S. and Müller, C. 2011. Intraspecific plant chemical diversity and its relation to herbivory. Oecologia **166**(1):175-186.

Klimes, J. 2007. Bud banks and their role in vegetative regeneration-a literature review and proposal for simple classification and assessment. Perspective in Plant Ecology, Evolution and Systematics **8**(3):115-129.

Kunelius, H.T. and Campbell, A.J. 1984. Performance of sod-seeded temperate legumes in grass dominant swards. Canadian Journal of Plant Science **64**(3):643-650.

Kupsch, T., France, K., Loonen, H., Burkinshaw, A., Willoughby, M., and McNeil, R. L. 2013. Guide to range plant community types and carrying capacity for the Central Parkland subregion of Alberta. Alberta Sustainable Resource Development, Government of Alberta.

Kurstjens, D. A. G. and Kropff, M. J. 2001. The impact of uprooting and soil-covering on the effectiveness of weed harrowing. Weed Research **41**(3):211-228.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2000. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on dry matter yield and forage quality. Canadian Journal of Plant Science **80**(4): 781-791.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2001. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on botanical composition. Canadian Journal of Plant Science **81**(4):673-683.

Larney, F. J. and Blackshaw, R. E. 2003. Weed seed viability in composted beef cattle feedlot manure. Journal of Environmental Quality **32**(3):1105-1113.

Ledgard, S. F. and Steele, K.W. 1992. Biological nitrogen fixation in mixed legume/grass pastures. Plant and Soil **141**:137-153.

Lenth, R.V. 2016. Least-Squares Means: The R Package lsmeans. Journal of Statistical Software **69**(1): 1-33. doi:10.18637/jss.v069.i01

Levassor, C., M. Ortega, and Peco, B. 1990. Seed bank dynamics of Mediterranean pastures subjected to mechanical disturbance. Journal of Vegetation Science **1**(3):339-344.

Li, X. Jia, X., Long, L., and Zerbe, S. 2005. Effects of biological soil crusts on seed bank, germination and establishment of two annual plant species in the Tengger Desert (N China). Plant and Soil **277**(1):375-385.

López-Mariño, A., Luis-Calabuig, E., Fillat, F., and Bermudez, F.F. 2000. Floristic composition of established vegetation and the soil seed bank in pasture communities under different traditional management regimes. Agriculture, Ecosystems and Environment **78**(3):273-282.

Ma, M., Zhou, X., and Du, G. 2010. Role of soil seed bank along a disturbance gradient in an alpine meadow on the Tibet Plateau. Flora-Morphology, Distribution, Functional Ecology of Plants **205**(2):128-134.

Malhi, S.S., Heier, K., Nielsen, K., Davies, W.E., and Gill, K.S. 2000. Efficacy of pasture rejuvenation through mechanical aeration and N fertilization **80**:813-815.

Malo, J.E., and Suárez, F. 1995. Establishment of pasture species on cattle dung: the role of endozoochorous seeds. Journal of Vegetation Science **6**(2):169-174.

Mayor, J. P. and Dessaint, F. 1998. Influence of weed management strategies on soil seedbank diversity. Weed Research **38**(2):95-106.

McCartney, D.H. 1993. History of grazing research in the Aspen Parkland. Canadian Journal of Animal Science **73**(4):749-763.

McLean, A. and Wikeem, S. 1985. Rough fescue response to season and intensity of defoliation. Journal of Range Management **38**(2):100-103.

Menalled, F. D., Kohler, K. A., Buhler, D. D. and Liebman, M. 2005. Effects of composted swine manure on weed seedbank. Agriculture, Ecosystems and Environment **111**(1):63-69.

Milchunas, D.G., Sala, O.E. and Lauenroth, W.K. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. The American Naturalist **132**(1):87-106.

Miller, A.J., Bork, E.W., Hall, L.M., and Summers, B. 2015. Long-term forage dynamics in pastures sprayed with residual broadleaf herbicide: A test of legume recovery. Canadian Journal of Plant Science **95**(1):43-53.

Moisey, D., Young, J., Lawrence, D., Stone, C., and Willoughby, M. 2012. Guide to range plant community types and carrying capacity for the Dry and Central Mixedwood Subregions in Alberta. Alberta Sustainable Resource Development, Government of Alberta.

Moore, R.J. 1975. The biology of Canadian weeds. 13. *Cirsium arvense* (L.) Scop. Canadian Journal of Plant Science **55**:1033-1048.

Moss, E.H. 2010. Flora of Alberta, 2nd ed. University of Toronto Press, Toronto.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, K., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson. G.L., Solymos, P., M., Stevens, M.H., Szoecs, E., and Wagner, H. 2017. vegan: Community ecology package. R package version 2.4-4. https://CRAN.R-project.org/package=veg

Otfinowski, R., and Kenkel, N.C. 2008. Clonal integration facilitates the proliferation of smooth brome clones invading northern fescue prairies. Plant Ecology **199**(2):235-242.

Otfinowski, R., Kenkel, N.C., Van Acker, R.C. 2008. Reconciling seed dispersal and seed bank observations to predict smooth brome (*Bromus inermis*) invasions of northern prairie. Invasive Plant Science and Management **1**:279-286.

Ott, J.P., Butler, J.L., Rong, Y., and Xu, L. 2016. Greater bud outgrowth of *Bromus inermis* than *Pascopyrum smithii* under multiple environmental conditions. Journal of Plant Ecology **10**(3):518-527.

Palit, R., Bai, Y., Romo, J., Coulman, B. and Warren, R. 2016. Seed production in *Festuca hallii* is regulated by adaptation to long-term temperature and precipitation patterns. Rangeland Ecology and Management **70**(2):238-243.

Peroni, P. A. and Armstrong, R. T. 2001. Density, dispersion and population genetics of a *Silene latifolia* seed bank from southwestern Virginia. Journal of the Torrey Botanical Society **128**(4):400-406.

Pleasant, J.M.T., and Schlather, K.J. 1994. Incidence of weed seed in cow (*Bos* sp.) manure and its importance as a weed source for cropland. Weed Technology **8**(2):304-310.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Rashford, B.S., Walker, J.A., and Bastian, C.T. 2011. Economics of grassland conversion to cropland in the prairie pothole region. Conservation Biology **25**(2):276-284.

Ren, L. and Bai, Y. 2016a. Smoke and ash effects on seedling emergence from germinable soil seed bank in Fescue Prairie. Rangeland Ecology and Management **69**(6):499-507.

Ren, L. and Bai, Y. 2016b. Smoke originating from different plants has various effects on germination and seedling growth of species in Fescue Prairie. Botany **94**(12):1141-1150.

Ren, L. and Bai, Y. 2017. Burning modifies composition of emergent seedlings in fescue prairie. Rangeland Ecology and Management **70**(2):230-237.

Renne, I.J. and Tracy, B.F. 2007. Disturbance persistence in managed grasslands: shifts in aboveground community structure and the weed seed bank. Plant Ecology **190**(1):71-80.

Romo, J.T., Grilz, P. L., Bubar, C. J. and Young, J. A. 1991. Influences of temperature and water stress on germination of plains rough fescue. Journal of Range Management **44**(1):75-81.

Romo, J.T. and Gross, D.V. 2011. Preburn history and seasonal burning effects on the soil seed bank in the Fescue Prairie. The American Midland Naturalist **165**(1):74-90.

Rowe, J.S. 1987. Status of the aspen parkland in the Prairie Provinces. Pp 27-44, In: Endangered Species in the Prairie Provinces, Proceedings of the 1st Prairie Conservation and Endangered Species Conference, January 24-26, Edmonton, Alberta.

Ruprecht, E. 2006. Successfully recovered grassland: a promising example from Romanian old-fields. Restoration Ecology **14**(3):473-480.

Sanderson, M.A., S.C. Goslee, K.D. Klement, and Soder, K.J. 2007. Soil seed bank composition in pastures of diverse mixtures of temperate forages. Agronomy Journal **99**(6):1514-1520.

Sanderson, M.A., Stout, R., Goslee, S., Gonet, J. and Smith, R.G., 2014. Soil seed bank community structure of pastures and hayfields on an organic farm. Canadian Journal of Plant Science **94**(4):621-631.

Schellberg, J., Möseler, B.M., Kühbauch, W., and Rademacher, I.F. 2001. Long-term effects of fertilizer on soil nutrient concentration, yield, forage quality and floristic composition of a hay meadow in the Eifel Mountains, Germany. Grass and Forage Science **54**(3):195-207.

Schwinning, S. and Parsons, A. J. 1996. Analysis of the coexistence mechanisms for grasses and legumes in grazing systems. Journal of Ecology **84**(6):799-813.

Sprinkle, J.W. 2010. Bud bank density regulates invasion by exotic plants. M.Sc. Thesis, Oklahoma State University.

Smith, S.R., J. H. Bouton and Hoveland, C.S. 1988. Alfalfa persistence and regrowth potential under continuous grazing. Agronomy Journal **81**:960-965.

Smoliak, S. 1974. Range vegetation and sheep production at three stocking rates on *Stipa-Bouteloua* prairie. Journal of Range Management **27**(1): 23-26.

Staden, J. V., Brown, N. A., Jäger, A. K., and Johnson, T. A. 2000. Smoke as a germination cue. Plant Species Biology **15**(2):167-178.

Tannas, K. E. 2004. Common Plants of the Western Rangelands – Volume 3: Forbs. Olds College. Alberta Agriculture and Rural Development, Edmonton, Alberta.

Thomas, A.G. 1985. Weed survey system used in Saskatchewan for cereal and oilseed crops. Weed Science **33**(1):34-43.

Toynbee, K. 1987. Prolific flowering year for plains rough fescue at the Kernen Prairie. Blue Jay **45**:142-143.

Tracy, B. F., and Sanderson, M. A. 2000. Seedbank diversity in grazing lands of the Northeast United States. Journal of Range Management **53**(1):114-118.

Travnicek, A.J., Lym, R.G. and Prosser, C. 2005. Fall-prescribed burn and spring-applied herbicide effects on Canada thistle control and soil seedbank in a northern mixed-grass prairie. Rangeland Ecology and Management **58**(4):413-422.

Turkington, R.A., Cavers, P.B. and Rempel, E. 1978. The biology of Canadian weeds.: 29. *Melilotus alba* Desr. and *M. officinalis* (L.) Lam. Canadian Journal of Plant Science **58**(2):523-537.

Walton, P.D., Martinez, R. and Bailey, A.W. 1981. A comparison of continuous and rotational grazing. Journal of Range Management **34**(1):19-21.

Wellstein, C., Otte, A., and Waldhardt, R. 2007. Seed bank diversity in mesic grasslands in relation to vegetation type, management and site conditions. Journal of Vegetation Science **18**:153-162.

Werner, P.A. and Rioux, R. 1977. The biology of Canadian weeds. 24. *Agropyron repens* (L.) Beauv. Canadian Journal of Plant Science **57**(3):905-919.

White, S.R., Bork, E.W., Karst, J., and J.F. Cahill Jr. 2012. Similarity between grassland vegetation and seed bank shifts with altered precipitation and clipping, but not warming. Community Ecology **13**(2):129-136.

Wiese, A. F., Sweeten, J. M., Bean, B. W., Salisbury, C. D. and Chenault, E. W., 1998. High temperature composting of cattle feedlot manure kills weed seed. Applied Engineering in Agriculture **14**(4):377-380.

Williams, E.D. 1984. Changes during 3 years in the size and composition of the seed bank beneath a long-term pasture as influenced by defoliation and fertilizer regime. Journal of Applied Ecology **21**:603-615.

Willms, W. D., and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Willms, W.D., Smoliak, S., and Dormaar, J.F. 1985. Effects of stocking rate on a rough fescue grassland vegetation. Journal of Range Management **38**(3):220-225.

Wilson, B. J., Wright, K. J. and Butler, R. C. 1993. The effect of different frequencies of harrowing in the autumn or spring on winter wheat, and on the control of *Stellaria media* (L.) vill., *Galium aparine* L. and *Brassica napus* L. Weed Research **33**(6):501-506.

Wilson Jr., R.G. 1979. Germination and seedling development of Canada thistle (*Cirsium arvense*). Weed Science **27**(2): 146-151.

Young, J. E., Sánchez-Azofeifa, G. A., Hannon, S. J. and Chapman, R. 2006. Trends in land cover change and isolation of protected areas at the interface of the southern Boreal Mixedwood and Aspen Parkland in Alberta, Canada. Forest Ecology and Management **230**(1):151-161.

Zhan, X., Li, L., and Cheng, W. 2007. Restoration of *Stipa krylovii* steppes in Inner Mongolia of China: Assessment of seed banks and vegetation composition. Journal of Arid Environments **68**:298-307.

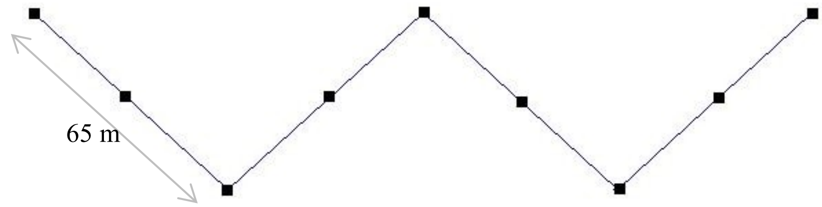


Figure 5.1. W-shaped transect used for sampling of vegetation and seed bank in each pasture. Each segment of the 'W' is 65 m, totaling 260 m. Black squares represent points where foliar cover was measured, every 32.5 m, using a 50 cm x 50 cm quadrat. Soil cores for seed bank assessment were sampled every 5 m (n = 53).

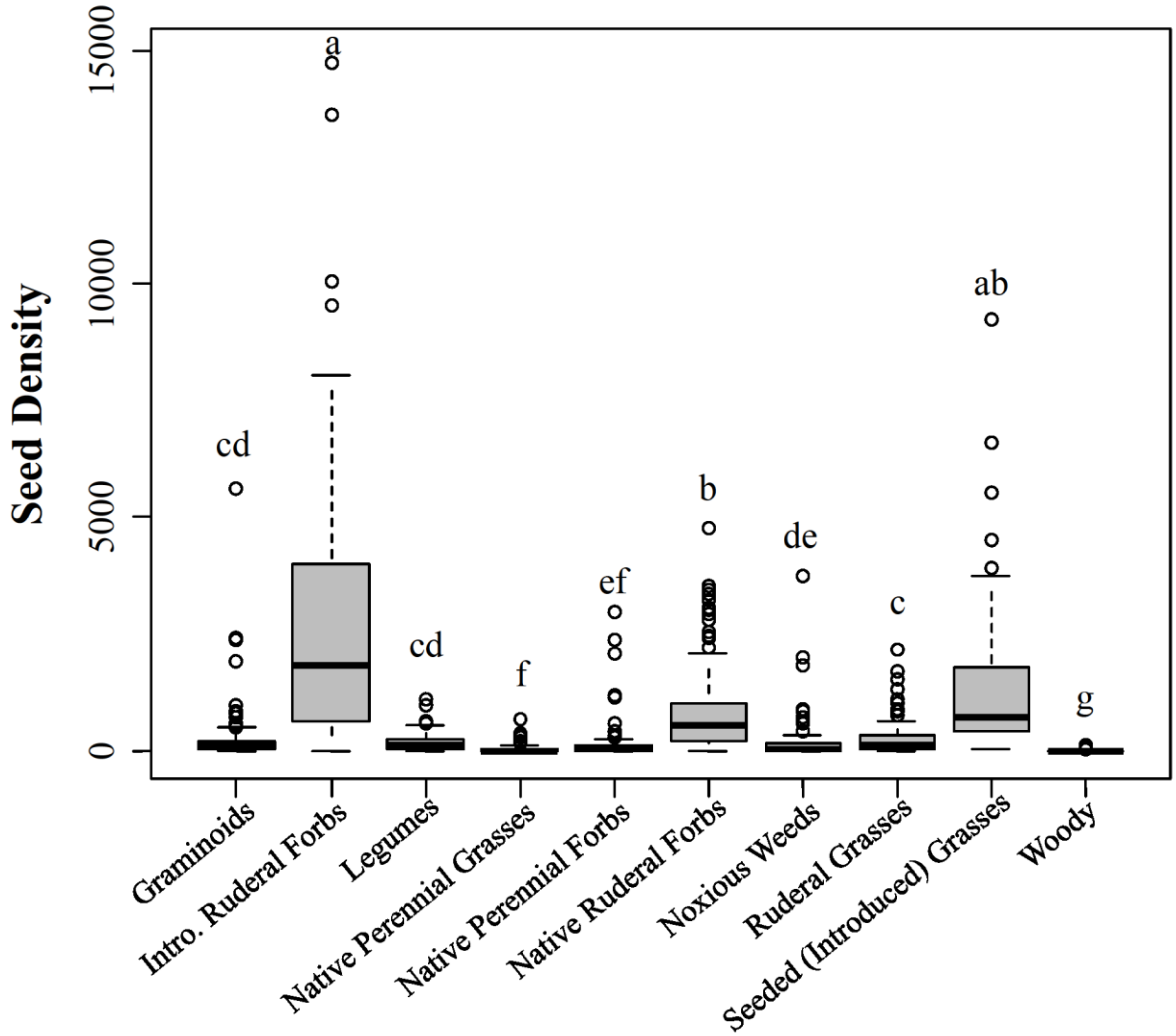


Figure 5.2. Median seed density (seeds/m^2 ± IQR) of various functional plant groups present within northern temperate pastures of north central Alberta, Canada (X^2 = 581.9, df = 9, P < 0.001).

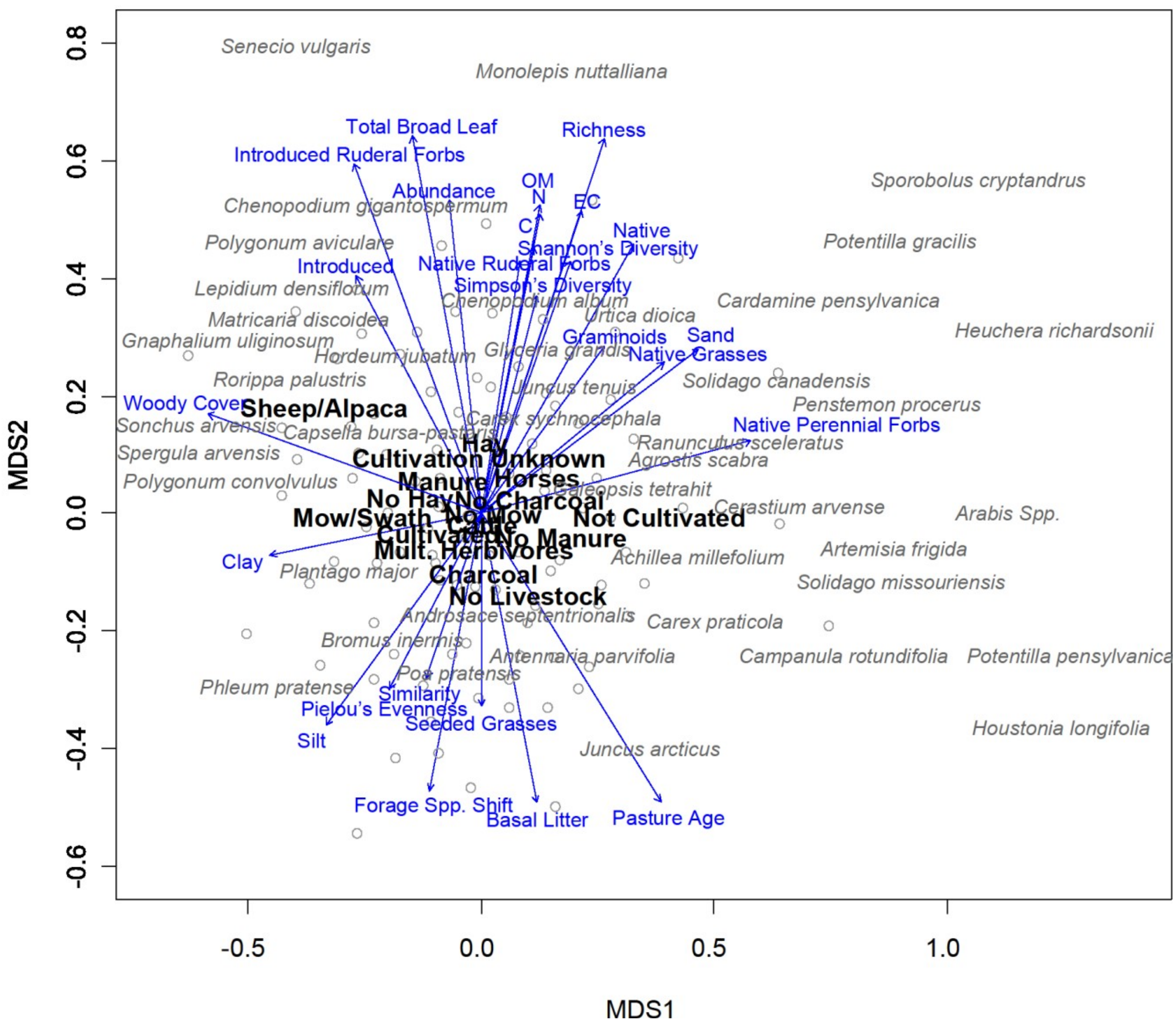


Figure 5.3. Resulting NMDS ordination of seed bank composition (distance = Bray-Curtis, dimensions = 2, stress = 0.31) collected from 102 pastures across north central Alberta during 2012 and 2013. Centroids of all management factors (bolded), plant species (grey text), as well as vectors for soil properties, RHA scores, functional group seed density, and various vegetation indices (blue text) plotted were significant at $P < 0.05$. Vectors 'Woody Cover' and 'Forage Spp. Shift' were derived from RHA scores. Longer vectors indicate sites with higher scores for the attributes. Significance tests are located in Tables C.2.1 to C.2.3. Significant management factors (Table C.2.1) are decomposed in Fig. 5.4.

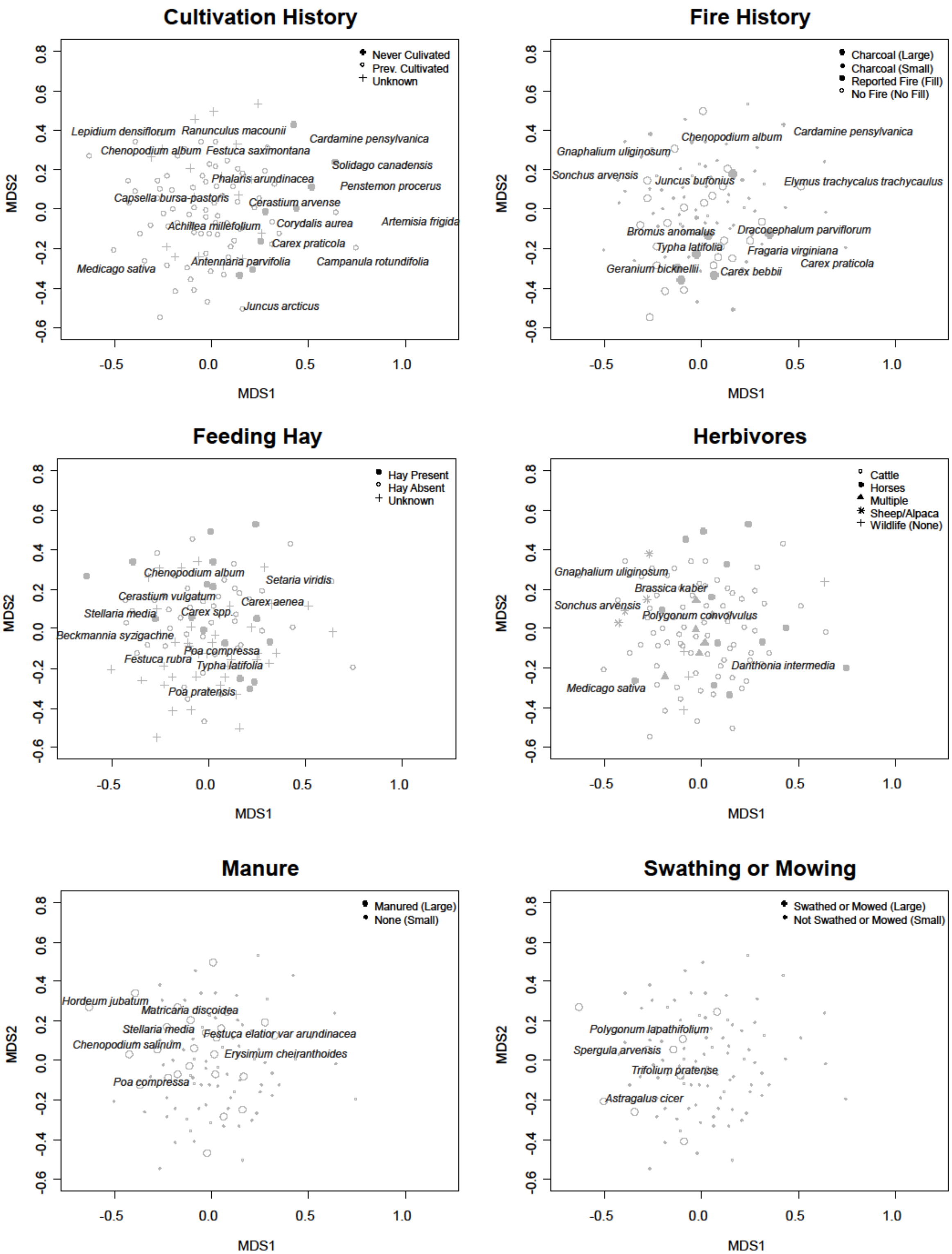


Figure 5.4. Resulting NMDS ordination of seed bank composition (distance = Bray-Curtis, dimensions = 2, stress = 0.31) using the same scores from Fig 5.3 and demonstrating the relationship between significant management factors (centroids) and their indicator plant species in the seed bank.

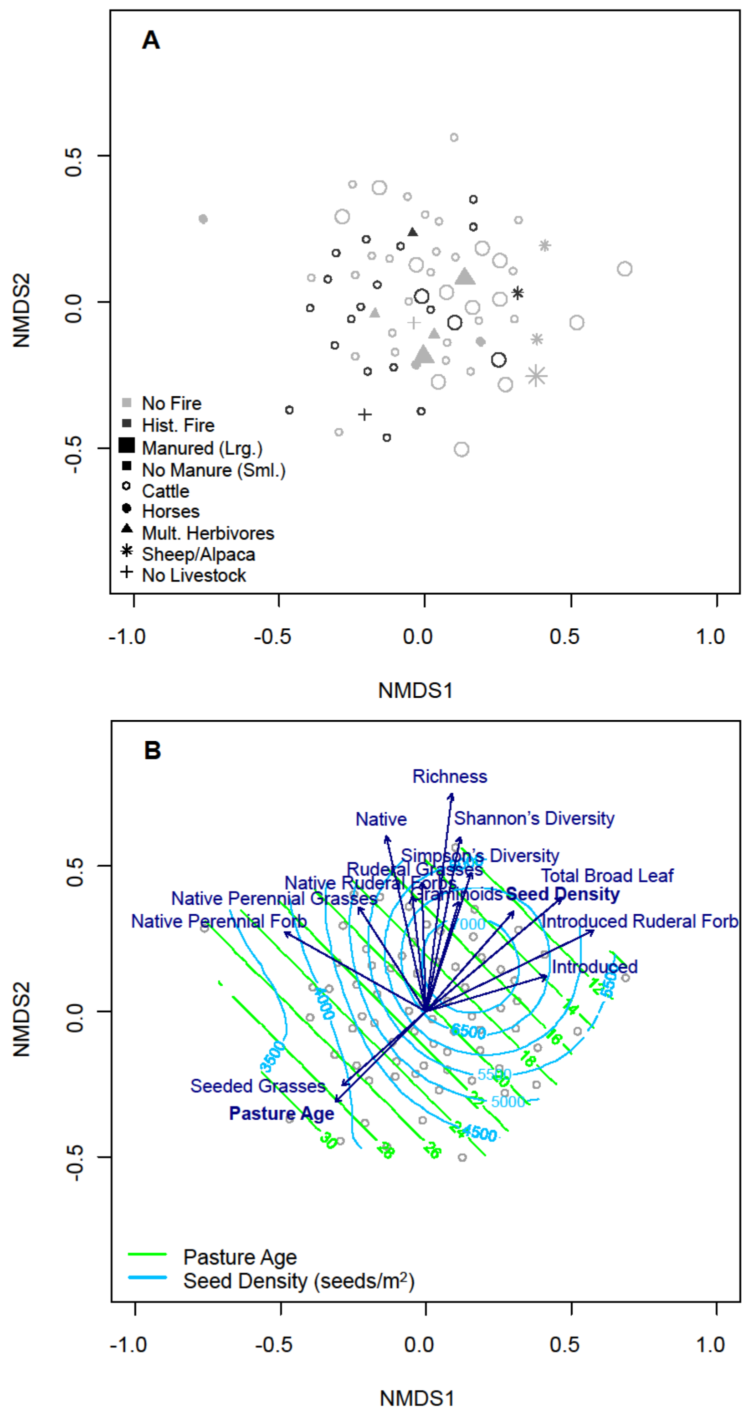


Figure 5.5. Resulting NMDS ordination of seed bank composition from pastures (n=71) where the approximate date of last cultivation was known (distance = Bray-Curtis, dimensions = 2, stress = 0.30). Panels A and B use the same scores, with panel A showing the relationship of site with significant management factors ($P < 0.05$), and panel B shows the relationship between pasture age and the abundance of various seed bank vegetation groupings.

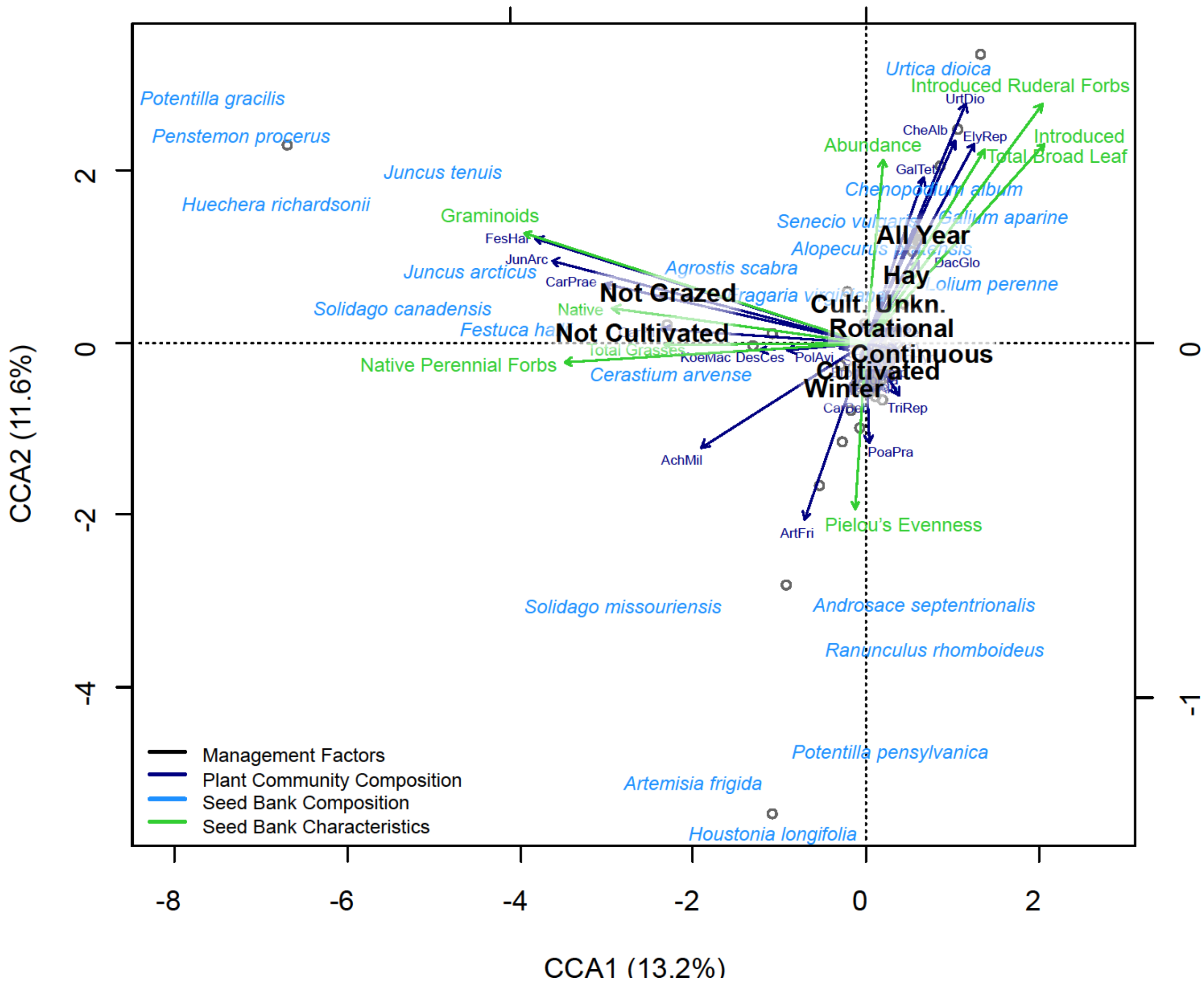


Figure 5.6. Results of a canonical correspondence analysis (CCA) of seed bank composition constrained by overlying plant community cover across 102 pastures of north central Alberta sampled in each of 2012 and 2013 [distance = Euclidean, axes = 27 (only the first 2 displayed in figure)]. The CCA model explained 56.0% of the variance in seed bank composition, with 17/27 axes significant. For simplification only the first two axes explaining nearly half the variance (24.8 %) are displayed. Plant community variables were selected using a step-wise permutational process. Management factors, and seed bank characteristics and species displayed were all significant at $P < 0.05$, while all plant community vectors included in the CCA model are displayed.

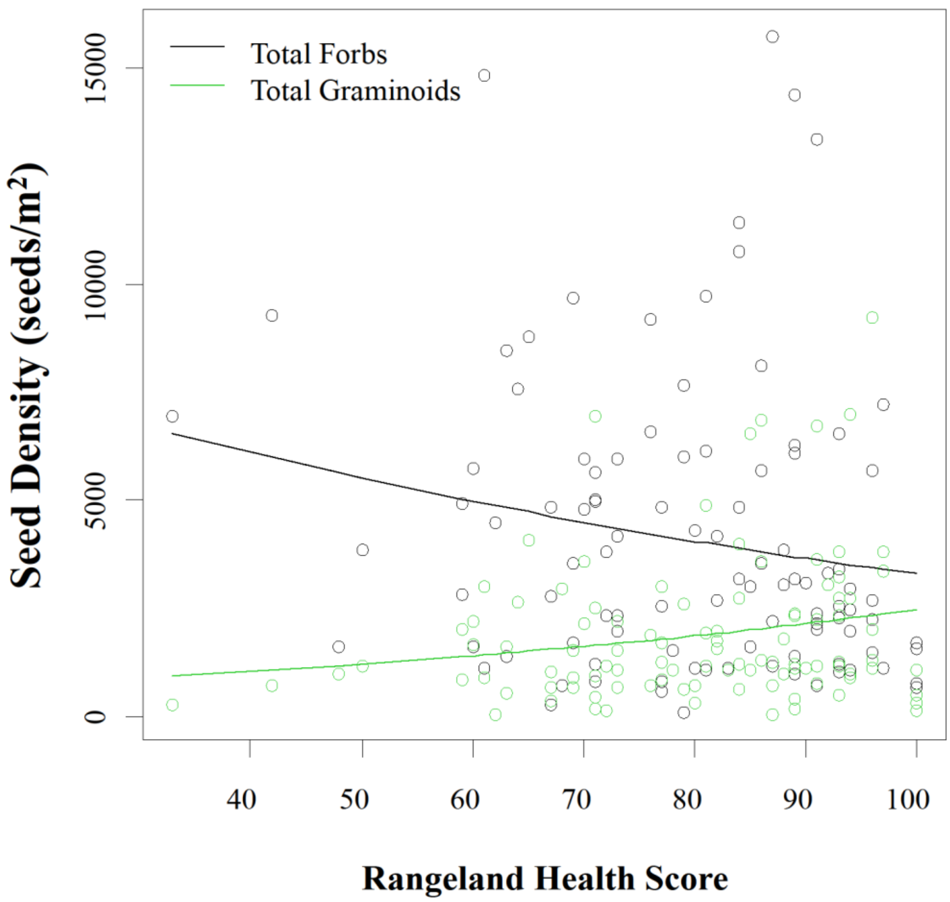
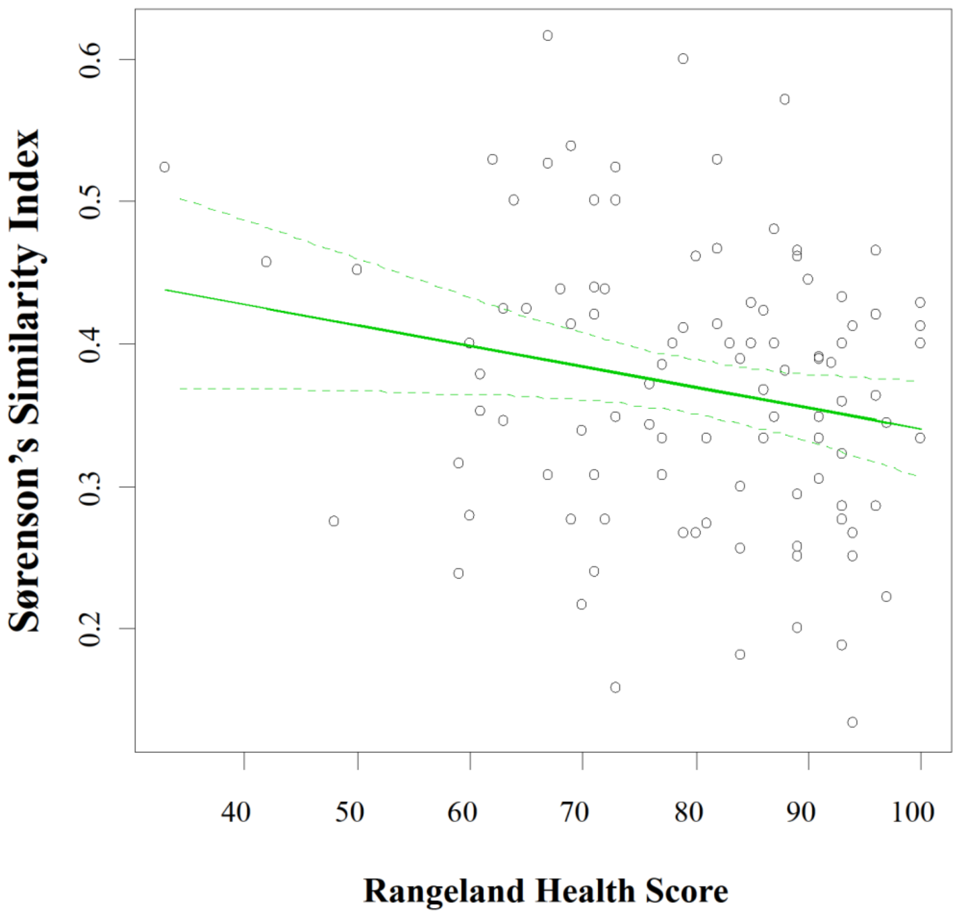


Figure 5.7. Relationships between the seed densities (seeds/m^2) of total forbs and total graminoids in pastures and measured rangeland health (RH) scores from field assessments analyzed with Poisson regression. Graminoids: $\log(y) = 0.0143 + 6.38$, $P < 0.001$; Forbs: $\log(y) = -0.0103x + 9.13$, $P < 0.001$.



.

Figure 5.8. Relationship between similarity (Sørenson Index) of the seed bank and overlying plant community, and associated range health scores determined from the field assessment ($R^2 = 0.03$; $P = 0.043$; 95% CI). The relationship between similarity and RH is: $y = -0.0014x + 0.4859$.

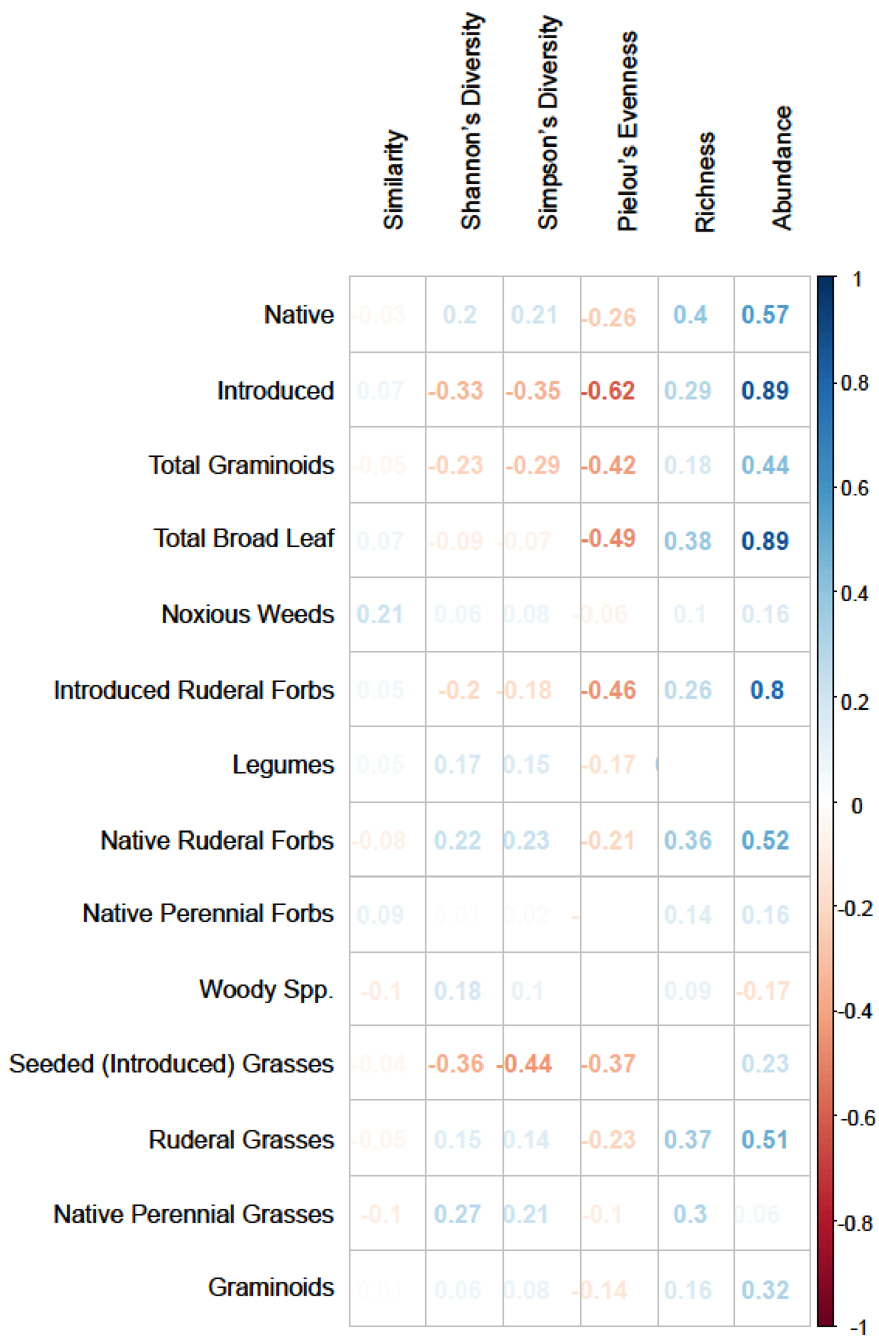


Figure 5.9. Summary table depicting correlations (*r*) in the density of various plant functional groups present in the seed bank of 102 pastures in north central Alberta, and total seed bank density as well as indices of seed bank similarity, diversity, evenness and richness. Only significant correlations are reported, blank cells had no significant relationships.

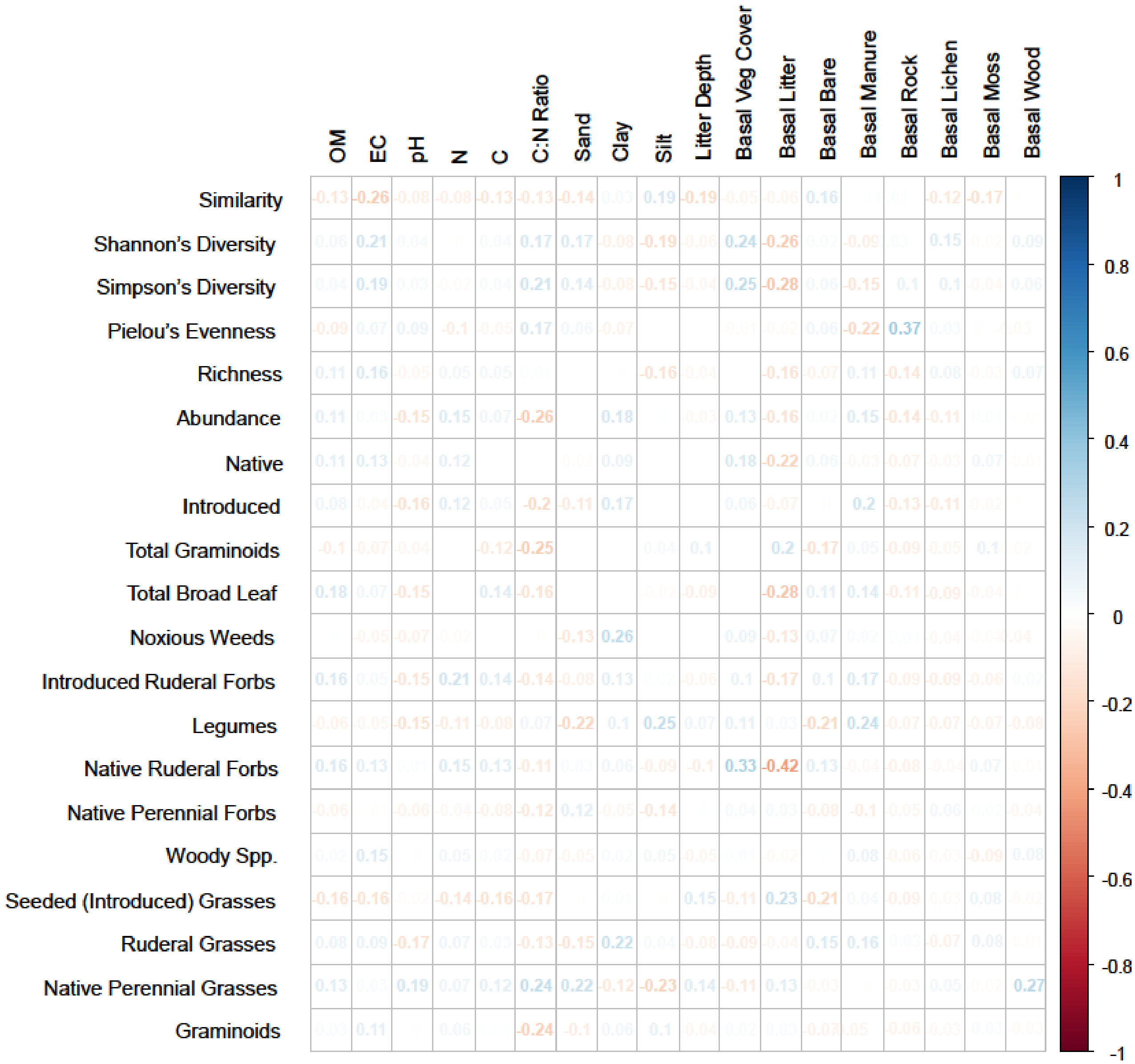


Figure 5.10. Correlations (r) of the relationship between seed density and various soil properties and basal cover characteristics in the above-ground vegetation. Only significant correlations are reported ($P < 0.05$); blank cells had no significant relationships.

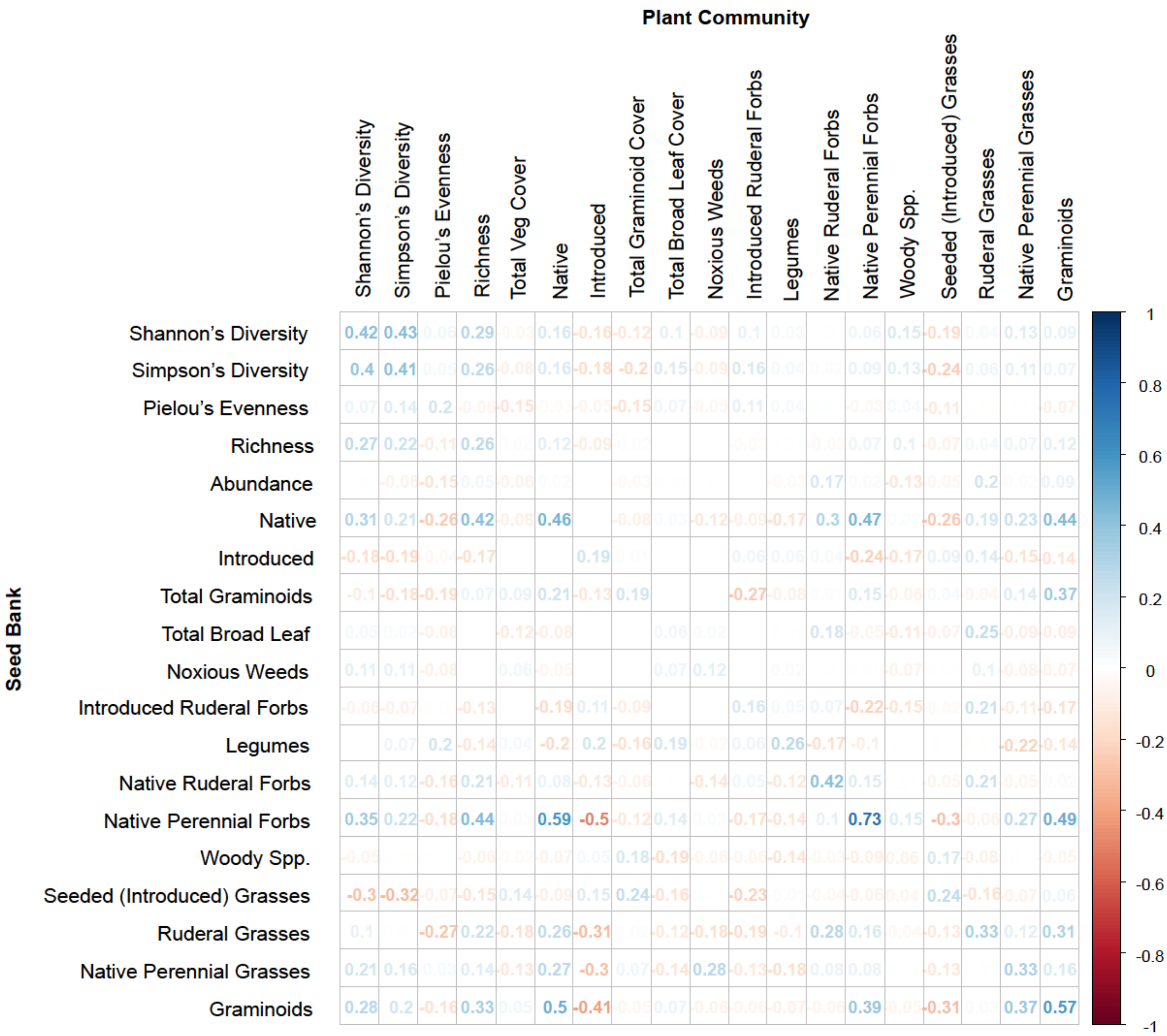


Figure 5.11. Correlations (*r*) of seed bank density and various above-ground plant community metrics, including various cover groupings and diversity indices. Only significant correlations are reported (P < 0.05), blank cells had no significant relationships.

Table 5.1. Summary of the 10 most dominant plant species abundance (and ranks) found in the aboveground community (based on % cover ± 1SD) and the seed bank (based on seed density; seeds/m^2 ± 1SD) of pastures sampled across north central Alberta during 2012 and 2013.

		Plant Community		**Seed Bank**	
Scientific Name	**Common Name**	**Foliar Cover (%)**	**Rank**	**Seed Density (seeds/m^2)**	**Rank**
Bromus biebersteinii Roem. & Schult.	Meadow Brome	5.0 (± 12.1)	**6**	17.5 (± 44.5)	39
Bromus inermis Leyss. subsp. *inermis*	Smooth Brome	13.7 (± 14.9)	**2**	1.6 (± 6.9)	76
Chenopodium album L.	Lamb's Quarters	0.10 (± 0.4)	44	475.2 (± 1652.0)	**4**
Dactylis glomerata L.	Orchardgrass	1.9 (± 6.9)	**10**	1.9 (± 10.4)	75
Elytrigia repens (L.) Gould	Quack Grass	8.1 (± 12.9)	**4**	9.1 (± 23.5)	53
Festuca rubra L.	Red Fescue	3.1 (± 7.7)	**7**	77.8 (± 389.6)	16
Gnaphalium uliginosum L.	Marsh Cudweed	0.00 (± 0.01)	87	454.4 (± 1327.4)	**5**
Medicago sativa L.	Common Alfalfa	2.3 (± 6.8)	**8**	13.3 (± 41.3)	48
Plantago major L.	Common Plantain	0.2 (± 0.8)	32	278.0 (± 528.6)	**6**
Poa palustris L.	Fowl Bluegrass	0.5 (± 1.6)	20	197.0 (± 310.6)	**8**
Poa pratensis L.	Kentucky Bluegrass	25.2 (± 17.3)	**1**	1097.4 (± 1286.3)	**1**
Potentilla norvegica L.	Rough Cinquefoil	0.03 (± 0.1)	64	232.7 (± 465.6)	**7**
Rorippa palustris (L.) Besser subsp. *palustris*	Yellow Cress	-	-	116.8 (± 474.5)	**10**
Taraxacum officinale F.H. Wigg.	Dandelion	8.6 (± 7.6)	**3**	535.7 (± 848.0)	**2**
Thlaspi arvense L.	Stinkweed	0.07 (± 0.4)	52	516.3 (± 1230.9)	**3**
Trifolium hybridum L.	Alsike Clover	2.0 (± 3.1)	**9**	64.0 (± 114.7)	19
Trifolium repens L.	White Clover	6.8 (± 9.9)	**5**	71.3 (± 122.3)	18
Veronica peregrina L.	Neckweed	-	-	186.7 (± 440.6)	**9**

Table 5.2. Results of the perMANOVA tests evaluating seed bank composition responses to individual pasture management factors based on the assessment of 102 sample sites examined across north central Alberta during 2012 and 2013.

Management Factor	Mean Square	F value	R^2	P Value
Owned or Rented	0.428	1.401	0.014	0.102
Previous Cultivation	0.457	1.505	0.030	**0.025**
Grazing System	0.307	1.002	0.020	0.471
Timing of Grazing	0.402	1.323	0.039	**0.048**
Gr. System * Timing of Gr.	0.403	1.331	0.052	**0.032**
Herbivore Type(s)	0.330	1.080	0.043	0.291
Herbicide	0.522	1.713	0.017	**0.032**
Fertilized	0.313	1.020	0.010	0.369
Manure Spreading	0.495	1.624	0.016	**0.037**
Harrowed	0.423	1.383	0.014	0.108
Aerated	0.381	1.244	0.012	0.200
Swathed or Mowed	0.396	1.294	0.013	0.159
Fed Hay in Pasture*	0.554	1.815	0.031	**0.016**
Burrowing Mammals	0.321	1.047	0.010	0.403
Fire (Survey)	0.407	1.331	0.013	0.130
Fire (Charcoal in Soil)	0.667	2.202	0.022	**0.007**
Grazing Intensity	0.313	1.021	0.051	0.422

Distance = Bray-Curtis, Permutations = 999

*Includes only 58 sites from the 2013 survey

Table 5.3. Results of the perMANOVA contrasts assessing management factor impacts on pasture seed bank composition.

Management	Contrast	Mean Square	F value	R^2	P Value
Previous Cultivation	Cultivated vs Never	0.52	1.74	0.02	**0.027**
	Cultivated vs Unknown	0.39	1.27	0.01	0.179
	Never vs Unknown	0.51	1.57	0.06	**0.028**
Timing of Grazing	Abandoned vs Year Rd	0.38	1.11	0.10	0.290
	Abandoned vs Growing Seas.	0.36	1.19	0.01	0.231
	Abandoned vs Winter	0.33	1.11	0.18	0.307
	Growing Seas. vs Year Rd.	0.40	1.31	0.01	0.137
	Growing Seas. vs Winter	0.44	1.48	0.02	0.069
	Year Rd. vs Winter	0.51	1.54	0.15	0.061
Grazing System * Timing of Gr.	Cont.+Y vs Abandoned	0.38	1.11	0.10	0.337
	Cont.+Y vs Cont.+G	0.53	1.77	0.04	**0.022**
	Cont.+Y vs Rotat.+G	0.32	1.02	0.02	0.415
	Cont.+Y vs Rotat.+W	0.51	1.54	0.15	0.067
	Cont.+G vs Abandoned	0.30	1.03	0.03	0.381
	Cont.+G vs Rotat.+G	0.41	1.36	0.02	0.106
	Cont.+G vs Rotat.+W	0.40	1.39	0.04	0.133
	Rotat.+G vs Abandoned	0.40	1.30	0.02	0.157
	Rotat.+G vs Rotat.+W	0.47	1.54	0.03	**0.044**
	Rotat.+W vs Abandoned	0.33	1.11	0.18	0.308

Grazing System: A = Abandoned, Cont. = Continuous, Rotat. = Rotational
Grazing Timing: A=Abandoned, G = Growing Season, W = Winter, Y= Year Round
Distance = Bray-Curtis, Permutations = 999
Bold: P < 0.05, Italics: P < 0.10, Grey: P > 0.10

Table 5.4. Summary of the indicator species analysis relating seed bank species composition to each of the management factors. Only those species with significance at P < 0.01 are shown. For a more complete list of species, see Table C.6.1.

Manage. Factor	Treatment Category	Species	A	B	P value
Cultivated	Never	*Achillea millefolium*	0.73	0.75	0.001
		Campanula rotundifolia	0.94	0.38	0.002
		Carex praticola	0.97	0.25	0.006
		Cerastium arvense	0.96	0.50	0.001
		Agrostis scabra	0.90	0.63	0.002
		Penstemon procerus	0.99	0.38	0.004
		Solidago canadensis	0.97	0.25	0.007
System	None (Abandoned)	*Medicago sativa*	0.75	0.75	0.008
		Danthonia intermedia	0.83	0.50	0.008
		Solidago canadensis	0.93	0.25	0.011
Timing	Winter Grazed	*Festuca rubra*	0.87	1.00	0.009
	Abandoned+Winter	*Medicago sativa*	0.88	0.71	0.005
System x Timing	Abandoned	*Danthonia intermedia*	0.85	0.50	0.008
	Winter Grazed	*Phleum pratense*	0.53	1.00	0.009
		Polygonum lapathifolium	0.80	0.67	0.009
Herbivores	Sheep/Alpaca	*Sonchus arvensis*	0.90	0.75	0.001
Manured	Manure Spread	*Chenopodium salinum*	0.95	0.60	0.001
Harrowed	Harrowed	*Chenopodium salinum*	0.80	0.44	0.003
		Polygonum lapathifolium	0.85	0.32	0.010
	Not Harrowed	*Bromus biebersteinii*	0.90	0.26	0.010
Hay In Pasture	Animals Fed Hay	*Chenopodium album*	0.76	0.75	0.006
		Urtica dioica	0.90	0.50	0.007
	No Hay	*Carex* Spp.	0.68	0.63	0.003
	Unknown	*Carex aenea*	1.00	0.23	0.004
Recent Fire	Fire (Survey)	*Bromus anomalus*	0.94	0.27	0.002
		Geranium bicknellii	0.89	0.33	0.010
		Sonchus arvensis	0.74	0.40	0.011
		Typha latifolia	0.93	0.33	0.002
Historical Fire	No Fire	*Gnaphalium uliginosum*	0.96	0.63	0.001
	Fire (Charcoal in Soil)	*Carex bebbii*	1.00	0.19	0.001

ISA was ran in R using *indicspecies:multipatt* (Caceres and Legendre, 2009).
A = Probability of a species occurring, B = Fidelity for that class

Table 5.5. Results of the indicator species analysis identifying those vegetation functional groups found in the seed bank that were associated with each pasture management factor assessed in the survey, as well as rangeland health measured in the field (P < 0.10).

Management Factor	Treatment Category	Plant Functional Group	A	B	P value
Ownership	Rented	Native Perennial Forbs	0.67	0.80	0.057
		Native Perennial Grasses	0.76	0.70	**0.007**
Cultivation	Not Cultivated	Graminoids	0.70	1.00	**0.003**
		Native Perennial Forbs	0.76	1.00	**0.002**
		Native Perennial Grasses	0.62	0.63	**0.041**
	Unknown History	Introduced Ruderal Forbs	0.56	1.00	**0.003**
		Introduced Species	0.48	1.00	**0.008**
		Native Ruderal Forbs	0.49	1.00	0.059
		Total Broad Leaf Plants	0.50	1.00	**0.009**
Grazing System	None (Abandoned)	Graminoids	0.75	0.75	0.062
		Total Grasses + Graminoids	0.50	1.00	**0.045**
	Continuous + Rotational	Noxious Weeds	1.00	0.70	**0.037**
	Continuous + None	Native Perennial	0.90	0.64	**0.049**
Timing of Grazing	Abandoned	Graminoids	0.78	0.75	0.084
	Winter	Introduced Species	0.40	1.00	0.072
		Legumes	0.48	1.00	0.080
		Seeded (Introduced) Grasses	0.59	1.00	**0.003**
		Total Grasses + Graminoids	0.50	1.00	**0.015**
	Abandoned + All Year + Winter	Legumes	0.88	1.00	0.002
Gr. System x Timing of Gr.	Continuous	Noxious Weeds	0.68	0.73	0.098
	Abandoned	Graminoids	0.75	0.75	0.061
		Legumes	0.47	1.00	0.084
		Total Grasses + Graminoids	0.50	1.00	**0.050**
	Continuous + Rotational	Noxious Weeds	0.98	0.70	**0.029**
	Continuous + Wildlife	Native Perennial Forbs	0.90	0.61	0.057
Aerated	Aerated	Legumes	0.77	1.00	**0.017**
	Not Aerated	Seeded (Introduced) Grasses	0.78	1.00	**0.033**
Feeding Hay	Fed Hay	Introduced Ruderal Forbs	0.46	1.00	**0.022**
		Introduced Species	0.41	1.00	0.072
		Total Broad Leaf Plants	0.43	1.00	**0.044**
	No Hay	Graminoids	0.56	0.91	**0.048**
		Native Species	0.45	1.00	0.084
Rangeland Health					
Plant Community Type	Modified-Tame	Graminoids	0.80	1.00	**0.001**
		Native Perennial Forbs	0.92	0.92	**0.001**
		Native Perennial Grasses	0.70	0.58	0.061
Grazing Intensity	U	Graminoids	0.54	0.75	0.085
		Total Grasses + Graminoids	0.28	1.00	0.083

A = Probability of occurring, B = Fidelity

Table 5.6. Results of the ANOVA tests results reporting effects of various pasture management factors on the seed density (seeds/m^2) of primary plant groupings, including total SB density, based on data collected across 102 pastures in north central Alberta during 2012 and 2013.

	Graminoids		**Broad Leaf**		**Native**		**Introduced**		**Total**	
Management	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Owned or Rented	0.285	0.595	0.238	0.627	0.102	0.750	2.226	0.139	1.322	0.253
Previous Cultivation	1.035	0.359	1.643	0.199	2.738	0.070	**4.626**	**0.012**	2.587	0.080
Grazing System	1.530	0.222	0.095	0.910	0.079	0.924	0.020	0.981	0.412	0.663
Timing of Grazing	**7.6121**	**<0.001**	0.114	0.952	1.277	0.287	2.299	0.082	1.874	0.139
Grazing System x Timing of Gr.	**7.0448**	**<0.001**	0.113	0.978	0.988	0.418	1.755	0.144	1.491	0.211
Herbivore Type(s)	1.783	0.139	0.982	0.421	0.097	0.983	1.123	0.350	1.411	0.236
Herbicide	0.194	0.661	2.890	0.092	3.698	0.057	0.107	0.744	0.211	0.647
Fertilized	0.499	0.482	**10.788**	**0.001**	**6.849**	**0.010**	0.612	0.436	2.110	0.150
Manure Spreading	3.149	0.079	0.665	0.417	0.055	0.815	0.059	0.809	0.083	0.774
Harrowed	0.541	0.464	0.027	0.871	0.129	0.720	0.001	0.976	0.220	0.640
Aerated	0.905	0.344	0.160	0.690	0.058	0.810	0.058	0.810	0.007	0.935
Swathed or Mowed	0.139	0.710	0.200	0.655	0.321	0.572	0.494	0.484	0.024	0.877
Fed Hay in Pasture Sampled*	1.951	0.168	0.005	0.947	1.466	0.231	1.466	0.231	0.082	0.775
Burrowing Mammals	0.730	0.395	2.590	0.111	0.006	0.941	1.062	0.305	0.879	0.351
Fire (Survey)	0.073	0.788	2.205	0.141	0.000	0.983	1.726	0.192	2.119	0.149
Fire (Charcoal in Soil)	0.075	0.784	**14.930**	**<0.001**	**6.584**	**0.012**	**5.011**	**0.027**	**9.886**	**0.002**
Rangeland Health										
Plant Community Type	0.271	0.604	0.098	0.755	3.601	0.061	**7.856**	**0.006**	0.332	0.566
Grazing Intensity	1.009	0.417	1.254	0.290	0.184	0.968	0.926	0.468	1.039	0.400
Health	1.611	0.205	0.845	0.433	0.471	0.626	0.024	0.976	0.034	0.966

Bold: $P < 0.05$, Black: $P < 0.10$, Grey: $P > 0.10$

*Includes only 58 sites from the 2013 survey

Table 5.7. Comparison of mean (±SE) seed bank density (seeds/m^2) of primary plant groupings in relation to different management factors and range health criteria. Data are based on 102 pasture sampled across north central Alberta during 2012 and 2013.

Management	Treatment	Graminoids	Broadleaf	Native	Introduced	Total
Cultivation	Cultivated			1299.3 (±188.7)	4237.9 (±342.6) ab	
	Never Cultivated			3205.3 (±585.4)	2177.6 (±1062.9) b	
	Unknown			2338.3 (±188.7)	5882.1 (±729.1) a	
Timing of Grazing	Abandoned	3628.3 (±752.9) ab			4385.0 (±1513.8)	
	All Year	580.9 (±532.4) c			5088.0 (±1070.4)	
	Growing Season	1777.8 (±161.4) b			4119.2 (±324.6)	
	Winter	5957.8 (±869.4) a			9040.0 (±1748.0)	
Grazing System x Timing	Abandoned	3628.3 (±742.2) a				
	All Year (Continuous)	580.9 (±524.9) c				
	Growing Season (Continuous)	2187.3 (±262.4) ab				
	Growing Season (Rotational)	1539.5 (±200.2) b				
	Winter (Rotational)	5957.8 (±857.0) a				
Herbicide	Sprayed in Last 3 Years		3637.2 (±846.1)	1088.8 (±435.0)		
	Not Sprayed Recently		4181.3 (±365.0)	1721.1 (±187.6)		
Fertilized	Fertilized		2118.3 (±1111.0) b	834.1 (±579.3) b		
	Not Fertilized		4287.3 (±345.6) a	1698.2 (±180.2) a		
Manure Spreading	Manured	1380.3 (±340.9)				
	Not Manured	2041.4 (±194.2)				
Fire (Charcoal in Soil)	Present		2535.3 (±579.5) b	1124.7 (±309.6) b	3180.3 (±542.5) b	4308.8 (±647.8) b
	Absent		4777.3 (±382.9) a	1839.0 (±204.6) a	4861.2 (±358.5) a	6704.0 (±428.0) a
Rangeland Health	**Score**					
Plant Community Type	Modified-Tame			2770.4 (±491.8)	2363.3 (±875.4) b	
	Tame			1468.8 (±179.6)	4615.3 (±319.7) a	

Black: $P < 0.05$, Grey: $P < 0.10$

Table 5.8. Results of the ANOVA test results reporting effects of various pasture management factors on the seed density (seeds/m^2) of specified functional plant groupings, based on data collected across 102 pastures in north central Alberta during 2012 and 2013.

	Native & Introduced				Introduced						Native											
	Legumes		Ruderal Grasses		Noxious Weeds		Ruderal Forbs		Seeded Graminoids		Ruderal Forbs		Perennial Forbs		Perennial Grasses		Graminoids		Woody Spp.	
Management	**X^2**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**
Owned or Rented	0.403	0.526	0.754	0.387	2.070	0.150	0.075	0.785	0.035	0.853	0.000	0.991	2.746	0.098	**5.960**	**0.015**	3.102	0.078	1.051	0.305
Previous Cultivation	1.487	0.476	1.667	0.194	0.202	0.903	**5.815**	**0.004**	0.259	0.773	1.358	0.262	**12.533**	**0.002**	3.936	0.140	4.211	0.122	2.189	0.335
Grazing System	2.828	0.243	0.043	0.958	4.548	0.099	0.344	0.710	0.571	0.567	0.883	0.417	3.213	0.201	0.953	0.621	0.202	0.904	2.420	0.298
Timing of Grazing	**9.330**	**0.025**	1.705	0.171	5.147	0.161	0.477	0.699	**3.723**	**0.014**	1.061	0.370	1.426	0.700	3.336	0.343	5.771	0.123	0.825	0.844
Grazing System x Timing of Gr.	**10.919**	**0.027**	1.524	0.201	5.208	0.267	0.520	0.721	**3.620**	**0.009**	0.805	0.525	4.399	0.355	3.427	0.489	5.938	0.204	2.709	0.608
Herbivore Type(s)	3.744	0.442	1.540	0.197	6.067	0.194	0.750	0.560	0.697	0.596	0.512	0.727	4.642	0.326	7.563	0.109	2.546	0.636	1.276	0.865
Herbicide	2.022	0.155	1.089	0.299	1.578	0.209	0.176	0.676	0.801	0.373	3.086	0.082	0.744	0.388	0.809	0.368	0.006	0.941	0.316	0.574
Fertilized	**4.597**	**0.032**	0.064	0.802	**4.971**	**0.026**	2.061	0.154	1.924	0.169	**5.339**	**0.023**	1.562	0.211	0.045	0.831	**6.993**	**0.008**	0.042	0.837
Manure Spreading	0.101	0.751	0.064	0.800	0.101	0.751	0.752	0.388	2.058	0.155	3.204	0.076	**4.160**	**0.041**	**5.223**	**0.022**	1.011	0.315	1.212	0.271
Harrowed	0.033	0.855	0.742	0.391	0.588	0.443	0.498	0.482	0.389	0.535	0.021	0.884	2.106	0.147	0.205	0.650	0.220	0.639	2.483	0.115
Aerated	**4.374**	**0.036**	0.755	0.387	1.167	0.280	0.173	0.678	2.117	0.149	0.001	0.996	<0.001	0.993	0.341	0.559	0.600	0.436	0.447	0.504
Swathed or Mowed	1.646	0.200	0.326	0.569	1.132	0.287	0.015	0.904	0.262	0.610	0.052	0.821	2.325	0.127	0.000	0.984	2.307	0.129	1.059	0.303
*Fed Hay in Pasture Sampled	0.161	0.688	1.466	0.231	0.001	0.979	0.638	0.428	0.068	0.795	0.128	0.722	1.003	0.316	0.752	0.386	3.410	0.065	3.397	0.065
Burrowing Mammals	0.274	0.601	0.002	0.967	3.084	0.079	2.926	0.090	0.727	0.396	0.336	0.564	0.981	0.322	1.391	0.238	0.140	0.708	0.321	0.571
Fire (Survey)	0.530	0.467	0.011	0.917	0.053	0.817	1.348	0.248	0.002	0.967	0.154	0.696	0.992	0.319	0.641	0.423	0.779	0.378	0.162	0.687
Fire (Charcoal in Soil)	2.597	0.107	0.507	0.478	0.000	0.988	**6.621**	**0.012**	0.513	0.475	**7.434**	**0.008**	0.207	0.649	0.004	0.948	0.112	0.738	0.359	0.549
Rangeland Health	**X^2**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**
Plant Community Type	0.414	0.520	0.183	0.670	0.979	0.322	**13.198**	**0.0004**	0.392	0.533	0.089	0.766	**16.127**	**0.0001**	3.504	0.061	3.643	0.056	0.054	0.817
Grazing Intensity	**11.626**	**0.040**	0.193	0.965	9.715	0.084	1.322	0.261	0.646	0.666	0.616	0.688	3.104	0.684	5.794	0.327	2.122	0.832	2.976	0.704
Health	3.695	0.158	0.461	0.632	1.877	0.391	0.559	0.574	3.004	0.054	0.896	0.412	0.382	0.826	1.969	0.374	0.833	0.659	1.393	0.498

Bold: P < 0.05, Black: P < 0.10, Grey: P > 0.10
*Includes only 58 sites from the 2013 survey
Note noxious weeds includes 1 graminoid species

Table 5.9. Comparison of mean (±SE) seed bank density (seeds/m^2) of specified functional plant groupings in relation to different management factors and range health criteria. Data are based on 102 pastures sampled across north central Alberta during 2012 and 2013.

		Native & Introduced		Native						
Management	**Treatment**	**Legumes**	**Noxious Weeds**	**Ruderal Forbs**	**Seeded Grasses**	**Ruderal Forbs**	**Perennial Forbs**	**Perennial Grasses**	**Graminoids**	**Woody Spp.**
Ownership	Owned						133.7 (±46.9)	35.2 (±9.5) b	310.8 (±73.7)	
	Rented						274.1 (±142.2)	112.0 (±28.8) a	54.8 (±223.4)	
Cultivation	Cultivated			2553.7 (±316.8) a			83.6 (±47.6) b			
	Never Cultivated			944.3 (±982.8) b			744.7 (±147.5) a			
	Unknown			4421.4 (±674.2) a			155.6 (±101.2) b			
Grazing System	Abandoned (None)		6.0 (±233.8)							
	Continuous		252.6 (±69.7)							
	Rotational		114.7 (±64.2)							
Timing of Grazing	Abandoned	268.1 (±95.3) ab			1864.8 (±568.4) ab					
	All Year	172.8 (±67.4) ab			452.8 (±464.1) b					
	Growing Season	137.8 (±20.4) b			1161.9 (±119.8) b					
	Winter	540.2 (±110.1) a			7375.8 (±803.8) a					
Grazing System x Timing of Gr.	Abandoned	268.1 (±93.8) ab			1864.8 (±560.7) ab					
	All Year (Cont.)	172.8 (±66.4) ab			452.8 (±457.8) b					
	Grow. Season (Cont.)	84.2 (±33.2) b			1421.9 (±179.6) b					
	Grow. Season (Rot.)	169.0 (±25.3) ab			963.1 (±157.0) b					
	Winter (Rot.)	540.2 (±108.4) a			7375.8 (±792.9) a					
Herbicide	Sprayed in Last 3 Years					570.5 (±242.0)				
	Not Sprayed Recently					935.0 (±104.4)				
Fertilized	Fertilized	39.7 (±66.2) b	18.5 (±156.3) b			410.4 (±322.0) b			26.5 (±235.3) b	
	Not Fertilized	168.9 (±20.6) a	186.0 (±48.6) a			923.0 (±100.2) a			310.8 (±73.2) a	
Manure Spreading	Manured					1233.5 (±191.1)	30.5 (±89.3) b	22.9 (±18.7) b		
	Not Manured					762.3 (±108.9)	185.4 (±50.9) a	49.2 (±10.6) a		
Aerated	Aerated	476.6 (±95.7) a								
	Not Aerated	144.4 (±19.3) b								
Fed Hay in Pasture Sampled	Hay								95.3 (±215.9)	11.1 (±4.2)
	No Hay								426.2 (±127.5)	1.1 (±2.5)
Burrowing Mammals	Present		177.9 (±60.9)	3085.8 (±370.6)						
	Absent		161.7 (±72.7)	2243.0 (±442.9)						
Fire (Charcoal in Soil)	Present			1562.9 (±501.6) b		607.3 (±172.5) b				
	Absent			3252.1 (±331.4) a		995.9 (±114.0) a				
Rangeland Health	**Score**									
Plant Community Type	Modified-Tame			1227.3 (±821.8) b			740.8 (±114.1) a	87.4 (±26.7)	834.1 (±196.7)	
	Tame			2940.2 (±300.1) a			68.3 (±41.7) b	36.8 (±9.8)	212.6 (±71.8)	
Grazing Intensity	U	268.1 (±98.3) a	6.0 (±237.3)							
	L	225.1 (±65.5) a	113.9 (±158.2)							
	LM	70.5 (±40.1) b	66.5 (±96.9)							
	M	142.3 (±33.7) ab	230.6 (±81.4)							
	MH	192.7 (±41.0) ab	219.7 (±99.0)							
	H	250.2 (±69.5) ab	241.3 (±167.8)							
Health	Healthy				1421.7 (±174.3)					
	Healthy with Problems				1058.6 (±256.2)					
	Unhealthy				357.5 (±713.2)					

Black: P < 0.05, Grey: P < 0.10

Table 5.10. Results of the ANOVA tests reporting effects of various pasture management factors on seed bank similarity (to aboveground vegetation), richness, diversity, and evenness, based on data collected across 102 pastures in north central Alberta during 2012 and 2013.

	Sørenson's Similarity		Seed Bank: Richness		Seed Bank: Shannon's Diversity		Seed Bank: Simpson's Diveristy		Seed Bank: Pielou's Evenness	
Management	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Owned or Rented	0.497	0.483	0.056	0.813	0.690	0.408	0.907	0.343	0.218	0.642
Previous Cultivation	0.198	0.821	0.147	0.863	0.377	0.687	0.438	0.647	1.552	0.217
Grazing System	0.430	0.652	0.929	0.399	**3.712**	**0.028**	**4.318**	**0.016**	0.519	0.597
Timing of Grazing	1.349	0.263	1.847	0.144	2.347	0.077	**2.804**	**0.044**	1.917	0.132
Grazing System x Timing of Gr.	1.008	0.407	1.374	0.249	2.201	0.074	**2.960**	**0.024**	1.723	0.151
Herbivore Type(s)	0.971	0.427	0.813	0.520	**3.244**	**0.015**	**3.216**	**0.016**	0.685	0.604
Herbicide	0.391	0.533	**6.725**	**0.011**	**13.674**	**<0.001**	**13.952**	**<0.001**	1.361	0.246
Fertilized	0.489	0.486	2.185	0.143	1.846	0.177	0.766	0.384	0.066	0.798
Manure Spreading	1.040	0.310	0.120	0.730	2.703	0.103	**3.968**	**0.049**	1.924	0.169
Harrowed	0.664	0.417	0.193	0.662	0.903	0.344	1.306	0.256	1.295	0.258
Aerated	0.732	0.394	2.695	0.104	**4.079**	**0.046**	**4.383**	**0.039**	0.137	0.712
Swathed or Mowed	0.082	0.775	0.466	0.496	0.244	0.623	0.109	0.742	0.645	0.424
Fed Hay in Pasture Sampled*	0.387	0.536	0.593	0.445	1.771	0.189	2.532	0.117	0.053	0.819
Burrowing Mammals	0.042	0.838	1.015	0.316	0.155	0.694	0.004	0.950	0.805	0.372
Fire (Survey)	0.001	0.971	0.014	0.906	0.848	0.359	1.029	0.313	0.087	0.768
Fire (Charcoal in Soil)	0.000	0.987	1.761	0.188	0.000	0.992	0.009	0.923	1.943	0.166
Rangeland Health	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Plant Community Type	0.233	0.630	0.007	0.931	0.286	0.594	0.206	0.651	0.000	0.985
Grazing Intensity	1.108	0.631	1.312	0.265	1.727	0.136	1.821	0.116	0.653	0.660
Health	1.655	0.196	0.000	1.000	0.532	0.589	0.737	0.481	0.443	0.644

Bold: P < 0.05, Black: P < 0.10, Grey: P > 0.10

*58 sites from the 2013 survey

Table 5.11. Comparison of mean (±SE) seed bank richness and diversity in relation to different management factors and range health criteria. Data are based on 102 pastures sampled across north central Alberta during 2012 and 2013.

Management	Treatment	Richness	Shannon's Diversity	Simpson's Diversity
Grazing System	Abandoned (None)		1.5 (±0.2) b	0.63 (±0.07) b
	Continuous		2.0 (±0.1) ab	0.74 (±0.02) ab
	Rotational		2.1 (±0.1) a	0.78 (±0.02) a
Timing of Grazing	Abandoned		1.5 (±0.2)	0.63 (±0.72) b
	All Year		1.9 (±0.2)	0.75 (±0.05) ab
	Growing Season		2.1 (±0.1)	0.77 (±0.02) a
	Winter		1.8 (±0.3)	0.66 (±0.08) b
Grazing System x Timing of Gr.	Abandoned		1.5 (±0.2)	0.63 (±0.07) b
	All Year (Cont.)		1.9 (±0.2)	0.75 (±0.05) ab
	Continuous		2.0 (±0.1)	0.75 (±0.03) ab
	Rotational		2.1 (±0.1)	0.78 (±0.02) a
	Winter (Rot.)		1.8 (±0.2)	0.66 (±0.08) ab
Herbivore Type(s)	Cattle		2.1 (±0.1) ab	0.77 (±0.02) ab
	Horses		1.8 (±0.1) ab	0.70 (±0.04) ab
	Multiple Species		2.4 (±0.2) a	0.85 (±0.06) a
	Other		1.8 (±0.2) ab	0.72 (±0.07) ab
	Wildlife (None)		1.5 (±0.2) b	0.63 (±0.07) b
Herbicide	Sprayed in Last 3 Years	17.8 (±1.6) b	1.6 (±0.1) b	0.64 (±0.03) b
	Not Sprayed Recently	22.4 (±0.7) a	2.1 (±0.1) a	0.78 (±0.01) a
Manure Spreading	Manured			0.81 (±0.03) a
	Not Manured			0.74 (±0.02) b
Aerated	Aerated		2.5 (±0.2) a	0.87 (±0.07) a
	Not Aerated		2.0 (±0.1) b	0.76 (±0.01) b

Table 5.12. Results of the perMANOVA tests evaluating seed bank composition responses to rangeland health categories based on the assessment of 102 sample sites examined across north central Alberta during 2012 and 2013.

RHA Category	Mean Square	F-Stat	R^2	P Value
Plant Community Type	0.656	2.163	0.021	**0.009**
Forage Cover	0.247	0.803	0.016	0.802
Cover of Tall Productive Forages	0.432	1.421	0.028	**0.045**
Weedy & Ruderal Cover	0.276	0.898	0.009	0.578
Hydraulic Function & Litter	0.376	1.234	0.036	0.125
Soil Erosion	0.485	1.601	0.031	**0.012**
Anthropogenic Bare Soil	0.454	1.494	0.029	**0.019**
Noxious Weed Cover	0.340	1.111	0.022	0.295
Noxious Weed Density	0.333	1.088	0.032	0.288
Woody Spp Cover	0.538	1.767	0.017	**0.027**
Woody Spp Density	0.442	1.455	0.029	**0.040**
Grazing Intensity	0.313	1.021	0.051	0.422
Health	0.307	1.002	0.020	0.466

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

Distance = Bray-Curtis, Permutations = 999

Table 5.13. Results of the perMANOVA contrasts of range health categories relationships with seed bank composition.

Rangeland Health Category	Scores	Mean Square	F Model	R^2	P Value
Cover of Tall Productive Forages	0 vs 7	0.45	1.40	0.08	0.084
	0 vs 14	0.52	1.73	0.02	**0.035**
	7 vs 14	0.36	1.17	0.01	0.242
Soil Erosion	4 vs 7	0.50	1.57	0.03	**0.045**
	4 vs 10	0.32	1.11	0.02	0.288
	7 vs 10	0.59	1.96	0.02	**0.011**
Anthropogenic Bare Soil	0 vs 3	0.53	1.54	0.07	**0.048**
	0 vs 5	0.76	2.54	0.03	**0.003**
	3 vs 5	0.17	0.56	0.01	0.937
Woody spp. Density	0 vs 2	0.28	0.96	0.03	0.485
	0 vs 4	0.53	1.77	0.02	**0.025**
	2 vs 4	0.39	1.27	0.02	0.162

Grazing System: A = Abandoned, C = Continuous, R = Rotational

Grazing Timing: A=Abandoned, G = Growing Season, W = Winter, Y= All Year

Distance = Bray-Curtis, Permutations = 999

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

Study II

Chapter 6

Legacy effects of pipeline disturbance on seed bank composition, associated aboveground vegetation and biological soil crusts in Dry Mixedgrass prairie

6.1 Abstract

Native grasslands are sensitive to industrial disturbances like pipeline construction that remove established native plants and disturb the soil profile. Given the severe impact to vegetation, post disturbance recovery of the plant community typically occurs from the accumulation of propagules in the soil seed bank and any active re-vegetation efforts. Germinable soil seed banks (SBs), plant communities (PCs), and biological soil crusts (BSCs) in Dry Mixedgrass (DMG) prairie were examined for potential legacy effects on 18 pipelines, varying from 66.3 to 1067 mm in diameter, installed between 1960 and 2007. Sampling occurred at various distances from the pipeline center (i.e. trench), across the adjacent work area, and up to 55 m away from the trench center within the native grassland. Disturbance legacies were found as represented by significantly altered plant community composition, diversity and biomass, both near the trench and within the adjacent right-of-way, and coincided with residual effects on soil properties, including higher salinity and pH along trenches. Trenches were associated with greater plant and litter biomass, which was attributed to introduced plants like *Melilotus* spp. and cool-season grasses. SBs lacked the distinct shifts in composition evident in the aboveground PC as the legacy effect of pipelines extended to further distances (due to seed dispersal and persistence) and could have resulted in more homogeneity. Notable effects on seed bank composition included greater densities of native forbs associated with coarse-textured soils ($P = 0.009$), reduced native graminoid seed densities along pipeline trenches with loam soils ($P = 0.015$), and high seed densities of *Melilotus* and native grasses commonly selected for reclamation along pipeline trenches. Wider diameter pipelines were often associated with weedy, introduced seed banks. Similarity in richness between the seed bank and vegetation was low, averaging 25.2% across all sampling distances. BSCs remained markedly divergent in the trench and

work areas associated with pipeline installation relative to adjacent native grassland, suggesting strong legacy effects of disturbance on this component of grasslands. Collectively, this research indicates pipelines have distinct residual effects on PCs, their underlying SBs, and in particular, BSCs, even several decades after pipeline installation.

6.2 Introduction

Oil and gas development is common across the Northern Great Plains and is known to alter native grassland composition and function such as grassland biodiversity, soils and nutrient cycling, and vegetation structure (Desserud et al. 2010; Desserud and Naeth 2011; Desserud and Naeth 2013; Elsinger 2009; Hammermeister et al. 2003; Hickman et al. 2010; Nannt 2014; Ostermann 2001; Petherbridge 2000). Such infrastructure increases habitat fragmentation and the invasibility of communities (Allred et al. 2015). In North America, an average of 50 000 new wells have been constructed per year since 2000. On rangelands supporting livestock production, this loss is equivalent to 5 million animal-unit-months (AUM) of grazing opportunities (Allred et al. 2015). Native prairie is a threatened ecosystem (Samson et al. 2004) that has undergone significant loss since European settlement, primarily through cultivation (Gauthier and Wiken 2003), while intact patches are susceptible to further degradation from oil and gas developments and other disturbances (i.e. gravel extraction, roads, etc.).

Disturbances such as pipelines and well-sites can provide an opportunity for invasive species to establish, reduce species richness and the density of native plants, function as corridors for seed dispersal (D'Antonio and Meyerson 2002; Ostermann 2001), and facilitate further invasion. In Alberta, pipeline reclamation practices have evolved. Prior to 1972, disturbance associated with pipelines and well-sites were typically allowed to recover naturally (Gramineae Services Ltd. 2013). Natural recovery entails revegetation through plant establishment from residual plant material [e.g., root fragments (Hamduon 1972) or bud bank (Klimes 2007)] or from the seed bank, including seed rain sourced from the adjacent community (Hutchings and Booth 1996). From 1972 to 1985, reclamation practices were developed that emphasized soil conservation and revegetation using agronomic species to ensure rapid establishment of protective ground cover and included species such as crested wheatgrass (*Agropyron cristatum*) and sweet clover (*Melilotus* spp.) (Gramineae Services Ltd. 2013); this process reduced unwanted weedy vegetation and promptly restored vegetation for other land uses such as grazing. After 1993, native grassland ecological function and integrity became a focus of reclamation in many areas, and seed mixes using native plant species found in the reference community are now mandated on public land (Government of Alberta 2001), and also recommended for affected private land containing native vegetation (Gramineae Services Ltd. 2013). However, even with the use of native plant cultivars, communities can fail to recover in composition, diversity, structure, and function, relative to the historical non-disturbed reference community (Hammermeister et al. 2003; Simmers and Galatowitsch 2010), potentially because seeded cultivars are capable of out-competing their wild genotypic relatives (Schröder and Prasse 2013), and in all likelihood, other native vegetation. In the Dry Mixedgrass Prairie for example, pipelines and well-sites seeded with native species are often characterized by taller grasses like green needlegrass (*Nassella viridula*) and slender wheatgrass (*Elymus trachycaulus* ssp. *trachycaulus*), and therefore lack resemblance to the *Hesperostipa-Agropyron* or *Hesperostipa-Bouteloua* communities they should emulate (Adams et al. 2013; Coupland 1961).

Both changes in ongoing reclamation guidelines and the length of time pipelines have had to recover are expected to have divergent effects on plant communities (Desserud et al. 2010; Desserud and Naeth 2013; Ostermann 2001), underlying soils (Jong and Button 1973; Naeth et al. 1987; Soon et al. 2000), and the associated seed bank (Petherbridge 2000). Previous studies have shown that areas disturbed by pipelines and other natural resource extraction activities exhibit improved vegetation recovery with longer intervals after disturbance, both in fescue grasslands of the Foothills Fescue (Desserud et al. 2010) and Aspen Parkland (Desserud and Naeth 2013), as well in the Mixedgrass Prairie (Rowland 2008; Wali 1999). However, recovery of rough fescue (*Festuca hallii* and *F. campestris*) was sensitive to construction methods and reclamation efforts (Desserud et al. 2010). In SE Alberta’s Dry

Mixedgrass prairie aridity can make recovery of late-seral communities easier, particularly where cool-season invasive grasses are less competitive (Dormaar and Smoliak 1985). Emergence of exotic grasses like sheep fescue (*Festuca ovina*), Canada bluegrass (*Poa compressa*), and Kentucky bluegrass (*Poa pratensis*) can persist for decades in disturbed and revegetated communities (Desserud et al. 2010). Moreover, greater alteration of soil properties and impeded vegetation recovery have both been demonstrated with larger pipelines due to the increased size of the disturbance area (Desserud and Naeth 2014; Naeth et al. 1987), both within the soil and aboveground during soil handling and pipeline installation. Finally, while several studies (listed above) have typically examined soil and vegetation recovery after industrial disturbance in native grassland ecosystems of Alberta, these studies have generally examined direct impacts of traffic and soil disturbance rather than adjacent (off-site) impacts. Henderson and Naeth (2005) documented the spread of crested wheatgrass from seeded field margins (i.e. access right-of-ways) into surrounding native grasslands in SE Alberta but did not assess secondary impacts of this encroachment on native vegetation (community or seed bank).

Compared to aboveground vegetation, there is little research delineating seed bank composition and formation across Western Canadian grasslands (Ambrose and Wilson 2003; Clements et al. 2007; Harker et al. 2000; Johnston et al. 1969; Otfinowski et al. 2008; White et al. 2012; Willms and Quinton 1995). Moreover, there is limited research assessing seed bank responses following severe soil disturbance and restoration (Helsen et al. 2015), as most seed bank studies target the effects of livestock grazing (Kinucan and Smeins 1992; Sanderson et al. 2007; Wilms and Quinton 1995) or a handful of other management factors such as fertilization (Williams 1984), plowing (Sanderson et al. 2007; Levassor et al. 1990), or fire (Gonzales and Ghermandi 2008; Ren and Bai 2016), often in combination with grazing. Petherbridge (2000) briefly examined seed bank densities of native and introduced species (but did not examine community composition) along the Express Pipeline in SE Alberta's Dry Mixedgrass prairie, reporting soils from stripped pipelines had greater seed densities and the majority of emergence was from introduced species. While germination of seeds from the seed bank contributed to greater ground cover, the emergence of preferred native perennials was limited (Petherbridge 2000). Seed banks provide a significant ecological service, restoring native plant diversity but also contributing introduced and invasive plant propagules.This issue is often acknowledged in reclamation/restoration literature, but the contributions of the seed bank are often speculative (Ostermann 2001).

Seed banks play a critical role in the restoration ecology of disturbed systems (Bakker et al. 1996; Bekker et al. 1997) and the recovery of grasslands (Ambrose and Wilson 2003; Willems and Bik 1998). Recovery potential is reduced when seed banks abundant and rich with native vegetation fail to form (Laughlin 2003). Disturbance events can impact seed bank composition and the density of plant species indicative of previous communities or states (Bakker et al. 1996), and composition evolves over time as plant communities recover (Wagner et al. 2006); as a result, disturbed soil seed banks hold a record of disturbance legacies (Clements et al. 2007). Seed banks are often dissimilar from the aboveground plant community (Hopfensperger 2007), and this dissimilarity is exacerbated under recent (Hopfensperger 2007; Renne and Tracy 2007), intense (Ma et al. 2010), or chronic ongoing (Martinez-Garza et al. 2011) disturbance events due to the accumulation of seed from ruderal species. Desirable species such as late-seral native grasses, which tend to form transient seed banks (Kinucan and Smeins 1992), are sensitive to extirpation in disturbed environments due to both a limited competitive ability and a reduction in source seed. In addition, presence of a species in the seed bank does not always translate into its expression within the aboveground plant community; species can be expressed (i.e. germinate, emerge and grow) when the seed bank is 'activated' by disturbance events (Bakker et al. 1996), or emerge stochastically given the occurrence of an ideal suite of environmental conditions. While germination alone does not always translate into survival within established grasslands with high competition, this process may be more likely in disturbed microenvironments with limited competitive stress as competition of established vegetation imposes stress on emergent seedlings (Booth and Swanton 2002).

Along with direct disturbance effects, invasion by exotic species can reduce the species richness and density of native plants in the seed bank (Gioria et al. 2014), and propagule pressure from introduced species is a mechanism facilitating further invasion (DiVittorio et al. 2007; Eschtruth and Battles 2009; Warren et al. 2012). Once native seed banks have been eliminated or modified, the successional trajectory of a plant community is likely to be further altered by introduced species, as shown by previous studies on aboveground vegetation in disturbed grasslands (Desserud and Naeth 2014; Elsinger 2009; Ostermann 2001). Managing seed banks within disturbed grasslands has conservation implications as deleterious annuals like cheatgrass (*Bromus tectorum*) can subsequently dominate the seed bank (Johnston 2011) and require intensive management to control (Meyer et al. 2007). Similarly, the abundance of problematic species like *A. cristatum* (a bunch grass) and sweet clovers (*Melilotus* spp., a biennial legume) along southern Alberta pipelines are solely dependant on seed banks for their reproduction and spread.

Biological soil crusts (BSCs) are comprised of cryptogamic species (i.e. lichens, mosses, spike-mosses, algae, and cyanobacteria) that are functionally important in arid and semi-arid grasslands for stabilizing the soil surface (Belnap 2003). Lichens can fix nitrogen and increase available phosphorus, retain soil moisture and regulate both seed bank formation and seedling recruitment (Johansen 1993; Li et al. 2005). However, BSCs are sensitive to disturbance and recover slowly (BLM 2001), thus protecting this overlooked community layer should be a management priority (Belnap 2003), especially during reclamation (Bowker 2007). Macrolichens with large or branching thalli (i.e. fruticose and foliose lichen species in locally abundant genera like *Cladonia* and *Xanthoparmelia*) are particularly fragile to mechanical disturbance (i.e. crushing) and are slow to recover (Belnap and Eldridge 2001; BLM 2001; Cole 1990).

While the relationship between soil crust and seed banks has been acknowledged in the ecological literature, particularly in xeric ecosystems (Bertiller and Ares 2011; Clements et al. 2007; Hawkes 2004; Li et al. 2005), there is limited research exploring these relationships in Dry Mixedgrass Prairie, especially under divergent management and disturbance regimes [e.g., between 1996 and 2006 in the *Restoration Ecology* journal, only 1.6% of studies presented data on BSCs (Bowker 2007)]. Within deserts, bare ground and disturbed crust contain greater soil seed densities than intact soil crusts (Li et al.

2005), and therefore we hypothesize that seed densities and crust cover along disturbed pipelines and in the adjacent work area will reflect similar reductions. Although biological soil crusts have been demonstrated to reduce overall seed bank density (Li et al. 2005), in prairie environments we hypothesize crusts could be beneficial for preventing the entry and persistence of seeds from introduced annuals and biennials into the soil. In contrast, native plants that co-exist with native cryptogams may have adaptations to assist their incorporation into the seed bank despite the presence of crusts [e.g. trypanocarpy[1] of *Hesperostipa* spp. (Boeken et al. 2004)]. For example, native plant germination and survival may be better adapted to occur in areas with prairie crusts than introduced species, and once native seedlings establish this can lead to a plant community composition with higher native cover and biomass (Belnap et al. 2001b; Belnap 2003). Establishment of invasive species can also be deleterious to biological crusts, reducing cover and richness of lichens and mosses (Belnap et al. 2006). Rough lichen crusts and crusts disturbed by livestock tend to have greater seed entrapment (Li et al. 2005). Thus far, the role of biological crusts in reclamation and vegetation restoration after industrial disturbance have received limited attention, even though surface disturbance and the absence of crusts can result in organic matter loss, soil erosion, and decreased soil microbial activity (Belnap 1995). Cover of crusts were acknowledged in research by Elsinger (2009), Low (2016) and Nannt (2014), all of which generalized the impacts of pipelines and well-sites as a single grouping, with Elsinger (2009) noting the lack of recovery for *Selaginella densa* (a major component of Dry Mixedgrass prairie crusts) on even old disturbances.

To quantify the legacy effects of pipeline age and diameter on current vegetation, seed bank, soil crusts and soil properties, we assessed key biophysical factors on and adjacent to 18 pipelines located within the University of Alberta's Mattheis Research Ranch in SE Alberta, Canada. Pipelines included in the survey were typically natural gas gathering or transport lines ranging in diameter and date of installation, situated on loamy and sandy loam soils within the Dry Mixedgrass natural subregion.

[1] Needle grasses have an awn that twists as the caryopsis dries. On the soil surface the awn untwists when wet, thereby boring the seed into soil. Other native grasses with similar behavior include *Avenula hookeri* and *Danthonia* spp.

Specific objectives were to 1) evaluate the impact of pipeline age and size on the aboveground vegetation, soil seed bank and BSC, and 2) quantify relationships among the aboveground plant community, seed bank and BSC at the plant-soil interface, particularly across increasing distances from pipelines. We initially intended to evaluate the effects of past reclamation efforts and seed mixtures but were unable to attain reliable records for the sites examined. Additionally, we set out to determine if pipelines served as vectors for the invasion of introduced vegetation into adjacent native grassland, and their effects on native grassland richness and diversity. Ultimately this information will help understand the long-term impact of pipelines on the ecological sustainability of native grassland ecosystems.

6.3 Methods

6.3.1 Study Area and Site Selection

Pipelines were sampled at the University of Alberta Mattheis Research Ranch situated in the Dry Mixedgrass Prairie natural subregion and contained range sites of both loamy and sandy-loam soils. Eight sites were sampled in the fall of 2013, and another 10 sites in spring 2014 (Fig. 6.1). The Mattheis Ranch is 5,000 ha in size, most of which is non-cultivated grassland (>90%), and has numerous energy developments, including more than 150 natural gas wells connected by an extensive network of pipelines. The ranch is custom grazed annually by about 725 cow/calf pairs. Long-term annual precipitation for the area is 330 mm (Fig. 6.2), with soils varying from Humic Regosols (sandy soils) to Brown and Dark Brown Chernozems (on loams). Areas with sandier soils were typically on gently rolling sand dunes formed by aeolian deposits following deglaciation in SE Alberta, which have since been stabilized by vegetation such as sandgrass (*Calamovilfa longifolia*). Other dominant vegetation included blue grama grass (*Bouteloua gracilis*), needle-and-thread grass (*Hesperostipa comata*), wheatgrasses (largely *Pascopyrum smithii*), and June grass (*Koeleria macrantha*) on loamier ecosites (Adams et al. 2013).

Pipelines were initially selected by visually inspecting the landscape for linear disturbances leading away from well sites or marked intersecting roadways. This was assisted by maps of well-sites and their associated pipeline network. Only upland sites within the matrix of native grassland surrounding the pipeline were selected (i.e. tame pastures and embedded wetlands were excluded). The location of pipelines ranged from moderately obvious to relatively non-descript (i.e. evident only with careful inspection). The pipelines sampled were distributed across the ranch following their relocation, with sampling taking place at a stratified random location therein in order to ensure access (i.e. facilitate sampling). Resulting sites represented various states of revegetation, including pipelines with notable disturbance, but not obviously seeded, to those likely revegetated (based on observed vegetation) by taller native species such as green needlegrass (*Nassella viridula*) and slender wheatgrass (*Elymus trachycaulus* sbsp. *Trachycaulus*), as well as agronomic invasive species like crested wheatgrass or sweet clover in addition to weeds.

6.3.2 Plant Community and Seed Bank Sampling

In theory, establishment of invasive plants and saturation of the seed bank will be a function of distance from disturbance and time passed since pipeline installation. We sampled the seed bank along 55 m long linear transects (n=15 intervals in 2013, and n=16 intervals in 2014) placed perpendicular to the edge of the pipeline trench every 5m (see Fig. 6.3), stretching along at least 70 m of pipeline. Sampling resolution was higher near the pipeline, replicating methods employed by Hansen and Clevenger (2005), where corridors were sampled 0, 5, 10, 25, 50, 100, and 150 m from the edge. I sampled at the following intervals: center of the pipeline trench (hereafter ‘center’), edge of the pipeline trench (hereafter ‘edge’), 0.5 m from the edge, and again 1, 2, 3, 5, 7.5, 10, 15, 20, 25, 35, 45, and 55 m away. Areas from 0.5 to 20 m were considered part of the ‘work area’ of the pipeline construction area, while those beyond 20 m were considered to be non-disturbed by pipeline construction. At each distance, soil cores were then bulked to assess germinable seed bank within each distance category. Although research shows that plants can migrate beyond 55m, using a scale of 150 m to 200 m from the pipeline could lead to confounding factors in the grassland environment at the Mattheis Ranch (e.g. dune blow-outs, ephemeral wetlands, other disturbances, etc.).

Plant community composition was assessed along a subset of six perpendicular transects within each of the 18 pipelines and at all distances except the pipeline edge, by estimating foliar cover of individual plant species within a single 50 cm x 50 cm quadrat. Ground cover components of mineral soil, rock, litter and total basal vegetation (area covered by grass crowns tillers) were also recorded at all sampling distances in 2013 and 2014, with soil crust components identified to broader taxonomic groups (i.e. lichen or moss). In 2015, a detailed description of biological soil crust species was recorded during soil and biomass sampling using a subset of sampling distances (pipeline center, 1 m, 5 m, 20 m, and 55 m) on all 18 pipelines. In total, we removed 225 soil cores and estimated foliar and ground cover at 84 points per pipeline. For biomass sampling, plants were clipped to ground level in the quadrat, then separated and bagged by individual species. Additionally, litter was harvested after recording the foliar cover and ground cover in each quadrat. Dry weights of each vegetation component were recorded after drying in a 55ºC oven for at least 2 days. Finally, while weighing individually bagged forb and grass species, inflorescences (Poaceae, Cyperaceae, Asteraceae, Fabaceae, etc.) or flowers (Boraginaceae, Campanulaceae, and other forbs) were counted to indirectly assess sexual reproductive effort, although these data are not presented here.

6.3.3 Soil Properties

Mineral soil cores were sampled (3.2 cm x15 cm deep, LFH removed) at each of the pipeline center, 1 m, 5 m, 20 m, and 55 m from the pipeline edge, with four cores taken at each distance. Soil samples were bulked within each distance, and then used to assess pH, electrical conductivity (EC), soil organic matter (OM), total nitrogen (N), and total carbon (C). Total OM was quantified by combusting 10 g of soil in a furnace at 450ºC for 4 hr and measuring subsequent mass loss. EC and pH were measured in a soil solution that was one-part soil and two-parts water. Soils were shaken for at least 30 minutes before measuring pH, and the soil solution filtered before measuring EC. Total carbon and nitrogen were measured using a LECO TruSpec CN elemental analyzer (LECOCorporation, St. Joseph, MI, USA). with samples that were ground to a powder with a ball mill to ~ 0.1 mm (Retsch MM400 Mixer Mill, Retsch, Haan, Germany). Soil texture was measured using the hydrometer method (Bouyoucos 1927) for soils sampled from the pipeline trench and 55 m from the disturbance. For texturing, 40 g of soil and 4 g of sodium hexametaphosphate (Calgon) were suspended in 1 L sedimentation tubes, and the proportion sand, silt and clay subsequently quantified. Additional single soil cores (3.2 cm x15 cm deep) were taken at each distance interval, later dried and weighed to measure bulk density, after which roots were removed, dried and weighed.

6.3.4 Characterizing the Germinable Seed Bank

Before preparing soil samples for germination, they were placed in cold storage (below 0ºC) to prevent germination. After thawing, roots, rocks and rhizomes were removed to eliminate the vegetative bud bank and coarse debris. Pots 25.4 cm were lined with sterilized sand to a depth of 2 cm to provide additional rooting depth. Soil samples were then spread on top of the sand layer to a maximum depth of 2.5 cm. Additional pots (n = 4) containing only sterilized sand were set aside to ensure the sand was weed free (which it was). Water was added as required to prevent desiccation and promote seedling emergence. Species composition of the seed bank was identified by allowing seeds in the topsoil to germinate in a greenhouse. All samples started and ended their germination period at the same time and were grown under similar conditions (16:8 hr day and night; 20ºC). Soil was stirred every three months to encourage further germination after plant emergence slowed. Seedlings were counted as they germinated and removed after identification for a total period of 12 months. Unidentifiable seedlings were grown out in pots until mature enough for identification. Plants derived from vegetative buds (bud bank) were uncommon and removed from the data set. Species noted to have emerged from the bud bank included Kentucky bluegrass (*Poa pratensis*), perennial sowthistle (*Sonchus arvensis*), and dandelion (*Taraxacum officinale*).

6.3.5 Pipeline Characteristics and Reclamation

Pipelines assessed in this investigation and their connecting wells were identified using the Alberta Energy Regulator (AER) "One Stop: Reclamation Certificate Mapping Tool", which provided publicly available licensing information and asset descriptions (i.e. permit and licensing dates, pipeline diameter, licensee, etc.) (AER 2016; in Appendix Table D.1). Supplementary data that was unavailable from the AER was also acquired from AbaData™ Oil and Gas Map Software. Records of permitted encumbrances were also examined for 'right-of-way' registrations; these were the best records of installation for disturbances installed by older companies (i.e. Alberta Gas Trunk Line Co. Ltd.[2]) that had undergone numerous corporate changes. Many pipelines selected for the study represented gathering lines with small diameter widths (60.3 – 88.9 mm), as well as larger transport lines (168 – 1067 mm), installed between 1960 and 2007, primarily for the transport of sweet and sour natural gas. Apart from the larger pipelines, disturbed trenches were often ~ 1 to 2 m in width, with an additional 20 m right-of-way on either side for equipment associated with construction. Information on well site installation was more detailed, and often used to infer information on pipeline installation. Record keeping and quality was particularly poor for older wells and difficult to link to pipelines included in the study. Thus, consultation with a reclamation specialist (Brian Lambert, AEP Reclamation and Remediation Policy Specialist, personal communication) was used to identify probable methods of pipeline installation. Limited information also existed on the seed mixes used during reclamation, leaving it unclear on whether the introduced forages present along pipelines had been initially seeded. This included forage grasses like crested wheatgrass (*Agropyron cristatum*), intermediate wheatgrass and native reclamation grasses like green needle grass (*Nassella viridula*) and slender wheatgrass (*Elymus trachycaulus* ssp. *Trachycaulus*).

6.4 Statistical Analysis

[2] Alberta Gas Trunk Line Co. Ltd. had undergone numerous corporate name changes and was eventually linked to NOVA Gas Transmission Ltd. owned by TransCanada which had inconsistent licence records through AER and AbaData™.

Plant community, seed bank, and soil crust composition from all 18 pipelines were analyzed using a combination of multivariate methods and regression, both assessed with R software (R Core Team 2017). Using the *adonis* function in the *vegan* package (Oksanen et al. 2017), permutational multivariate analysis of variance (perMANOVA) was used to test for significant differences in communities (each of aboveground vegetation cover and biomass, belowground seed bank, or biological soil crust at the soil surface, but run separately) relative to the fixed effects of progressively greater sampling distances from the pipeline trench (trench center, trench edge, 0.5, 1, 2, 3, 5, 10, 15, 20, 25, 35, 45, and 55 m, assessed as a continuous variable for perMANOVA), varied pipeline installation dates (used as a continuous variable), and pipeline diameters (as a continuous variable). Soil properties and ground cover variables were also tested using perMANOVA to identify soil characteristics influential on the composition of communities. Pairwise comparisons were conducted with perMANOVA for each community, where each distance within each pipeline was regarded as its own step-wise treatment 'community'.

An indicator species analysis (ISA), using the *multipatt* function in the *indicspecies* package of R (De Caceres and Legendre 2009), was used to identify significant species responses to increasing distances from pipeline centers, as well as pipeline age and size classes. ISA required discrete class variables; thus, pipeline age was categorized in 10 year intervals (0-60 years) since initial disturbance, and three pipeline diameters were tested (60.3 mm, 88.9 mm, and >168.3 mm). Patterns in community data from across sites were evaluated using non-metric multidimensional scaling (NMDS) ordination with the *metaMDS* function in *vegan* using a Bray-Curtis distance metric, and 999 permutations (Oksanen et al. 2017). Solutions were limited to 2 dimensions to maintain interpretive quality of the results. Soil properties (texture, EC, pH, and OM%, C%, N%, and C:N ratio), plant production metrics (dry standing biomass and litter biomass), and community indices (richness, Shannon's diversity, Pielou's evenness, and Sorensen's similarity) were included as vectors in associated NMDS biplots when identified as significant ($P < 0.05$) using the *envfit* function in *vegan* (Oksanen et al. 2017). Individual species included in ordination graphs were limited to conservative significance levels (often $P < 0.001$) to reduce clutter in resulting ordinations and minimize the risk of type 1 errors.

Multivariate analyses of biological soil crust communities required the use of a dummy variable (x =1) (necessary for perMANOVA and NMDS) in the community matrix because there were numerous plots (largely on the pipeline disturbed areas, particularly the trench) that lacked soil crust. Relationships among ground cover components (i.e. biological soil crust composition and the proportion of bare ground and litter) were also evaluated using NMDS joint biplots; sample locations that lacked soil crusts typically included abundant bare soil and litter, thus the dummy variable was excluded.

Compiled count data on seed densities for each plant trait grouping (graminoids, introduced and native perennials, introduced and native biennials, and native and introduced annuals) were zero-inflated, and were further analyzed using generalized linear models (GLMs) in R using linear logistic regression set to the Poisson distribution. This process related seed density responses of each vegetational grouping to either the distance from pipeline disturbance, pipeline age, or pipeline diameter. Simple linear regression was possible for variables such as seed bank richness and diversity. Relative seed densities for the top 15 species in the soil seed bank were compared between the "undisturbed" prairie (25 m to 55 m) and the pipeline trench (center) using generalized linear mixed models (GLMMs) in R using the *lme4* package (Bates et al. 2015), similarly set to the Poisson distribution and using each pipeline site as a random (blocking) factor.

Soil properties (texture, EC, pH, and OM%, C%, N%, and C:N ratio) were further analysed to determine their relationship to pipeline age, diameter, and date of installation in an additive linear mixed effects model using the *lme4* package, with individual pipeline sampling site as a random (blocking) factor.

Further, non-linear models were created to show the area (distance) away from trenches impacted, with non-linear equations and coefficients generated reported in supporting tables. A logistic growth curve was also fitted to biological soil crust cover to demonstrate the area of impact. The model for logistic growth was as follows:

$$y = \frac{\Theta_1}{1 + \exp[-(\Theta_2 + \Theta_3 x)]} + \varepsilon$$

where y is the response (i.e. soil crust cover) and x is the predictor (i.e. distance from pipeline). Values of Θ were determined through a self-starting function for logistic non-linear models (*Sslogis*) predicting the asymptote (Θ_1), x-mid (Θ_2), and scaling values (Θ_3).

Values for biological crust cover were zero inflated, thus boxplots, medians, and their comparison using a Bonferroni corrected Kruskal-Wallis test (*agricolae* package in R (De Mendiburu 2017)) were also included with the non-linear function.

6.5 Results

We observed 123 different vascular plant species during our survey of plant community composition aboveground, with 120 angiosperms (reproducing by seed). From the germinable soil seed bank 96 species emerged, sharing 72 angiosperm species in common. Several species emerged from the seed bank that were not present in the aboveground community that could be characterized as ruderals like stinkweed (*Thlaspi arvense*), spiny-leaved sowthistle (*Sonchus asper*), and tumble mustard (*Sisymbrium altissimum*), or hydrophytic species such as broadleaf cattail (*Typha latifolia*). The seed bank contained 27 introduced species and one noxious weed - perennial sow thistle (*Sonchus arvensis*). Canada thistle (*Cirsium arvense*) was found in the aboveground community but not the seed bank.

The plant community aboveground was dominated by native perennial grasses (ranked in descending order as follows: *Calamovilfa longifolia*, *Bouteloua gracilis*, *Hesperostipa comata*, *Koeleria macrantha*, and *Pascopyrum smithii*, respectively) in the undisturbed area (25 m to 55 m from pipeline) while some native grasses were displaced from trenches (pipeline center) and instead had a relatively greater presence of introduced species (ranked *Calamovilfa longifolia*, *Poa pratensis*, *Hesperostipa comata*, *Melilotus officinalis*, and *M. alba* respectively) (Table 6.1). Compared to the dominant species in the plant community, the germinable seed bank from the undisturbed grassland contained greater seed densities of forbs, while native perennial grass emergence was low [ranked fringed sage (*Artemisia frigida*), pygmy flower (*Androsace septentrionalis*), foxtail barley (*Hordeum jubatum*), low sedge (*Carex duriuscula*), and curly dock (*Rumex crispus*)]. Seed densities of ruderal and introduced species were similar, being highest at the pipeline center (ranked *Artemisia frigida*, *Hordeum jubatum*, *Melilotus alba*, *Melilotus officinalis*, and *Rumex crispus*). There were marked differences in the composition of Dry Mixedgrass aboveground vegetation and seed banks, which are described in detail below.

6.5.1. Aboveground Plant Composition

Aboveground vegetation was influenced by all aspects of pipeline disturbance (distance from pipeline, disturbance age, pipeline diameter) and all two-way and three-way interactions ($P \leq 0.011$), with the interaction of pipeline diameter and age having the strongest influence on plant communities ($R^2 = 0.05$; Table 6.2). Plant communities on the pipeline trench differed from all sampling sites off the trench, including those in work areas and non-disturbed grassland ($R^2 = 0.10$; $P < 0.01$; Table 6.3). Vegetation on and near the trench (pipeline center, 0.5 m, 1 m, and 2 m) also differed compared to areas 55 m from the pipeline edge ($P \leq 0.016$; Table 6.3). In general, plant communities in non-disturbed grassland differed from those on the pipeline ROW (Table 6.3).

Plant community composition also varied in relation to measured edaphic factors, with a gradient in total soil C strongly associated with community shifts ($R^2 = 0.16$; $P = 0.001$; Table 6.4). Gradients in soil salinity ($R^2 = 0.09$) and pH ($R^2 = 0.05$) were also associated with variation in plant community composition ($P = 0.001$; Table 6.4). Vegetation was also associated with the interaction of total soil carbon and nitrogen, and soil carbon and salinity ($P < 0.036$; Table 6.4). Additionally, litter cover ($R^2 = 0.10$) and biomass ($R^2 = 0.07$) were associated with shifts in plant community, followed by soil exposure

($R^2 = 0.04$) and the proportion of the soil surface occupied by stems (or crowns) ($R^2 = 0.03$) ($P < 0.004$; Table 6.5). Total soil crust cover was not associated with shifts in plant communities ($P = 0.179$; Table 6.5). These shifts along gradients are reflected in Fig. 6.4, described in detail below.

Distinct differences in plant community composition along pipelines were reflected in the indicator species analysis (Table 6.6). Species with greater cover on pipeline trenches creeping into the adjacent native grassland included quackgrass (*Elytrigia repens*) (center – 1 m; P = 0.003), green needlegrass (*Nassella viridula*) (center – 2 m; P = 0.007), white sweet clover (*Melilotus alba*) (center - 3 m; P = 0.001), yellow sweet clover (*Melilotus officinalis*) (center – 5 m; P = 0.001), and *Agropyron cristatum* (center – 10 m; P = 0.082), most of which were introduced species. Introduced goat's beard (*Tragopogon dubius*) was present along pipeline trenches and dispersed up to 45 m away (P = 0.026). The native species slender wheatgrass (*Elymus trachycaulus* ssp. *trachycaulus*) was greater in cover adjacent to the pipeline trench (0.5 m) and established up to 3 m away (P = 0.024). Native species negatively affected by pipeline installation were grasses like *Bouteloua gracilis* (0.5 m – 55 m; P = 0.001) and Sandberg's bluegrass (*Poa secunda*) (1 m to 55 m; P = 0.001), as well as prairie club-moss (*Selaginella densa*) (2 m – 55 m; P = 0.003). Native forbs generally did not exhibit significant responses at P = 0.05, though 2 species exhibited weak increases at intermediate distances ($P < 0.10$). Nuttall's evening primrose (*Oenethera nuttallii*) was present at 5 m, 7.5, 10 m, 15 m, 20, 25 m, and 45 m (P = 0.057), while the native parasite clustered broomrape (*Orobanche fasiculata*) was present at 5 m, 15 m, and 25 m (note that neither of these species responded positively to the pipeline trench - center to 3 m). Finally, a coarser ISA contrasting the pipeline trench, adjacent right-of-way, and non-disturbed grassland also revealed other indicators. Along trenches (center and edge), the introduced species cicer milkvetch (*Astragalus cicer*), smooth brome (*Bromus inermis* ssp. *inermis*), and Russian wild rye (*Elymus junceus*), together with several native ruderal grasses and forbs like Canada fleabane (*Conyza canadensis*), Flodman's thistle (*Cirsium flodmanii*), tumble grass (*Schedonnardus paniculatus*), and wild licorice (*Glycyrrhiza lepidota*), were more abundant (P < 0.05). In contrast, many native species were associated with areas sampled off the pipeline work area (Table 6.6).

A number of plant species indicators were associated with pipeline age and diameter. The narrowest pipelines (60.3 mm) were associated with numerous native grasses and forbs like *Artemisia frigida*, *Bouteloua gracilis*, *Koeleria macrantha*, moss phlox (*Phlox hoodii*), *Carex duriuscula*, prairie spike-moss (*Selaginella densa*), Sandberg's bluegrass (*Poa secunda*), and scarlet butterfly weed (*Gaura coccinea*) (P ≤ 0.015; Table 6.7). Native slender wheatgrass and green needlegrass were also associated with narrower pipelines (P ≤ 0.026; Table 6.7). While no introduced species were associated with small diameter pipelines, moderate diameter lines (88.9 mm) were associated with more ruderal and introduced species, such as Canada thistle (*Cirsium arvense*), dandelion (*Taraxacum officinale*), fowl bluegrass (*Poa palustris*), *Hordeum jubatum*, and perennial sowthistle (*Sonchus arvensis*), with the appearance of select native perennial indicator species like lance-leaf scurf-pea (*Psoralea lanceolata*), prairie sage (*Artemisia ludoviciana*), and sun loving sedge (*Carex pensylvanica*) (P < 0.05; Table 6.7). Large diameter pipelines (168.3 mm or greater) were associated with introduced grasses like *Agropyron cristatum* and smooth brome (*Bromus inermis* ssp. *inermis*) (P = 0.001; Table 6.7). Older pipelines were associated with introduced grasses like *Poa pratensis* or *Agropyron cristatum*, as well as introduced legumes (P < 0.012; Table 6.8). More recent disturbances included introduced Asteraceae species like *Tragopogon dubius* and *Taraxacum officinale* (P = 0.001; Table 6.8).

When the interactions of distance + age, and distance + diameter, were examined, some notable plant community indicators emerged (Table 6.9). *Melilotus* was associated with pipeline trenches and occurred at all pipeline diameters (P < 0.05), and there was a marginal tendency for *Glycyrrhiza lepidota*, a native legume, to behave in a similar way (P = 0.056). In contrast, *Agropyron cristatum* was associated with trenches and right-of-ways on larger pipelines (>168.3 mm) (P = 0.001). The narrowest diameter pipelines had *Poa pratensis* present only on trenches, but on wider pipelines (88.9 mm and >168.3 mm) it was an indicator across all sampling distances.

Plant community NMDS ordination (stress = 0.22, dimensions = 2, distance = Bray-Curtis) showed strong divergence in vegetation among loamy and sandy loam soil (largely along axis 1), as well as in relation to the abundance of introduced plant species (largely axis 1; Fig. 6.4). Introduced species such as quack grass (*Elytrigia repens*), *Poa pratensis*, *Melilotus* spp., and *Agropyron cristatum*, together with disturbance tolerant native ruderals, were all clustered together and were negatively associated with the vector for distance from pipeline. Invaded pipelines dominated by *Poa pratensis* and *Melilotus* were associated with higher total biomass resulting from introduced vegetation. Native biomass also tended to be higher near pipeline disturbances, but total foliar cover of native species was associated with greater distances from the pipeline. In contrast, many native species were associated with areas either far from pipelines or at intermediate distances, as evidenced by the proximity of biplot vectors for native cover and increasing distance from pipeline. Also evident within the ordination was that loamier ecosites were generally associated with greater soil C, N, OM, silt composition, and greater cover of biological soil crust. One sample location was unique from all other sites due to greater soil salinity (apparent in Fig. D.2). Sandier soils generally exhibited greater sensitivity to pipeline disturbance than loamy soil, with sampling distances near the pipeline having greater introduced cover in sandy soil. On sandy loams greater introduced cover was associated with higher litter loads, greater biomass from introduced species, and higher plant species richness and diversity. Pipeline disturbance was also associated with *Elymus trachycaulus* spp. *trachycaulus*, *Nassella viridula*, and *Taraxacum officinale* on loamier soils. Wider pipelines were associated with greater introduced cover along the pipeline trench; however, these soils had greater bulk density and C:N ratio. Biplot vectors for native cover and distance from pipeline were associated with each other, indicating native cover was reduced near the immediate pipeline disturbance. Inclusion of the Sorenson's similarity index indicated that more similar seed bank richness was associated with loamier soils and greater native plant cover.

Univariate tests (Table 6.10 and 6.11) revealed that native cover was reduced directly along pipelines and this interacted with pipeline diameter ($P = 0.0004$), where diameters $\geq$ 168.3 mm had the largest decline in native cover to 49.8 ± 10.6 % while native cover in ROW and non-disturbed areas was 72.1% and 92.4%, respectively. Narrower pipelines had less significant reductions in native cover although a trend was apparent. This pattern was similarly reflected in introduced cover that was higher along pipeline trenches and interacted with pipeline diameter ($P = 0.01$) resulting in the greatest introduced cover along trenches (36.6 ± 11.8 %) for pipelines ≥ 168.3 mm. Trenches had higher species richness ($P < 0.001$), Shannon's diversity ($P = 0.021$), evenness ($P = 0.032$), and lower beta diversity, whereas Shannon's diversity interacted with diameter ($P = 0.044$) to produce higher community diversity along trenches and the right-of-way when trenches were ≥ 168.3mm.

Ordination of the individual biomass of plant species responded similarly to plant community composition measured via foliar cover (stress = 0.22, dimensions = 2, distance = Bray-Curtis) (Fig. 6.5). Areas sampled near trenches were associated with greater introduced biomass, which was comprised of species like *Agropyron cristatum*, and was further associated with native species like *Cirsium flodmanii, Hordeum jubatum,* and *Symphoricarpos occidentalis*. Wide diameter pipelines were associated with a greater biomass of relatively phreatophytic species like *Carex duriuscula, Carex pensylvanica,* and *Juncus balticus*. Unique trends included a strong association between litter cover and total native biomass. Univariate tests on community biomass (Table 6.12 and 6.13) revealed that total introduced plant biomass ($P = 0.004$), introduced forb biomass ($P = 0.0001$), and total biomass ($P = 0.002$) were highest specifically along trenches at 518.9 ± 69.4 kg/ha, 267.5 ± 46.7 kg/ha, and 1709.0 ± 126.7 kg/ha, respectively. Native biomass was not affected by pipeline disturbance ($P \geq 0.05$). Overall community productivity also differed in relation to soil texture, where sandy loams were more productive (1469.3 ± 87.3 kg/ha) than loamy ecosites (880.5 ± 163.3 kg/ha), with a similar trend also reflected in the mass of fallen litter on the soil surface ($P = 0.002$).

6.5.2. Seed Bank Composition

Germinable seed bank composition was affected by all aspects of pipeline disturbance ($P = 0.001$), including all two-way interactions ($P \leq 0.015$; Table 6.2). Soil properties also influenced the seed bank, with soil salinity ($R^2 = 0.09$; $P = 0.001$), total carbon ($R^2 = 0.05$; $P = 0.001$), and pH ($R^2 = 0.03$; $P = 0.001$) associated with relatively strong shifts in composition (Table 6.4). Aspects of ground cover also regulated seed bank composition, with litter cover ($R^2 = 0.03$; $P = 0.003$) and fallen litter biomass ($R^2 = 0.03$; $P = 0.001$) having the strongest effect. Total biological soil crust cover ($R^2 = 0.02$; $P = 0.020$) and ground exposure ($R^2 = 0.03$; $P = 0.018$) were also associated with seed bank composition gradients (Table 6.5). The area of ground occupied by plant stems did not influence seed bank composition, but it did interact with biological soil crust and bare ground ($P < 0.04$; Table 6.5).

Seed bank composition varied with sampling distance in a manner unlike the aboveground vegetation, with the largest differences between the trench and areas 15 to 35 m away (Table 6.3). However, undisturbed areas 55 m away did not differ from any areas outside the trench (Table 6.3). Seed bank composition nevertheless differed between all 3 generalized areas, including the trench, right-of-way and undisturbed grassland ($P \leq 0.044$; Table 6.3).

Indicator species within the seed bank among pipelines resembled trends observed in the aboveground plant community (Table 6.14). Seeds associated with pipeline trenches were from *Elymus trachycaulus* sbsp. *trachycaulus* dispersing up to 2 m ($P = 0.010$), while *Melilotus alba* dispersed up to 2 m ($P = 0.010$) and *Melilotus officinalis* up to 3 m ($P = 0.010$). *Nassella viridula* had higher densities along pipeline edges and nearby (0.5 m) and exhibited the potential to disperse up to 10 m ($P = 0.049$). Tumble grass was associated with areas relatively close to the pipeline trench (0.5 m to 2 m; $P = 0.040$). Non-disturbed areas (25 m to 55 m) were associated with seed from narrow-leaf hawksbeard (*Crepis tectorum*) and sand dune wallflower (*Erysimum capitatum*) ($P < 0.027$).

The smallest diameter pipelines had seed banks rich in native perennial grasses like *Bouteloua gracilis*, *Koeleria macrantha* and Sandberg's bluegrass (*Poa secunda*), along with weedy native forbs like pepper grass (*Lepidium densiflorum*) and reflexed rockcress (*Arabis holboellii* ssp. *retrofracta*) ($P < 0.034$; Table 6.15). More ruderal and introduced species were associated with intermediate sized pipelines (88.9 mm), like *Rumex crispus*, rough cinquefoil (*Potentilla norvegica*) and rough false pennyroyal (*Hedeoma hispida*) ($P < 0.01$; Table 6.15). In contrast, large pipelines (≥168.3 mm) had seed banks greater in *Artemisia frigida*, *Calamovilfa longifolia*, sand dropseed (*Sporobolus cryptandrus*), and white sweet clover (*Melilotus alba*) ($P < 0.01$; Table 6.15). Similar to the aboveground herbaceous plant community, older pipelines were associated with introduced perennials and biennial sweet clovers in the seed bank, while recent disturbances had seed banks containing introduced Asteraceae species ($P < 0.05$; Table 6.16).

There were also a number of important indicators emerging when the interactions between distance × decade and distance × diameter were assessed (Table 6.17). *Koeleria macrantha* was found in the seed bank at all sampling distances for the smallest diameter pipelines (60.3 mm), and in moderate sized pipelines (88.9 mm). However, *Koeleria macrantha* was less abundant within the seed bank of trenches and was not an indicator for large pipelines ($P = 0.011$). The native annual *Androsace septentrionalis* was present in the seed bank of pipelines of all diameters and sampling distances, except the trenches of wider diameter pipelines (>168.3 mm) ($P = 0.038$). Similar to aboveground vegetation, *Melilotus officinalis* was associated with pipelines of all diameters, while white sweet clover was an indicator for wider disturbances ($P < 0.003$).

Like aboveground vegetation, NMDS ordination of seed bank composition (stress = 0.28, dimensions = 2, distance = Bray-Curtis) exhibited divergence in seed bank among loamy and sandy loam soils, particularly along axis 2 (Fig. 6.6). Less distinct groupings of seed bank attributes were evident along axis 1. Axis 1 was positively associated with total seed density, richness, and diversity, and included both introduced species (specifically introduced perennial forbs) and native grasses. In contrast, native perennial forbs were negatively associated with axis 1, and most other seed bank components, as well as accompanying biomass pools, were in-between (Fig. 6.6); densities of introduced grasses, native biennial forbs, and introduced biennial forbs corresponded with total introduced seed densities. Pipeline age and diameter were associated with greater total biomass and introduced grass seeds in the seed bank, while distance was unrelated to either ordination axis ($P = 0.122$; Table D.6).

Seed banks from loam soils were associated with greater soil fertility (C, N, and OM) and silt, and were associated with higher similarity in species richness relative to the aboveground plant community (Fig. 6.6), as well as greater biological soil crust cover. Species present in the germinable seed bank of loamy soils included *Bouteloua gracilis*, *Koeleria macrantha*, narrow-leaf hawksbeard (*Crepis tectorum*), and Pennsylvania cinquefoil (*Potentilla pensylvanica*) ($P < 0.001$). Similar to plant communities, seed banks within sandier soils were more sensitive to pipeline disturbance, being associated with greater introduced seed densities (relatively larger symbols in Fig. 6.6); sandy areas were also associated with greater densities of native perennial forbs like *Artemisia frigida* and plains wormwood (*Artemisia campestris*) ($P < 0.001$). Seed densities of native and introduced biennials were associated with greater soil pH and litter biomass. Saline soils corresponded with greater densities of introduced seed, which contributed to higher total seed densities and species richness. Species found in the seed bank of saline soils were typically halophytic native species like fowl bluegrass (*Poa palustris*), *Hordeum jubatum*, saline saltbush (*Atriplex subspicata*), or introduced annuals like black medic (*Medicago lupulina*), dwarf snapdragon (*Chaenorhinum minus*), and perennials like *Rumex crispus* ($P < 0.001$).

Ordination of seed bank composition from soil collected only from pipeline trenches (pipeline center and edge) (stress = 0.24, dimensions = 2, distance = Bray-Curtis) showed divergence in seed bank composition driven by the density of introduced seed in the topsoil (Fig. D5). Seed banks on pipelines lacking high densities of introduced seed included more native forbs and grasses like *Bouteloua gracilis*, *Artemisia frigida*, and *Carex duriuscula* ($P < 0.05$). Seed bank densities of native and introduced vegetation were generally not associated with pipeline characteristics, edaphic factors, or ground cover ($P > 0.05$).

Seed densities of plant growth forms were influenced by distance from pipeline (Fig. 6.7), with introduced biennials (primarily *Melilotus* spp.) distinctly increasing within 10 m of the trench ($P < 0.001$). Introduced perennial forb and grass seeds were also more abundant closer to pipelines, while introduced annual forbs were more abundant further from pipeline trenches ($P < 0.001$). Densities of native forbs, grasses, and graminoids were all reduced with closer proximity to pipelines ($P < 0.001$; Fig. 6.7). Additionally, the narrowest pipeline diameter class of 60.3 mm was associated with the highest seed densities of native biennial forbs (species like *Erysimum* spp. or *Arabis/Boechera* spp.) (Table 6.18). High densities of introduced perennial forbs were associated with pipeline diameters of 88.9 mm (Table 6.18). Pipeline age also influenced observed seed densities (Fig. 6.8). Recent pipelines were associated with higher densities of introduced annual forbs, introduced grasses, native biennial forbs, and grass-like species ($P < 0.001$). Older pipelines were associated with higher densities of introduced perennial forbs, native perennial forbs, and native grasses (Fig. 6.8).

Seed density of native graminoids was influenced by the interaction of distance from pipeline and soil texture ($P = 0.015$), resulting in lower seed densities along trenches when soils were loamy (155.3 ± 377.8 seeds/m^2) compared to non-disturbed loamy grassland (430.6 ± 370.6) (Table 6.19 and 6.20). Density of introduced forbs in the seed bank was affected by the interaction between pipeline diameter and distance from the disturbance ($P = 0.041$), where introduced seed densities differed most along wider lines and peaked in both the trench and non-disturbed grassland, while densities were lowest in the ROW. Native forb seed densities were affected by soil texture ($P = 0.009$), peaking in sandier soils (Table 6.19 and 6.20). Characteristics of seed bank diversity (richness, diversity, evenness, and similarity to the plant community) exhibited limited significant responses to fixed factors (Table 6.21). Beta diversity differed among sampling distances ($P = 0.038$), peaking in non-disturbed grassland and being lowest along the right-of-way (Table 6.20 and 6.21). Linear regressions of seed bank diversity with distance revealed several significant relationships (Fig. 6.9). Seed bank species richness ($R^2 = 0.018$, $P = 0.029$), Shannon's diversity ($R^2 = 0.028$, $P = 0.006$), and Pielou's evenness ($R^2 = 0.021$, $P = 0.017$) were significantly higher near pipeline disturbance and declined with increasing distance into the non-disturbed prairie (Fig. 6.9). Similarity in species richness between the plant community and seed bank did not differ with distance ($P = 0.134$) (Fig. 6.9).

The relationship between seed banks and soil properties was explored further in a correlation matrix (Fig. D.6.) Soil salinity had a strong association with total seed density ($r = 0.62$), introduced perennial forbs ($r = 0.87$) and native grasses ($r = 0.82$). Introduced biennial forb seed density (consisting primarily of *Melilotus*) was positively correlated with higher soil pH ($r = 0.43$). Later seral grasses generally declined with greater litter cover ($r = -0.37$) and litter biomass ($r = -0.36$) and tended to be more abundant in loamier soils (indicated by silt, $r = 0.43$) with higher soil fertility (OM, C, N, $r = 0.41$, 0.38, and 0.34 respectively). Conversely, native perennial forbs were more abundant in soil with poor fertility and sandier textures, and native biennial forbs followed a similar but weaker pattern.

6.5.3. Biological Soil Crust Composition

Soil crusts were comprised primarily of prairie club-moss (*Selaginella densa*), pebbled pixie-cup (*Cladonia pyxidata*), and vagabond rockfrog (*Xanthoparmelia camtschadalis*) (Table D.8). Evidence of soil crust organisms was found at 49.2% of observation points, with one study site entirely lacking cryptogams due to salinity. Crusts had the potential to cover up to 88.9% of the soil surface; however, only 23.6% of observation points had crust cover exceeding 5%. Cover of cryptogamic species was significantly reduced by industrial activities (i.e. trenching, traffic) closer to the pipeline trench (i.e. on the right-of-way) and extended up to 20 m from the trench itself (Fig. 6.10), which translated into effects on biological crust community composition. Despite the reduction in crust cover near the trench, chrono-sequences stratified by sampling distance found no significant improvement in cover over time ($P \geq 0.364$) (Table D.9).

All aspects of pipeline disturbance observed (distance from pipeline, pipeline age and diameter) had a strong influence on soil crust composition ($P = 0.001$; Table 6.22), with markedly different soil crust assemblages occurring along a gradient from the pipeline trench to the non-disturbed grassland ($P < 0.05$; Table 6.23). The diameter of pipelines interacted with pipeline age ($P = 0.001$) and distance ($P = 0.004$), but there was no relationship between age and distance ($P = 0.998$; Table 6.22). These effects were reflected in the indicator species analysis of biological crust composition. The cyanobacteria commune known as 'nostoc' was the only organism associated with close proximity to pipelines, occurring 1 m from the edge of the trench ($P = 0.004$). No cryptogamic species detected were associated directly with pipeline trenches (Table 6.24). *Cladonia pyxidata* ($P = 0.006$) and *Selaginella densa* ($P = 0.048$), and to a lesser extent upstanding shadow lichen (*Phaeophyscia constipata*) ($P = 0.095$), occupied areas 1 m to 55 m from the pipeline edge. Rosette pixie-cup (*Cladonia pocillum*) ($P = 0.008$), marginally frosted lichen (*Physconia muscigena*) ($P = 0.104$) and star moss (*Tortula ruralis*) ($P = 0.077$), were all associated with areas further from pipeline disturbance (≥20 m). Cow pie lichen (*Diploschistes muscorum*) ($P = 0.074$) and split-leg soldiers (*Cladonia cariosa*) ($P = 0.093$) exhibited some sensitivity to pipeline construction, occurring at least 5 m from the trench (Table 6.24).

When the interaction of distance and diameter were considered, *Phaeophyscia constipata* was associated with the non-disturbed area on narrower pipelines (60.3 and 88.9 mm) and the right-of-way on pipelines 88.9 mm ($P = 0.041$), indicating sensitivity to disturbances associated with wider diameter pipelines. Vagrant lichen *Xanthoparmelia camtshadalis* was associated with narrower diameter pipelines (60.3 to 88.9 mm) ($P = 0.082$), bristly haircap moss (*Polytrichum piliferum*) was associated with wider pipelines (88.9 to >168.3 mm) ($P = 0.013$), and crustose bracted sulphur lichen (*Fulgensia bracteata*) was associated only with the larger diameter pipelines (>168.3 mm) ($P = 0.020$; Table 6.24). There was a significant interaction between disturbance age and distance for *Cladonia pocillum*, which was an indicator for the non-disturbed area adjacent to pipelines installed within the last 10 years.

Ground cover variables (bare ground, litter, stems) had a strong influence on soil crust composition ($P = 0.001$), with litter cover explaining most of the variation ($R^2 = 0.29$). Fallen litter mass did not explain variation in soil crust communities ($P = 0.40$; Table 6.26). Soil crusts were also associated with soil properties, with total soil carbon explaining the most variation ($R^2 = 0.13$; P = 0.001), while soil salinity and pH were also significant (P ≤ 0.028; Table 6.25). There was also a significant two-way interaction between soil carbon and pH (P = 0.003; Table 6.25). These gradients were apparent in the NMDS ordinations produced (discussed below).

The NMDS ordination of soil crust composition (stress = 0.13, dimensions = 2, distance = Bray-Curtis) showed a reduction of biological soil crusts on pipeline trenches and an associated shift in diversity and composition (Fig. 6.11). Later seral crust communities with greater cover and representation of larger, fragile fruticose and foliose lichens like spiny shield lichen (*Cetraria aculeata*), split-peg soldiers (*Cladonia cariosa*), and Wyoming rock-shield (*Xanthoparmelia wyomingica*), were all associated with areas beyond the right-of-way (20 m) in non-disturbed grassland, and corresponded with soil characteristics found in loamier soils such as greater soil fertility (OM, C, N). Soil crust richness and diversity were associated with more moderate crust cover and conditions that supported some moss and crustose or squamulose lichen species like *Candelaria vitellina*, cow pie lichen (*Diploschistes muscorum*), elegant disc lichen (*Buellia elegans*)*, Placidium squamulosum*, soil paint lichen (*Acarospora schleicheri*), or other (unknown) moss (P < 0.05; Table D.11). Soil crust communities were disassociated from the vectors for litter cover, litter biomass, total plant community biomass, C:N ratio, soil bulk density, pH, salinity (EC), and pipeline width (Fig. 6.11, Panel B). Soil exposure was associated with closer proximity to pipelines but also occurred on sandier ecosites. Bare soil was associated with mosses like bristly haircap (*Polytrichum piliferum*), dry calcareous Bryum moss (*Bryum caespiticum*) or star moss (*Tortula ruralis*), and lichens like *Fulgensia bracteata*, hammered shield lichen (*Parmelia sulcata*), *Phaeophyscia constipata*, and wand lichen (*Cladonia rei*), in addition to non-lichenized cyanobacteria—nostoc (P < 0.05).

Biplots of seed bank characteristics and species were fit to the same ordination (Fig. 6.11, Panel C) to identify the relationship between soil crusts and seed bank composition. Notably, greater similarity between the seed bank and aboveground vegetation was associated with areas of higher soil crust cover and later seral crust communities. Areas with enhanced crust cover supported a germinable seed bank comprised of native species like *Bouteloua gracilis*, *Koeleria macrantha*, Pennsylvania cinquefoil (*Potentila pensylvanica*), and the native ruderal common pepper grass (*Lepidium densiflorum*). The lone introduced species favored in the seed bank by crust was redroot pigweed (*Amaranthus retroflexus*) ($P < 0.05$). Seed banks with greater similarity and an accumulation of native species had lower total seed density than the seed banks containing more introduced species where soil crust cover and richness were reduced. Tickle hair grass (*Agrostis scabra*) occurred in the seed bank when soil crust cover was more intermediate in abundance and hosted greater species richness and diversity ($P < 0.05$). Seed banks richer in graminoids and native annual forbs like *Androsace septentrionalis* occurred closer to pipeline disturbance or where there was greater soil exposure. Seeds from introduced biennial forbs and perennial forbs like *Melilotus alba* and *Sonchus arvensis* were associated with higher litter loads and edaphic factors more unfavorable for soil crusts (higher pH and salinity).

Similar to Fig. 6.11, the ordination in Fig. 6.12 shows shifts in soil crust composition when ground cover variables were included like bare ground and litter cover. Many of the trends in biological crust community shifts were similar, except that the shifts in ground cover on pipeline trenches became more apparent. In Fig. 6.14 the divergent roles of litter cover and bare ground were more apparent, where the vector for bare soil corelated with the cover of moss species and nostoc, while litter cover was not associated with cryptogamic species except for *Selaginella densa* and *Cladonia pyxidata*, which may be more tolerant of litter cover. A greater number of significant species emerged from the seed bank (biplots in Fig. 6.12, Panel C). Introduced *Crepis tectorum* seed density was associated with greater crust cover ($P < 0.05$). Where there was greater soil exposure (i.e., on trenches), graminoids like *Carex duriuscula*, introduced biennial forbs like sweet clovers, and *Elymus trachycaulus* ssp. *trachycaulus* were common.

Seed densities of native and introduced plants were influenced by ground cover including biological soil crusts, bare ground, and litter cover (Fig. 6.13). Higher biological crust cover was associated with lower amounts of both native and introduced seed ($P < 0.001$); however native seed densities were relatively higher than introduced seed densities overall. Greater bare soil was associated with reductions in native seed densities, while introduced seed densities increased (P < 0.001), with a threshold at ~40% soil exposure where introduced seed densities surpassed native seed densities. Both native and introduced seed densities increased with litter cover, however native seed exhibited a greater positive response to litter (P < 0.001).

The relationship between seed banks and soil crusts was explored further in a correlation matrix (Fig. D.6.) where only significant relationships (P < 0.05) where included. Later seral grasses in the seed bank were positively associated with total biological soil crust cover (r = 0.51) contributed primarily by *Cladonia pyxidata* (r = 0.49), *Selaginella densa* (r = 0.48), and *Xanthoparmelia camtschadalis* (r = 0.38); early successional crustose lichens like *Fulgensia bracteata* (r = -0.07) and *Thelenella* spp. (r = -0.07) were negatively correlated with later seral grass seed density. Native annual forb density was positively correlated with *Phaeophyscia consipata* (r = 0.47). Total introduced seed and total introduced perennial forbs consistently exhibited negative correlations between seed density and cryptogams (r = -0.01 to -0.14). Introduced grass seed density was positively correlated with two crustose lichens, *Diploschistes muscorum* (r = 0.18) and *Ochrolechia upsaliensis* (r = 0.19). Graminoid seed banks (*Carex, Juncus, Typha*) were typically negatively correlated with lichens and *Selaginella densa,* and positively correlated with total moss cover (r = 0.11).

6.5.4 Soil Properties

Pipeline disturbance modified soil properties on the trench. The predominant soil texture was sandy loam (n=13), while the remaining sites were loamy (n=4). Loamy soils had about two-fold more total C, N, and organic matter (P < 0.001), indicating greater soil fertility, while sandy loams had near three-fold greater root density (P < 0.001) (Tables 6.27, 6.28, 6.29). Additive linear mixed effects models (Table 6.21) showed that total N, C:N ratios, electrical conductivity (EC), and pH changed with distance from the trench. Trenches had lower total soil N and C, and resulted in a greater C:N ratio, while higher

EC and pH levels were associated with the trench. When distance was analyzed as a fixed factor, significantly higher EC and pH ($P < 0.001$) was found primarily within 1 m of the trench (Table 6.28 and 6.29). Soil OM was also found to change with distance ($P = 0.010$), however the highest soil OM was associated with 20 m distance (3.0 %) while the lowest OM was associated with 55 m at 2.5 % (Table 6.28 and 6.29). Observed C:N ratios were also sensitive to pipeline diameter, where wider diameters correlated with a higher C:N ratio (Table 6.21). Time since disturbance was not linked to changes in soil properties.

6.6 Discussion

Pipeline disturbance had a strong influence on plant communities, seed banks, and soil crusts. Previous research on oil and gas disturbance in Alberta's grasslands has had greater focus on plant communities (Desserud et al. 2010; Desserud and Naeth 2013) and soil (Naeth et al. 1987), while seed bank (Petherbridge 2000) and biological crusts have received little to no attention. This research revealed divergent responses between seed banks and plant communities and highlights the strong influence of industrial disturbance legacies on a variety of ecosystem properties, including soil conditions, aboveground community composition and biomass, as well as cryptogamic communities on the soil surface.

6.6.1 Legacy Effects of Pipelines on Soil

Disturbances associated with pipeline construction were associated with legacy effects on soil properties. Installation of pipelines requires trenching and soil handling, which often results in the perturbation of soil horizons (Hammermeister et al. 2003; Naeth et al. 1987), and in some cases introduces new materials like sand and gravel (older installations). Lower soil horizons and underlying parent material contain higher concentrations of salt and carbonates (Soil Classification Working Group 1998) and are the likely source of higher salinity concentrations and pH levels found within the top 15 cm of soil sampled from pipeline trenches (Jong and Button 1973; Soon et al. 2000). Similarly, Jong and

Button (1973) found higher soil pH and salt concentrations along pipeline trenches in SE Saskatchewan, and this effect was more profound on more recent disturbances; similar results were found by Naeth et al. (1987) as well as Soon et al. (2000). Soil properties did not differ with disturbance age, indicating either slow recovery or an inability to recover. Sample areas associated with pipeline trenches also had an impact on soil fertility, with higher C:N ratios and lower soil organic matter, suggesting there was either a net release of soil carbon and nitrogen, or more likely that part of the topsoil was lost due to admixing of surface and underlying soil horizons during pipeline installation (Hammermeister 2003). Naeth et al. (1987) also found significant declines in soil organic matter along pipeline trenches constructed in solonetzic Dry Mixedgrass prairie, and estimated the time required to recover at least half of the lost organic matter at 50 years. For annually cultivated farmland, soil organic matter could potentially be restored in a shorter period (Shi et al. 2014; Soon et al. 2000).

We did not measure attributes of soil structure, but it is suspected that altering pore spaces and potential aeration (Jong and Button 1973) along recently disturbed trenches affected the competitiveness and survival of certain plant species as the community recovered. Naeth et al. (1987) found that solonetzic prairie soils were susceptible to compaction, Culley (1982) found that pipeline construction tended to cause greater compaction on medium to fine textured soils, and similar results have been reported in other studies (Olsen and Doherty 2012; Ostermann 2001; Soon et al. 2000). Differences in soil properties created during reclamation can affect the success of colonizing species, including introduced annual weeds (Desserud and Hugenholtz 2015), and in turn lead to greater competitive success by the latter during the initial revegetation phases of recovery, particularly those involving the seed bank. Soil changes from pipeline installation also tended to be confined to areas relatively close to the trench, including in comparison to the overlying plant communities, similar to the findings of Xiao et al. (2014).

Presence of pipelines can also alter the soil microenvironment in other ways. Buried pipelines can alter soil temperature according to Naeth et al. (1993), raising soil temperature in the winter and spring along trenches, with adjacent lateral effects as well. This could physiologically influence plants (i.e., roots) and soil organisms directly, and could be linked to long-term differences in communities along trenches. After trenching in pipelines, the soil surface can also be altered. On sandier sites, we often noted slumping of soil directly over the trench. We suspect soil could have settled over time, or in cases where vegetated dunes were transected by the pipeline, erosion could have removed soil prior to revegetation. These micro-depressions could potentially hold moisture during heavy rainfall events, thereby favoring some plant species over others. We also noted sites where the soil was raised along the trench, presumably due to unequal soil replacement, and which could result in greater run-off. We did not measure these differences in microsites and link them to communities but suggest they could explain some of the community dynamics. Other observations included the presence of cattle trails parallel to pipeline trenches, which could result in higher trampling and forage utilization near pipelines. Legacy marks of vehicular traffic were occasionally found parallel to pipelines or within the work area of sandier sites, which likely impacted soil properties (i.e., via compaction, water infiltration), and vehicular traffic could have increased the presence of introduced species (Wilson 1988).

Significant effects found here of pipeline construction on soil properties were typically consistent with past research (Jong and Button 1973; Naeth et al. 1987; Soon et al. 2000), however they served as a useful tool when interpreting the responses of plant, seed bank, and soil crust communities. Further associations between communities, disturbance, and soil properties can be attained from relationships in ordinations and perMANOVA tests and are discussed in each of the applicable sections below.

6.6.2 Plant Communities

Shifts in plant communities along pipelines were shaped primarily by soil texture and ecosite (Adams et al. 2013; Lane et al. 1998), where sandier soils exhibited greater sensitivity to pipeline disturbance as indicated by a greater cover of introduced *Melilotus* spp., *Elytrigia repens,* and *Poa pratensis*. Soil texture also influenced community productivity and fallen litter biomass accumulation, where sandier ecosites were more productive. Soil texture can influence plant community succession and recovery post disturbance. In arid grasslands, siltier soils (an indicator for our loamy ecosites) can lead to greater recovery of native perennial grasses like *Bouteloua gracilis* (Coffin and Lauenroth 1994). In general, coarse-textured soils are more prone to erosion following surface disturbance, which negatively impacts their recovery (Li et al. 2004). In the current study, a single solonetzic ecosite was sampled that exhibited significant sensitivity to pipeline disturbance as indicated by increased halophytic and perennial weeds like *Cirsium arvense, Hordeum jubatum, Rumex crispus,* and *Sonchus arvensis*. Soil salinity was also higher along pipeline trenches, which likely exacerbated the presence of undesirable halophytic ruderals along revegetated pipelines. Due to the high representation of introduced perennial plants, noxious weeds, and unpalatable vegetation, the pipeline disturbance on solonetzic soil has likely resulted in lower rangeland health (Desserud et al. 2010; Nasen et al. 2011), ecological function and forage value.

Pipeline characteristics (distance from disturbance, diameter, and construction date) and all interactions among them significantly influenced shifts in plant communities. Sampling distances further from the pipeline were associated with greater cover of late seral grasses like *Calamovilfa longifolia*, suggesting long-term legacy reductions in the cover of later seral grass species following industrial disturbance. The latter has been reported in fescue prairies (Desserud et al. 2010) and Mixedgrass prairies (Ostermann 2001) in Alberta. While some later seral grasses like *Hesperostipa comata* were found to be apparently resilient to pipeline disturbance in the current study, it remains unclear whether the recovery resulted from revegetation efforts (albeit unlikely given the age of pipelines) or natural recovery through seed rain.

Plant communities established along trenches were distinct from all other sampling distances, including nearby adjacent vegetation on the ROW (i.e., 50 cm, 1 m, or 2 m from the trench's edge). Numerous species established along the trench demonstrated invasibility into adjacent native vegetation. *Agropyron cristatum* exhibited high invasibility, significantly encroaching into the non-disturbed (i.e. native) vegetation up to 10 m from the pipeline edge. Sites containing *A. cristatum* were likely seeded during reclamation to revegetate the disturbed area (Marlette and Anderson 1986). Pipelines more likely to be revegetated with *A. cristatum* also tended to be older and wider in diameter, likely indicating use of introduced forages to revegetate disturbed areas, presumably in an attempt to revegetate/stabilize exposed bare soil and soils prone to erosion (like the vegetated dunes at Mattheis) (Gramineae Services Ltd. 2013; Willms et al. 2005)). Once *A. cristatum* establishes and invades native grassland it can become dominant and persist in the community (Henderson and Naeth 2005). Invasion is aided by characteristics such as drought tolerance, grazing resistance and a tall stature, which enables it to intercept relatively more light compared to shorter and slow growing native grass species, as well as producing abundant biomass (Christian and Wilson 1999; Vaness and Wilson 2007). *Festuca ovina* has been identified as a problematic species among revegetated pipelines in fescue grasslands (Desserud et al. 2010) and was present along the Dry Mixedgrass pipelines we sampled; however, its presence was relatively limited and it was not associated with strong shifts in plant communities. Ostermann (2001) reported that trenches were associated with rhizomatous species which could explain the increases in introduced *Elytrigia repens* and even the establishment of native grasses like *Calamovilfa longifolia* along trenches.

Pipeline diameter was strongly associated with plant community shifts for at least one site where soil bulk density and the C:N ratio were distinctly elevated, indicating wider diameter pipeline installation was more likely to be associated with greater soil compaction and lower soil fertility. Increased soil removal and subsequent replacement with large pipelines may be more likely to result in greater topsoil admixing, as well as increased heavy equipment exposure over a longer time period, both of which would directly enhance compaction and bulk densities.

Pipelines dominated by the invasive plants *Poa pratensis* and *Melilotus*, were associated with both higher total biomass and introduced plant biomass, which likely resulted in greater litter cover and biomass. Invasive species have been shown to increase surface litter and alter litter quality, ultimately changing soil nitrogen dynamics (Evans et al. 2001). Paradoxically, native plant biomass also tended to be greater near pipeline disturbances, but the total foliar cover of native species was positively associated with greater distances from the disturbance. The simultaneous decline in native richness on pipelines suggests the high biomass is associated with a small number of native species that presumably thrive under disturbed conditions, similar to that of introduced species. This is likely attributed to taller grasses like *Calamovilfa longifolia*, *Nassella viridula*, and *Elymus trachycaulus* spp. *trachycaulus* or a few productive nativeplants like *Symphoricarpos occidentalis* or *Cirsium flodmannii* which had higher biomass along pipeline disturbance (Fig. 6.5).

Along pipelines we found increases in shrub cover and biomass from *Rosa* and *Symphoricarpos occidentalis*. Hickman (2010) similarly found increases in silver sage (*Artemisia cana*) along pipelines. Mechanisms proposed for shrub increases along pipelines include soil admixing, the break-up of hard soil layers like the Bnt horizon in solonetzic sites, fragmentation of rhizomes for clonal species (Luo and Zhao 2015), or improved soil moisture (Hickman 2010). During field surveys, we did note the presence of small depressions that could have aided in water collection, which in turn, may have promoted shrubs. Our NMDS of plant biomass also showed increased representation of relatively hydrophytic vegetation like *Equisetum laevigatum, Juncus balticus, Carex duriuscula,* and *C. pensylvanica*. Also of note is that the vegetated dune ecosites assessed in the region may have had access to a relatively shallow water table, and if so, it is possible that the microenvironment created along pipelines in these areas improved access to water. Finally, short-term exposure of bare soil immediately following pipeline installation could have allowed shrubs the opportunity to encroach under reduced competition from herbaceous vegetation, as shrub encroachment in healthy arid grasslands is typically limited (Lyseng et al. 2018).

6.6.3 Seed Banks

Seed bank composition was separated along two main edaphic gradients, soil texture and salinity. Overall, sandier soils generally contained higher seed densities than loams, this is likely attributed to differences in the plant communities established on different ecosites and the potential influence of soil texture on seed bank formation. Incorporation of seeds into the soil seed bank can be influenced by seed traits and their interaction with the soil surface (including barriers to seed entry like litter or biological crusts (Li et al. 2005; Facelli and Pickett 1991)), seed burial is faster [deeper burial in a standard unit of time] in coarse textured soils when compared fine textured soil (Benvenuti 2007). When you consider sandier soils were associated with vegetated dunes (which had less developed crusts than loams), where soil exposure was higher there was a higher probability of seed rain becoming incorporated into the seed bank. Faster rates of seed burial could have also reduced losses due to granivory and enabled captured relatively larger seeds which typically incorporate slowly (Thompson et al. 1993). Soil texture also interacted with distance from pipeline disturbance. On loamier ecosites, native graminoids were significantly higher in the seed bank of non-disturbed prairie compared to trenches. *Bouteloua gracilis* was a native perennial grass associated with the seed bank of loamier soils, while sandier soils contained native forbs like *Artemisia campestris,* but also contained greater densities of introduced grasses like *Poa pratensis*. Salinity was associated with the emergence of more halophytic, ruderal species, which in turn, resulted in greater seed bank richness and diversity. Of note, a similar association between seed bank richness and salinity emerged in Chapter 5 where Parkland-Boreal pastures were examined.

The distinct impacts of pipelines on vegetation patterns observed aboveground (with distance from pipeline for example) were not reflected as distinctly in the seed bank. Instead, seed bank composition along the pipeline trench did not differ from adjacent sampling distances until 15 m away from the trench edge (and up to 35 m). Moreover, sampling distances at 45 m and 55 m were only marginally different than the pipeline trench. This could indicate a few things, including 1) greater legacy impacts of pipeline construction on seed banks than established vegetation (Xiao et al. 2014), 2) the seed banks of non-disturbed grasslands are inherently weedy and could have an accumulation of propagules from ongoing disturbances across the landscape that were not accounted for (e.g., grazing and the influence of distant cultivated agricultural lands), or 3) certain plant species have dispersal mechanisms that can readily travel distances of 40 to 55 m, thereby increasing their similarity in the seed bank. Notably, we did see weedy indicators from the Asteraceae family emerge at further distances, from genera adapted to wind dispersal. During our surveys we did not sample beyond 55 m because the heterogeneous (i.e. dunes, wetlands, saline flats) and fragmented (e.g. access roads, other disturbances) landscape was likely to confound ecosite changes with pipeline legacy effects. Other studies have observed off-site effects adjacent to linear disturbances up to hundreds of meters away (Hansen and Clavenger 2005; Xiao et al. 2014). In hindsight, our understanding of non-disturbed Dry Mixedgrass prairie seed banks could have been better described had we sampled additional non-disturbed control sites that were further away (>100 m) from disturbances, roads, and other confounding features like dune blowouts or wetlands in representative ecosites (loamy and sandy loam).

Of note is that pipeline diameter exacerbated disturbance effects on germinable seed banks. Introduced seed densities were highest on sites with wide diameter pipelines (particularly along the trench), and surprisingly, extended into the non-disturbed prairie at these locations. Wider diameter pipelines were also associated with increased native annual, biennial, and perennial forbs.

Introduced species in the germinable seed bank that were indicators of pipeline disturbance included *Melilotus alba* (center to 2 m) and *M. officinalis* (center to 3 m), which are prolific seed producers (Turkington et al. 1978) capable of invading natural environments (Turkington et al. 1978; Wolf et al. 2008). Select native species were also indicators of pipeline disturbance, including rare tumble grass (*Schedonnardus paniculatus*) (S2), which is adapted to disturbed grasslands (ACIMS 2015), and *Elymus trachycaulus* ssp. *trachycaulus* and *Nassella viridula,* which are both commonly used for revegetating disturbed prairie. Indicator species for non-disturbed prairie seed banks included ruderal species like *Crepis tectorum* and *Erysimum capitatum*.

Compared to the aboveground plant community the seed bank was less diverse, likely caused by poor representation of perennial grasses and forbs. Native legumes, aside from a few rare occurrences, were absent in the seed bank, with no germination of *Dalea, Pediomellum, Thermopsis,* and *Vicia*, all of which were abundant aboveground. This result was somewhat not surprising given that legume seeds have an extended longevity due to a hard seed coat (Russi et al. 1992), and instead suggests that the existing native legume plants are relatively long-lived, with little recruitment over time. Similarly, in Chapter 5, low emergence of native legumes was observed despite their common occurrence in tame and modified grasslands, both studies show introduced legumes tend to form large persistent seed banks with lower thresholds (i.e. soil moisture, scarification, etc.) for inducing germination. Other seed bank surveys from the Northern Great Plains similarly lack reporting of germinable native legume seed banks (Johnston et al. 1969; White et al. 2012; Willms and Quinton 1995). Low diversity in the seed bank and the absence of dominant species aboveground resulted in low similarity in richness, which averaged across all sites and sampling distances was 25.2 %. Low similarity in grassland seed banks has been reported in other seed bank studies (Eriksson and Eriksson 1997; Hopfensperger 2007; Tracey and Sanderson 2000; Williams 1984). Within ordinations, similarity vectors were significantly correlated with loamier soils and well-developed soil crusts, indicating similarity was affected by a multitude of factors. Aspects of seed bank diversity were nevertheless responsive to pipeline disturbance, with higher richness, Shannon's diversity, and evenness occurring on or near pipeline trenches. Higher seed diversity was generally attributed to introduced species that accumulated in the soil adjacent to the disturbance. Higher evenness near the disturbance indicates disturbed seed banks in Dry Mixedgrass prairie have relatively similar representation of species seeds in the soil, while non-disturbed grassland is more likely to have a few dominant species with higher propagule pressure (i.e., contributed by native forbs *Artemisia frigida* and *Androsace septentrionalis,* which had the highest seed densities). Seed bank beta diversity was also lower near pipelines indicating greater site diversity was represented in trenches.

High dissimilarity between seed banks and plant communities also revealed that species emerged from the seed bank that were absent from the existing vegetation. Previously when discussing vegetation responses, we mentioned higher biomass of hydrophytic species along trenches. Within the seed bank, unique occurrences of hydrophytic and ruderal species occurred, the most peculiar of which was *Typha latifolia* occurring in non-disturbed native topsoil and along trenches in significantly higher densities. *Typha* is typically associated with prairie marshes, which were widely interspersed throughout the study area, indicating this species has small seeds with well-adapted dispersal mechanisms that ensure seed bank formation across landscapes and likely forms a persistent seed bank (Grace and Harrison 1986), but does not experience suitable conditions for germination in prairie soil. Otherwise, species that typically emerged from the soil that were uncommon aboveground tended to be native ruderals and their relative densities were influenced by aspects of pipeline disturbance (i.e. increases in *Hedeoma hispida* or *Schedonnardus paniculatus* along trenches). Species that were rare or absent aboveground but abundant in the seed bank likely formed a persistent seed bank (Kinucan and Smeins 1992).

Grasses that tend to dominate in both biomass and foliar cover are often known to occur at relatively low densities in seed banks (Kinucan and Smeins 1992; Willms and Quinton 1995), except for the introduced species *Poa pratensis* (Parkland seed bank study in Chapter 5; Sanderson et al. 2007; Tracy and Sanderson 2000; Travnicek et al. 2005). Notably, native grasses that tend to dominate Dry Mixedgrass prairie like *Bouteloua gracilis*, *Koeleria macrantha*, *Nassella viridula*, *Pascopyrum smithii*, and on Mattheis' sandier ecosites - *Calamovilfa longifolia* (Adams et al. 2013), were all relatively uncommon in the seed bank. In non-disturbed native grasslands, the seed bank had greater representation of forbs and included two graminoids (*Hordeum jubatum* and *Carex duriuscula*), both of which tended to increase with disturbance. The native grass that was typically associated with total native grass seed density in ordinations of the seed bank data was *Hordeum jubatum*, which is known to be associated with higher soil salinity, is well adapted for seed dispersal, and readily germinates (Badger and Ungar 1994). *Hordeum jubatum* seeds can occur at densities up to 479,200 seeds/m^2 and form a persistent seed bank (Badger and Ungar 1994) and germinates readily along a wide temperature gradient (5ºC to 30ºC) at a rate of ~91% (Galinato and Van der Valk 1986).

Agropyron cristatum was hypothesized to form a seed bank along linear disturbance and migrate; surprisingly however, it also exhibited limited abundance in the seed bank. In other studies, *A. cristatum* has been found to become a dominant species in the seed bank (Marlette and Anderson 1986) and has been shown to germinate in an *in-situ* seed bank study (Ambrose and Wilson 2003). *A. cristatum* did germinate in our study, but the number of occurrences were low and insufficient to draw conclusions regarding its relationship to other grassland structural layers. One study found, long-term persistence was found when *A. cristatum* stands were sprayed with herbicide over multiple years (4 to 7) and clipped to prevent flowering (3 years) resulting in consistent germination (Ambrose and Wilson; Wilson and Pärtel 2003), suggesting this species forms a persistent seed bank (Pyke 1990), in turn posing long-term management challenges. *A. cristatum* relies entirely on its seed bank to disperse and spread, this suggests *A. cristatum* is capable persisting, spreading, and maintaining its population even when it has a relatively low seed density. This relationship likely requires further examination. Low propagule pressure from decadent *A. cristatum* stands could be beneficial for resorting native cover.

Based on the observed germinable seed bank composition in the non-disturbed prairie we postulate that disturbance of native Dry Mixedgrass prairie will likely cause a release of primarily ruderal, early seral native forbs like *Artemisia frigida* and *Androsace septentrionalis* initially. Both Ren and Bai (2016) and Willms and Quniton (1995) found the seed bank of Saskatchewan and Alberta's fescue grasslands were dominated by *A. septentrionalis*, indicating this native species likely forms a large persistent seed bank in many prairie communities [including Dry Mixedgrass prairie] and likely contributes significant functions (i.e. ground cover, soil stabilization, etc.) aiding recovery from severe soil surface disturbance. Graminoids that would most likely emerge include *Hordeum jubatum*, *Carex duriuscula*, *Koeleria macrantha*, *Poa pratensis*, and *Agrostis scabra* (ranked by relative seed densities). *H. jubatum, C. duriuscula,* and *A. scabra* are early seral native grasses unlikely to remain competitive once later seral perennial grasses establish. Presence of *P. pratensis* in native grassland seed banks is concerning, as it can outcompete native grasses and displace native grassland (De Keyser et al. 2015) and disturbance (i.e. defoliation) to established native vegetation can increase its competitiveness and risk of encroachment (Bork et al. 2017).

6.6.4 Biological Soil Crusts

We found that Dry Mixedgrass prairie soil crusts were very sensitive to the legacy of pipeline surface disturbance resulting in unique cryptogam communities at all sampling distances from the trench and reduced overall cryptogam cover at least 20 m from the disturbance. Significant losses of biological crust components on surface disturbances in Alberta's grasslands have been reported by Elsinger (2009) and Hickman (2010), specifically of *Selaginella densa,* a dominant component of most crusts across the arid Canadian plains. Through ISA a handful of cryptogamic species like *Cladonia pyxidata*, *Selaginella densa*, and *Phaeophyscia constipata* occurring 1-55 m from the pipeline were identified as perhaps more resilient to pipeline disturbance, exhibiting some recovery; however, most cryptogamic species were indicative of distances ≥ 5 m away like *Diploschistes muscorum* or ≥ 20 m like *Cladonia pocillum*, suggesting they were likely less resilient and required longer recovery times than had already occurred (i.e. up to 50 years). The lone positive indicator 'species' directly on pipeline trenches was nostoc, which is a non-lichenized cyanobacteria commune adapted to disturbances like eroded slopes in grasslands (Paul et al. 1971) and has come to occupy a niche on recently disturbed soils (Belnap 1995). Nostoc likely plays a key role in improving soil nitrogen, and thereby aiding succession of other cryptogams and vegetation (Dodds et al. 1995; Paul et al. 1971; Nemergut et al 2007). Similar changes in soil crust communities (i.e., decreased lichen, increased cyanobacteria) with increasing disturbance have been reported in other studies (Belnap 1995; Evans and Belnap 1999). Increases in cyanobacteria and the loss of lichen due to disturbance are also consistently associated with increased bare ground, reduced litter, and an increase in exotic plants (Belnap 1995). Belnap (1995) also found changes in lichen species richness and composition with surface disturbance, where disturbance (in this case, concentrated trampling by people) eliminated all lichens. Although reductions in biological crusts are often a result of direct soil handling and physical removal from the trench, human and vehicular traffic near the trench and on the ROW likely caused additional reductions in the adjacent native grassland. These disturbances may also be subject to ongoing disturbance as the areas adjacent to pipeline is occasionally traveled by inspectors (on foot or vehicular).

Wider diameter pipelines were associated with further reductions in biological crust cover, likely resulting from increased soil removal and handling, greater traffic impacts during construction, and the creation of edaphic and microsite characteristics that were unfavorable for soil crust formation equivalent to that reflected in the non-disturbed grassland. Increased soil pH and salinity were observed along pipeline trenches, which can cause shifts in crust composition along a gradient (Belnap et al. 2001a). More acidic soils will favor lichens with algal photobionts while alkaline and saline soils will favour cyanobacteria (Belnap et al. 2001a). Cryptogamic species also responded to pipelines based on life strategies and growth form. Wider diameter pipelines had a greater presence of crustose lichen species like *Fulgensia bracteata*, which tend to be early successional species that colonize bare soil. Communities of moss species, crustose lichens, and squamulose lichens occurred in plots with greater bare ground adjacent to pipeline trenches. Assemblages of the latter are known to occur in earlier seral crust communities (Belnap and Eldridge 2001) and could indicate a trajectory towards recovery. Non-disturbed prairie on loamier soils exhibited later seral communities, as exhibited by an abundance of fruticose and foliose lichens (Belnap and Eldridge 2001); the structure of the thalli of these lichens makes them particularly sensitive to disturbance.

Other negative influences on biological soil crust communities included high litter cover, which was associated with pipeline trenches, and greater introduced plant species. Based on interpretation of the soil crust NMDS, *Selaginella densa* and *Cladonia pyxidata* were somewhat tolerant to litter, and *S. densa* could also be found when hand-raking litter (i.e. when collecting biomass). Coverage by thin litter layers can be beneficial for maintaining a moist microenvironment that keeps cryptogams metabolically active (Belnap et al. 2001a). However, the high litter loads produced by *Melilotus* and introduced grasses along pipelines were likely deleterious to crust communities.

Within plant communities, we found that pipelines constructed in sandier prairie ecosites had a more profound effect on composition, specifically favoring introduced plant species. Soil crusts are ecologically important for reducing soil erosion by aggregating and binding soil particles, thereby stabilizing the soil surface (Belnap 1995; Guo et al. 2008). Since sandier soils are more prone to erosion, pipelines lacking crust redevelopment may experience greater subsequent soil erosion (Li et al. 2004), in turn, lowering soil fertility and favoring early seral plant communities. Re-establishment of biological crust aids in the establishment of vegetation and recovery of dune ecosites (Guo et al. 2008), which were amply represented at the Mattheis Research Ranch study area. It should be noted that crust development tends to be greater on soils that contain more silt, like loams (Anderson et al. 1982), and is consistent with the pipelines observed at Mattheis. Shifts in biological soil crust communities were also associated with total soil carbon. This is likely a product of pipeline disturbance (i.e. on the trench) and local soil texture, where slight decreases in C were associated with pipeline trenches and loamier soils had higher soil fertility.

The loss of cryptogamic species at the soil surface could further indicate changes in soil microbial and fungal life forms in soil. Lichens have recently been described as the association of a photobiont (alga or cyanobacteria), a fungus, and most recently, a basidiomycete yeast (Spribille et al. 2016) for some lichen families. Industrial disturbances in prairies impact soil microbial communities and population sizes (Anderson et al. 2008; Viall et al. 2014). Lack of lichen formation could be indicative of changes in microsite conditions that negatively impact one of more of the organisms required for cortex formation. Importantly, this results in a loss of ecosystem function, including site stability, biodiversity, nutrient cycling, and its services affecting seed bank formation (Li et al. 2005).

6.6.5 Dynamics Between Plant Communities, Seed Bank, and Biological Soil Crusts

Ground cover dynamics (litter cover, bare soil, and biological soil crust cover) influenced seed bank composition, potentially through their ability to capture or shield the soil surface from seed rain (Li et al. 2005). A higher density of native grasses like *Bouteloua gracilis* and *Koeleria macrantha* were associated with higher similarity in species richness to the aboveground community and was likely facilitated by higher biological soil crust cover. Soil crusts are both a feature of healthy intact prairies, but also influence seed bank formation (Li et al. 2005). It is possible that characteristics of prairie soil crusts co-evolved with prairie plant communities and native grasses may have adapted characteristics to enter the seed bank through the crusts associated with them. Seeds of both *B. gracilis* [0.35 – 0.6 mg (Carren et al. 1987)] and *K. macrantha* [2.5 – 3 mm and 0.32 mg (Dixon 2000)] are relatively small, which could enable capture by the rough textured crust's surface and promote subsequent entry into the seed bank. These species are also bunch grasses that provide interstitial space for soil crusts to form.

Biological crust cover, litter cover, and bare ground influenced relative seed densities of native and introduced plants. Gelbard et al. (2003) found that exotic species richness was negatively correlated with biological crust cover, while we found reduced exotic seed density with crusts. This suggests intact biological crusts serve as a barrier to exotic plant propagules and thereby help native grassland resist exotic plant encroachment (Gelbard et al. 2003). Introduced species benefited from greater soil exposure in the current study, which was characteristic of pipeline surface disturbance on the ROW, and in particular the trenched area. Overlays of seed bank composition over soil crust composition showed that select small seeded introduced species like *Amaranthus* spp. and *Crepis tectorum* had mechanisms that aided in seed bank formation when soil crust cover was high. Overall, native seed abundance was also reduced with high biological crust cover though they remained relatively more abundant than that of introduced species. Native seed densities were also greater when litter cover was high, suggesting litter helps capture native seeds, potentially by protecting them from predation by granivores like rodents (Reed et al. 2006).

Invasive *Melilotus* species had higher seed densities concentrated along pipeline disturbance. While seed from this large statured biennial species was expected to migrate into the non-disturbed native plant community, we found that *Melilotus* cover was instead tightly correlated with dispersal and density in the seed bank. In a small field trial, we found that most new seedlings of *Melilotus* established within 1 m of the parent plant in high densities (unpublished data). Aboveground, *Melilotus* had a tendency to occur along all trenches regardless of pipeline diameter; the seed bank of *M. officinalis* was similarly associated with trenches of all diameters, while *M. albus* was associated with wider trenches (>168.3 mm). Densities of *Melilotus* seeds were relatively high in the trenches of older (41 to 50 years) pipelines. *Melilotus* also exhibited a negative relationship with soil crusts, forming a denser seed bank were greater soil exposure was recorded, which tended to become exacerbated by pipeline disturbance.

6.6.6 Comments on Reclamation and Revegetation

Overall, we found few significant relationships within the plant, germinable seed bank, and soil crust community data with pipeline age, outside of specific ISA and perMANOVA tests. To date, soil properties did not indicate recovery, and biological crusts remained significantly altered due to pipeline construction. We suspect that this indicates a strong legacy effect of oil and gas disturbance on native prairies, and/or slow recovery processes, which have been reported in other studies observing soils and vegetation (Naeth et al. 1987; Nasen et al. 2011; Viall et al. 2014). Sites revegetated with relatively abundant native cover likely had plant communities formed under minimal disturbance conditions (narrow diameter pipelines) and natural recovery.

Native grasses *Nassella viridula* and *Elymus trachycaulus* sbsp. *Trachycaulus* are often used to reclaim industrial disturbance in Dry Mixedgrass prairie, and we found these species increasing along disturbed areas. Pipelines at the Mattheis Research Ranch did not have records of reclamation; thus we cannot ascertain with confidence whether these sites were seeded. Due to the age of many disturbances and the relatively small size of gathering lines, it is possible they established with natural recovery as seed rain became naturally available on the landscape. Aboveground, *E. trachycaulus* sbsp. *trachycaulus* tended to establish adjacent to younger pipelines (0 to 10 years), where it was found up to 3 m away from the trench. However, associated seed rain resulted in higher densities of seed entering the soil along the pipeline trench and up to 2 m from the trench edge. *E. trachycaulus* sbsp. *trachycaulus* seed bank formation was influenced by pipeline disturbance and ground cover dynamics, tending to accumulate where there was greater soil exposure, and responded similarly to the conditions that favored *Carex duriuscula*, *Juncus balticus*, and *Melilotus* spp.

Past research has shown that native plant communities can recover when left unseeded, and when compared to revegetation efforts with introduced species, natural recovery can result in healthier, ecologically functional prairie in the long-term, where resulting communities can have reduced coverage of bare soil and be equally productive to those areas with introduced vegetation (Wilson 1989). Native grasses like *Bouteloua gracilis, Hesperostipa comata,* and *Koeleria macrantha* can increase over-time on non-seeded prairie disturbances like wellsites (Hammermeister et al. 2003), while seeded mixtures containing wheatgrasses (like *Elymus lanceolatus* or *Elymus trachycaulus*) can outcompete other desirable later seral grasses like *H. comata* (Hammermeister et al. 2003) or *Festuca halli* (Desserud et al. 2010), and wild genotypes of their own species (Schröder and Prasse 2013). Wheatgrass domination could result from higher N availability post disturbance that could subside as N availability declines (Hammermeister et al. 2003) thereafter allowing other native species to increase (Desserud and Naeth 2014). In a study by Willms et al. (2005), seed mixtures containing *Agropyron cristatum* or *Leymus junceus* were compared to native mixtures or monocultures of species like *Pascopyrum smithii, Bouteloua gracilis* or *Nasella viridula*, native grasses out performed introduced grasses in their ability to produce biomass and improve soil fertility (Willms et al. 2005). Hence, native cultivars are likely still valuable for their ability to restore ecological function.

Reducing bare ground during restoration is often key to the successional trajectory of that community, as bare ground warms the soil surface and facilitates the accumulation and subsequent development of introduced propagules in the soil (Wilson 1989). Selection of introduced species that quickly stabilize the soil and fix nitrogen can achieve short-term recovery of productive and palatable vegetation (Gardiner and Wiken 2003; Halvorson and Bauer 1984). Species like *Melilotus* were promoted as an early seral nitrifier, but evidence of it persisting long after initial seeding have been reported in numerous studies (Hickman 2010; Klemow and Raynal 1981; Stoa 1933; Turkington et al. 1978; Wilson

1989). These legumes were once commonly included in reclamation seed mixtures for oil and gas disturbance and along prairie roadways (Simmers and Galatowitsch 2010) primarily because of their salt resistant properties (Ghaderi-Far et al. 2010; Rogers et al. 2008) and ability to withstand waterlogging (Rogers et al. 2008). Hickman (2010) reported that *Melilotus* is still used for reclamation in Alberta, largely intended as a short-term cover crop.

Sweet clover reproduces solely through seed and has a biennial lifecycle; by the third year of plant development a lignified-skeleton of the previous year's growth remains, and a cluster of seedlings can often be observed growing in close proximity to the parent plant. The biennial life cycle of sweet clover can make this species an unpredictable supply of forage, as perennials typically offer greater stability in long-term forage supply. Individual sweet clover plants can create their own micro-environment and alter the surrounding area by nitrifying the soil, shading neighbors, and increasing relative humidity (Riper and Larson 2009). These characteristics make it a strong competitor against relatively short-statured native grasses and forbs, and in the case of arid grasslands, is also adapted to resource limited environments. This process of producing and dropping seed, and facilitating seedling spread, can effectively facilitate invasion of sweet clover. Moreover, this process can occur together with a number of other associated (i.e. passenger) plant species that quickly join the initial invader in opportunistically colonizing the new environmental conditions at the resulting micro-site (MacDougal and Turkington 2005).

6.6.7 Further Research

Overall studies observing the influence of industrial or significant anthropogenic disturbance on seed banks is limited (Petherbridge 2000). Reclaiming disturbances and establishing native vegetation can be difficult, thus in many trials managers aim to establish early or mid-seral communities that they predict will have favorable longer-term community trajectories. However, conditions of reclamation like alterations to soil (Desserud and Hugenholtz 2015) and seasonal precipitation (Boeken and Shachak

1998) can alter community recovery trajectories and result in alternative stable-states as trajectories of recovery are unpredictable (Matthews and Spyraes 2010; Suding 2011).

Restoring a seed bank abundant in native perennial vegetation in grasslands should warrant some priority in setting restoration goals. More studies monitoring seed bank recovery and formation following restoration efforts should also be conducted. Seed banks could potentially be restored by applying native hay (Desserud and Naeth 2011; Desserud and Hugenholtz 2017), transferring seed banks naturally stored in topsoil (Zhang et al. 2001), raking litter from undisturbed areas, or preserving the seed and bud bank of topsoil *in situ*. Studies like these making observations of seed bank formation over time and observing *in-situ* seedling recruitment would improve our understanding of community recovery potential, the ecology of seed bank formation, the potential competitiveness of species in the seed bank, seed persistence, and much more.

Further studies attempting to monitor or restore soil crusts in xeric environments like the northern plains are nearly non-existent. Limited research suggests restoration can occur when the topsoil is inoculated with early seral species like cyanobacteria (Bowker 2007; Wang et al. 2009). During preliminary presentation of this research, industry representatives mentioned they would sometimes attempt to rake propagules of biocrusts (i.e. moss fragments) from undisturbed areas onto the reclaimed disturbance. Recovery of soil crusts is known to be slow, thus minimization of surface disturbance remains important. Further experimental research into biological crust recovery is required, especially in arid grassland ecosystems where interactions between seedling recruitment (Delach and Kimmerer 2002), seed bank formation (Li et al. 2005), and grassland plant community assembly likely occur (Belnap et al. 2001b; Belnap 2003).

As mentioned in Chapter 5, our understanding of plant propagules stored could have been enhanced by also observing the bud bank. During greenhouse study set-up coarse roots were removed (primarily from sandgrass and western wheatgrass). For many species of grasses however, vegetative reproduction through buds is more common than that from seed (Coffin and Laurenroth 1989; Klimes 2007; Sprinkle 2010). Additional observations of *in-situ* seedling recruitment in plots at variable levels of disturbance and with different soil cover (litter vs. bare ground vs. soil crust) could further enrich our understanding of plant recruitment in a competitive established community.

Surveys of plant communities and disturbances including cryptogamic communities are relatively rare (Bowker 2007). These organisms provide a number of key ecological services (C fixation, N fixation, soil moisture retention, erosion control, etc.) that are greatly understudied, particularly in northern temperate grasslands. Not-surprisingly therefore, recovery of cryptogamic communities is often over looked when determining reclamation success, with emphasis placed on the successional trajectory of the plant community alone. We recommend surveys for reclamation certification or assessments of restoration efforts acknowledge aggregate groups or functional groupings of cryptogams in temperate grasslands. Our results identified strong legacy effects on crusts from pipelines installed decades ago, meaning cryptogamic communities recovery very slowly without aided restoration efforts.

I would also like to acknowledge that my personal knowledge of cryptogamic diversity has improved since this study began and if we were to go back we would likely see greater representation of inconspicuous crustose and squamous lichens, for example, the relatively inconspicuous *Cladonia* spp. that rarely produce podetia[3] like *C. robbinsii* and *C. dahliana.* The latter species could have been inadvertently grouped during sampling into *C. pyxidata*. We also ignored lichens that occurred on vegetation (epiphytes), litter (usually crustose), rocks, and filamentous species in soil. Including these species in a survey would greatly increase our level of understanding regarding cryptogamic community responses to disturbance.

[3] A secondary thallus common in *Cladonia* that elevates apothecia (cups baring spores). This is a secondary growth form of *Cladonia* that is fruticose, while the non-fruiting primary thalli are squamulose.

Additional ecological questions could be addressed from these data that would further enrich our understanding of the dynamics between vegetation, seed banks, and soil crusts in grasslands. Restoration activities often aim to recover visible plant diversity, overlooking hidden or ‘dark diversity’ (Pärtel et al. 2011), including that in the seed bank (Moeslund et al. 2017) where data like this offers insight. Species-specific influences on seed banks and plant communities could be examined for species like *Melilotus* spp. or *Poa pratensis*. This seed bank data could also be characterized based on seed traits (i.e. size, adaptations for seed bank entry, etc.) and these dispersal mechanisms (i.e. traits) then further linked to disturbance features, soils, and plant communities.

6.7 Conclusions and Management Implications

Seed banks contain a record of disturbance legacy that can be overlooked in surveys of the aboveground vegetation. In this study, along with the aboveground vegetation, the seed bank, underlying soil properties, and corresponding biological crust also exhibited strong legacy effects. Time since pipeline disturbance had limited apparent effects on plant communities and seed banks, notably introduced forages like *Agropyron cristatum* and *Melilotus* seed densities were often linked to older installations. More importantly, biological soil crusts had significant reductions along trenches and exhibited nearly no recover along pipeline trenches. Biological soil crusts play a major role in soil surface stability, soil fertility, seed bank composition, and ultimately plant establishment. Efforts to restore this community layer should be addressed in reclamation and restoration projects within grasslands, although it is unclear what options remain in place to do so (i.e. BSC inoculation). Wider diameter pipelines were also often associated with greater community alteration, and therefore warrant greater attention during restoration. Although we sampled a limited number of ecosites, soil texture and soil salinity were both found to interact with pipeline disturbance to impact recovery, suggesting unique restoration guidelines are needed for different ecosites.

6.8 Literature Cited

AbaData[tm] Oil and Gas Map Software. 2016. Accessed December 2016. http://www.abacusdatagraphics.com/abadata.asp?gclid=CjwKEAiArIDFBRCe_9DJi6Or0UcSJAAK1nFvOjqjBaXTWN8BJLqkQfFOijSTxvv8qZDLR9U-vUjfgxoC-Czw_wcB

Adams, B.W., Richman, J., Poulin-Klein, L., France, K., Moisey, D. and McNeil, R.L. 2013. Rangeland plant communities for the Dry Mixedgrass natural subregion of Alberta. Second Approximation. Rangeland Management Branch, Policy Division, Alberta Environment and Sustainable Resource Development, Lethbridge, Pub. No. T/040.

Alberta Agriculture and Forestry. 2016. AgroClimatic Information Service. Accessed Feb 9, 2017 for – the Verger monitoring station and historical records were accessed for T 22 – R 14 – W4. http://agriculture.alberta.ca/acis/alberta-weather-data-viewer.jsp

[ACIMS] Alberta Conservation Information Management System. 2015. 2015 ACIMS Plant Species Ranking. Accessed October 2, 2017.

[AER] Alberta Energy Regulator. 2015. Statistical Series 59: Alberta Drilling Activity Monthly Statistics. Date Modified: 2015-04-28. https://osi.alberta.ca/osicontent/Pages/OfficialStatistic.aspx?ipid=948

[AER] Alberta Energy Regulator. 2016. One Stop: Reclamation Certificate Mapping Tool. Date Accessed: 2016-10. https://extmapviewer.aer.ca/Onestop/RecCert/public/index.html

Allred, B.W., Smith, W.K., Twidwell, D., Haggerty, J.H., Running, S.W., Naugle, D.E., and Fuhlendorf, S.D. 2015. Ecosystem services lost to oil and gas in North America: Net primary production reduced in crop and rangeland. Science **348**(6233):401-402.

Ambrose, L.G. and Wilson, S.D. 2003. Emergence of the Introduced Grass *Agropyron cristatum* and the Native Grass *Bouteloua gracilis* in a Mixed-grass Prairie Restoration. Restoration Ecology **11**(1):110-115.

Anderson, D.C., Harper, K.T. and Holmgren, R.C. 1982. Factors influencing development of cryptogamic soil crusts in Utah deserts. Rangeland Ecology and Management **35**(2):180-185.

Anderson, J.D., Ingram, L.J. and Stahl, P.D. 2008. Influence of reclamation management practices on microbial biomass carbon and soil organic carbon accumulation in semiarid mined lands of Wyoming. Applied Soil Ecology **40**(2):387-397.

Badger, K.S. and Ungar, I.A. 1994. Seed bank dynamics in an inland salt marsh, with special emphasis on the halophyte *Hordeum jubatum* L. International Journal of Plant Sciences **155**(1):66-72.

Bakker, J.P., Poschlod, P., Strykstra, R.J., Bekker, R.M., Thompson, K. 1996. Seed banks and seed dispersal: important topics in restoration ecology. Acta Botanica Neerlandica **45**(4):461-490.

Bates, D., Maechler, M., Bolker, B., and Walker, S. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software **67**(1):1-48. doi:10.18637/jss.v067.i01.

Bekker, R.M., Verweij, G.L., Smith, R.E.N., Reine, R., Bakker, J.P. and Schneider, S. 1997. Soil seed banks in European grasslands: does land use affect regeneration perspectives? Journal of Applied Ecology **34**(5):1293-1310.

Belnap, J. 1995. Surface disturbances: their role in accelerating desertification. In Desertification in Developed Countries. Springer, Netherlands. pp. 39-57.

Belnap, J. 2003. The world at your feet: desert biological soil crusts. Frontiers in Ecology and the Environment **1**(4):181-189.

Belnap, J., Büdel, B. and Lange, O.L. 2001a. Biological soil crusts: characteristics and distribution. In Biological soil crusts: structure, function, and management. Springer Berlin Heidelberg. pp 3-30.

Belnap, J. and Eldridge, D. 2001. Disturbance and recovery of biological soil crusts. In Biological soil crusts: structure, function, and management. Springer Berlin Heidelberg. pp. 363-383.

Belnap, J., Phillips, S.L. and Troxler, T. 2006. Soil lichen and moss cover and species richness can be highly dynamic: the effects of invasion by the annual exotic grass *Bromus tectorum*, precipitation, and temperature on biological soil crusts in SE Utah. Applied Soil Ecology **32**(1):63-76.

Belnap, J., Prasse, R. and Harper, K.T. 2001b. Influence of biological soil crusts on soil environments and vascular plants. Biological soil crusts: structure, function, and management. Springer Berlin Heidelberg. pp. 281-300.

Bertiller, M.B. and Ares, J.O. 2011. Does sheep selectivity along grazing paths negatively affect biological crusts and soil seed banks in arid shrublands? A case study in the Patagonian Monte, Argentina. Journal of Environmental Management **92**(8):2091-2096.

[BLM] Bureau of Land Management. 2001. Biological soil crusts: Ecology and Management, Technical Reference 1730-2. Denver, CO, USA.

Boeken, B. and Shachak, M., 1998. The dynamics of abundance and incidence of annual plant species during colonization in a desert. Ecography **21**(1):63-73.

Booth, B.D., and Swanton, C.J. 2002. 50th anniversary—invited article: Assembly theory applied to weed communities. Weed Science **50**(1):2-13.

Bork, E. W., Adams, B. W., and Willms, W. D. 2002. Resilience of foothills rough fescue, Festuca campestris, rangeland to wildfire. Canadian Field-Naturalist **116**(1):51-59.

Bork, E.W., Hewins, D.B., Tannas, S. and Willms, W.D. 2017. *Festuca campestris* density and defoliation regulate abundance of the rhizomatous grass *Poa pratensis* in a fallow field. Restoration Ecology **26**(1):82-90.

Bouyoucos, G.J. 1927. The hydrometer method as a new method for the mechanical analysis of soils. Soil Science **23**(5):343-354.

Bowker, M.A. 2007. Biological soil crust rehabilitation in theory and practice: an underexploited opportunity. Restoration Ecology **15**(1):13-23.

Carren, C.J., Wilson, A.M., Cuany, R.L. and Thor, G.L. 1987. Caryopsis weight and planting depth of blue grama I. Morphology, emergence, and seedling growth. Journal of Range Management **40**(3):207-211.

Christian, J.M. and Wilson, S.D. 1999. Long-term ecosystem impacts of an introduced grass in the northern Great Plains. Ecology **80**:2397-2407.

Clements, D.R., Krannitz, P G., and Gillespie, S.M. 2007. Seed bank responses to grazing history by invasive and native plant species in a semi-desert shrub-steppe environment. Northwest Science **81**(1):37-49.

Coffin, D.P. and Lauenroth, W.K. 1994. Successional dynamics of a semiarid grassland: effects of soil texture and disturbance size. Plant Ecology **110**(1):67-82.

Cole, D.N. 1990. Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park. The Great Basin Naturalist **50**(4):321-325.

Cox, R.D. And Allen, E.B. 2008. Composition of Soil Seed Banks in Southern California Coastal Sage Scrub and Adjacent Exotic Grassland. Plant Ecology **198**(1):37-46.

Coupland, R.T. 1961. A reconsideration of grassland classification in the Northern Great Plains of North America. The Journal of Ecology **49**(1):135-167.

Culley, J.L.B., Dow, B.K., Presant, E.W. and MacLean, A.J. 1982. Recovery of productivity of Ontario soils disturbed by an oil pipeline installation. Canadian Journal of Soil Science **62**(2):267-279.

D'Antonio, C., and Meyerson, L.A. 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restoration Ecology **10**(4):703-713.

De Caceres, M., Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. Ecology, URL http://sites.google.com/site/miqueldecaceres/

De Keyser, E.S., Dennhardt, L.A., and Hendrickson, J. 2015. Kentucky bluegrass (*Poa pratensis*) invasion in the Northern Great Plains: a story of rapid dominance in an endangered ecosystem. Weed Science **64**(3):409-420.

De Mendiburu, F. 2017. agricolae: Statistical Procedures for Agricultural Research. R package version 1.2-6. https://CRAN.R-project.org/package=agricolae

Delach, A. and Kimmerer, R.W. 2002. The effect of *Polytrichum piliferum* on seed germination and establishment on iron mine tailings in New York. The Bryologist **105**(2):249-255.

Desserud, P., Gates, C.C., Adams, B. and Revel, R.D. 2010. Restoration of foothills rough fescue grassland following pipeline disturbance in southwestern Alberta. Journal of Environmental Management **91**(12):2763-2770.

Desserud, P.A. and Hugenholtz, C. 2015. Do three invasive species: *Amaranthus blitoides*, *Descurainia sophia* and *Bassia scoparia*, respond to soil properties? Ecological Restoration **33**(2):127-130.

Desserud, P.A. and Hugenholtz, C.H. 2017. Restoring industrial disturbances with native hay in Mixedgrass Prairie in Alberta. Ecological Restoration **35**(3):228-236.

Desserud, P.A. and Naeth, M.A. 2011. Promising results restoring grassland disturbances with native hay (Alberta). Ecological Restoration **29**(3):215-219.

Desserud, P.A. and Naeth, M.A. 2013. Natural recovery of rough fescue (*Festuca hallii* (Vasey) Piper) grassland after disturbance by pipeline construction in Central Alberta, Canada. Natural Areas Journal **33**(1):91-98.

Desserud, P.A. and Naeth, M.A. 2014. Predicting grassland recovery with a state and transition model in a natural area, Central Alberta, Canada. Natural Areas Journal **34**(4):429-442.

DiVittorio, C.T., Corbin, J.D. and D'Antonio, C.M. 2007. Spatial and temporal patterns of seed dispersal: an important determinant of grassland invasion. Ecological Applications **17**(2):311-316.

Dixon, J.M. 2000. *Koeleria macrantha* (Ledeb.) Schultes (*K. alpigena* Domin, *K. cristata* (L.) Pers. pro parte, *K. gracilis* Pers., *K. albescens* auct. non DC.). Journal of Ecology **88**(4):709-726.

Dodds, W.K., Gudder, D.A. and Mollenhauer, D. 1995. The ecology of *Nostoc*. Journal of Phycology **31**(1):2-18.

Dormaar, J.F. and Smoliak, S. 1985. Recovery of vegetative cover and soil organic matter during revegetation of abandoned farmland in a semiarid climate. Journal of Range Management **38**(6):487-491.

Elsinger, M.E. 2009. Reclamation status of plains rough fescue grasslands at Rumsey Block after well site and pipeline disturbance. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Eriksson, A. and Eriksson, O. 1997. Seedling recruitment in semi-natural pastures: the effects of disturbance, seed size, phenology and seed bank. Nordic Journal of Botany **17**(5):469-480.

Eschtruth, A.K. and Battles, J.J. 2009. Assessing the relative importance of disturbance, herbivory, diversity, and propagule pressure in exotic plant invasion. Ecological Monographs **79**(2):265-280.

Evans, R.D. and Belnap, J. 1999. Long-term consequences of disturbance on nitrogen dynamics in an arid ecosystem. Ecology **80**(1):150-160.

Evans, R.D., Rimer, R., Sperry, L. and Belnap, J. 2001. Exotic plant invasion alters nitrogen dynamics in an arid grassland. Ecological Applications **11**(5):1301-1310.

Galinato, M.I. and Van der Valk, A.G. 1986. Seed germination traits of annuals and emergents recruited during drawdowns in the Delta Marsh, Manitoba, Canada. Aquatic Botany **26**:89-102.

Gardiner, D.T. 1993. Revegetation status of reclaimed abandoned mined land in western North Dakota. Arid Land Research and Management **7**(1):79-84.

Gauthier, D.A., and Wiken, E. B. 2003. Monitoring the conservation of grassland habitats, prairie ecozone, Canada. Environmentavl Monitoring and Assessment **88**(1):343-364.

Gelbard, J.L. and Belnap, J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology **17**(2):420-432.

Ghaderi-Far, F., Gherekhloo, J., and Alimagham, M. 2010. Influence of environmental factors on seed germination and seedling emergence of yellow sweet clover. Planta Daninha **28**(3):463-469.

Gioria, M., Jarosik, V., and Pysek, P. 2014. Impact of invasions by alien plants on soil seed bank communities: emerging patterns. Evolution and Systematics **16**:132-142.

Gonzalez, S. and Ghermandi, L. 2008. Postfire seed bank dynamics in semiarid grasslands. Plant Ecology **199**:175-185.

Government of Alberta. 2001. Native plant revegetation guidelines for Alberta.

Grace, J.B. and Harrison, J.S. 1986. The biology of Canadian weeds.: 73. *Typha latifolia* L., *Typha angustifolia* L. and *Typha* x *glauca* Godr. Canadian Journal of Plant Science **66**(2):361-379.

Gramineae Services Ltd. 2013. Recovery strategies for industrial development in native prairie: the Dry Mixedgrass natural subregion of Alberta. Prepared by Gramineae Services Ltd. for Alberta Environment and Sustainable Resource Development.

Gross, J. and Ligges, U. 2015. nortest: Tests for normality. R package version 1.0-4. https://CRAN.R-project.org/package=nortest

Guo, Y., Zhao, H., Zuo, X., Drake, S. and Zhao, X. 2008. Biological soil crust development and its topsoil properties in the process of dune stabilization, Inner Mongolia, China. Environmental Geology **54**(3):653-662.

Halvorson, G.A. and Bauer, A. 1984. Yield and botanical composition of a grass-legume mixture on reclaimed land as affected by N and P fertilizer. Agronomy Journal **76**(3):355-358.

Hamdoun, A.M. 1972. Regenerative capacity of root fragments of *Cirsium arvense* (L.) Scop. Weed Research **12**(2):128-136.

Hammermeister, A.M., Naeth, M.A., Schoenau, J.J. and Biederbeck, V.O. 2003. Soil and plant response to wellsite rehabilitation on native prairie in southeastern Alberta, Canada. Canadian Journal of Soil Science **83**(5):507-519.

Hansen, M.J. and Clevenger, A.P. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation **125**(2):249-259.

Harker, K.N., Baron, V.S., Chanasyk, D.S., Naeth, M.A., and Stevenson, F.C. 2000. Grazing intensity effects on weed populations in annual and perennial pasture systems. Weed Science **48**(2):231-238.

Hawkes, C.V. 2004. Effects of biological soil crusts on seed germination of four endangered herbs in a xeric Florida shrubland during drought. Plant Ecology **170**(1):121-134.

Henderson, D.C. and Naeth, M.A. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions **7**(4):639-650.

Hickman, L.K. 2010. Reclamation Outcomes on Energy Disturbances in Silver Sagebrush Communities. M. Sc. Thesis, University of Calgary, Department of Environmental Science. Calgary, Alberta.

Hickman, L.K., Desserud, P.A., Adams, B.W. and Gates, C.C. 2013. Effects of disturbance on silver sagebrush communities in Dry Mixed-Grass prairie. Ecological Restoration **31**(3):274-282.

Hopfensperger, K.N. 2007. A review of similarity between seed bank and standing vegetation across ecosystems. Oikos **116**(9):1438-1448.

Hutchings, M.J. and Booth, K.D. 1996. Studies on the feasibility of re-creating chalk grassland vegetation on ex-arable land. I. The potential roles of the seed bank and the seed rain. Journal of Applied Ecology **33**(5):1171-1181.

Iverson, L.R. and Wali, M.K. 1982. Buried, viable seeds and their relation to revegetation after surface mining. Journal of Range Management **35**(5):648-652.

Johansen, J.R. 1993. Cryptogamic crusts of semiarid and arid lands of North America. Journal of Phycology **29**:140-147.

Johnston, D. 2011. Movement of weed seeds in reclaimed areas. Restoration Ecology **19**(4):446-449.

Johnston, A., Smoliak, S. and P.W. Stringer. 1969. Viable seed populations in Alberta prairie topsoils. Canadian Journal of Plant Science **49**(1):75-82.

Jong, E.D., and Button, R.G. 1973. Effects of pipeline installation on soil properties and productivity. Canadian Journal of Soil Science **53**(1):37-47.

Kinucan, R.J. and Smeins, F.E. 1992. Soil seed bank of a semiarid Texas grassland under three long-term (36-years) grazing regimes. The American Midland Naturalist **128**(1):11-21.

Klemow, K.M., and Raynal, D.J. 1981. Population ecology of *Melilotus alba* in a limestone quarry. The Journal of Ecology **69**(1):33-44.

Klimes, J. 2007. Bud banks and their role in vegetative regeneration–a literature review and proposal for simple classification and assessment. Perspectives in Plant Ecology, Evolution and Systematics **8**(3):115-129.

Knott, D.M., Wenner, E.L. and Wendt, P.H. 1997. Effects of pipeline construction on the vegetation and macrofauna of two South Carolina, USA salt marshes. Wetlands **17**(1):65-81.

Lane, D.R., Coffin, D.P. and Lauenroth, W.K. 1998. Effects of soil texture and precipitation on above-ground net primary productivity and vegetation structure across the Central Grassland region of the United States. Journal of Vegetation Science **9**(2):239-250.

Langhans, T.M., Storm, C. and Schwabe, A. 2009. Biological soil crusts and their microenvironment: impact on emergence, survival and establishment of seedlings. Flora-Morphology, Distribution, Functional Ecology of Plants **204**(2):157-168.

Laughlin, D.C. 2003. Lack of native propagules in a Pennsylvania, USA, limestone prairie seed bank: futile hopes for a role in ecological restoration. Natural Areas Journal **23**(2):158-164.

Lenth, R.V. 2016. Least-Squares Means: The R Package lsmeans. Journal of Statistical Software **69**(1): 1-33. doi:10.18637/jss.v069.i01

Levassor, C., M. Ortega, and Peco, B. 1990. Seed bank dynamics of Mediterranean pastures subjected to mechanical disturbance. Journal of Vegetation Science **1**(3):339-344.

Li, X. Jia, X., Long, L., and Zerbe, S. 2005. Effects of biological soil crusts on seed bank, germination and establishment of two annual plant species in the Tengger Desert (N China). Plant and Soil **277**(1):375-385.

Li, X.Y., Liu, L.Y., and Wang, J.H. 2004. Wind tunnel simulation of aeolian sandy soil erodibility under human disturbance. Geomorphology **59**(1):3-11.

Luo, W. and Zhao, W. 2015. Burial depth and diameter of the rhizome fragments affect the regenerative capacity of a clonal shrub. Ecological Complexity **23**:34-40.

Low, C.H. 2016. Impacts of a six year old pipeline right of way on *Halimolobos virgata* (Nutt.) OE Schulz (slender mouse ear cress), native Dry Mixedgrass prairie uplands, and wetlands. Ph. D. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Lyseng, M.P., Bork, E.W., Hewins, D.B., Alexander, M.J., Carlyle, C.N., Chang, S.X., and Willms, W.D. 2018. Long-term grazing impacts on vegetation diversity, composition and exotic species presence across an aridity gradient on northern temperate grasslands. Plant Ecology **219**(6):649-663.

Ma, M., Zhou, X. and Du, G. 2010. Role of soil seed bank along a disturbance gradient in an alpine meadow on the Tibet plateau. Flora-Morphology, Distribution, Functional Ecology of Plants **205**(2):128-134.

MacDougall, A.S. and Turkington, R. 2005. Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology **86**(1):42-55.

Macdonald, I.D. 2005. Status of the slender mouse-ear-cress (*Halimolobos virgata*) in Alberta. Fish and Wildlife Division, Alberta Sustainable Resource Development. Alberta Wildlife Status Report No. 55.

Marlette, G.M., and Anderson, J.E. 1986. Seed banks and propagule dispersal in crested-wheatgrass stands. Journal of Applied Ecology **23**(1):161-175.

Martínez-Garza, C., Osorio-Beristain, M., Valenzuela-Galván, D. and Nicolás-Medina, A. 2011. Intra and inter-annual variation in seed rain in a secondary dry tropical forest excluded from chronic disturbance. Forest Ecology and Management **262**(12):2207-2218.

Matthews, J.W. and Spyreas, G. 2010. Convergence and divergence in plant community trajectories as a framework for monitoring wetland restoration progress. Journal of Applied Ecology **47**(5):1128-1136.

McGeehan, S.L. and Naylor, D.V. 1988. Automated instrumental analysis of carbon and nitrogen in plant and soil samples. Communication in Soil Science and Plant Analysis **19**(4):493-505.

Meyer, S.E., Quinney, D., Nelson, D.L. and Weaver, J. 2007. Impact of the pathogen *Pyrenophora semeniperda* on *Bromus tectorum* seedbank dynamics in North American cold deserts. Weed Research **47**(1):54-62.

Moeslund, J.E., Brunbjerg, A.K., Clausen, K.K., Dalby, L., Fløjgaard, C., Juel, A. and Lenoir, J. 2017. Using dark diversity and plant characteristics to guide conservation and restoration. Journal of Applied Ecology.

Naeth, M.A., Bailey, A.W., and McGill, W.B. 1987. Persistence of changes in selected soil chemical and physical properties after pipeline installation in solonetzic native rangeland. Canadian Journal of Soil Science **67**(4):747-763.

Naeth, M.A., Chanasyk, D.S. and McGill, W.B. 1993. Soil temperature regime in mixed prairie rangeland after pipeline construction and operation. Canadian agricultural engineering **35**(2):88-95.

Nannt, M.R. 2014. Impacts of distance to pipeline disturbance on mixed grass prairie and *Halimolobos virgata* (Nutt.) OE Schulz (slender mouse ear cress). M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Nasen, L.C., Noble, B.F. and Johnstone, J.F. 2011. Environmental effects of oil and gas lease sites in a grassland ecosystem. Journal of Environmental Management **92**(1):195-204.

Nemergut, D.R., Anderson, S.P., Cleveland, C.C., Martin, A.P., Miller, A.E., Seimon, A. and Schmidt, S.K. 2007. Microbial community succession in an unvegetated, recently deglaciated soil. Microbial Ecology **53**(1):110-122.

Neville, M., Alexander, M., Adams, B., DeMaere, C., Lawrence, D., and McGillvray, S. 2016. Principles for minimizing surface disturbances in native grassland: principles, guidelines, and tools for all industrial activity in native grassland in the prairie and parkland landscapes of Alberta. Alberta Environment and Parks, Edmonton, Alberta.

Noyd, R.K., Pfleger, F.L., Norland, M.R. and Sadowsky, M.J. 1995. Native prairie grasses and microbial community responses to reclamation of taconite iron ore tailing. Canadian Journal of Botany **73**(10):1645-1654.

Olson, E.R. and Doherty, J.M. 2012. The legacy of pipeline installation on the soil and vegetation of southeast Wisconsin wetlands. Ecological Engineering **39**:53-62.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, K., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson. G.L., Solymos, P., M., Stevens, M.H., Szoecs, E., and Wagner, H. 2017. vegan: Community ecology package. R package version 2.4-4. https://CRAN.R-project.org/package=veg

Ostermann, D.K. 2001. Revegetation assessment of a twelve-year-old pipeline on native rangeland in southern Alberta. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Otfinowski, R., Kenkel, N.C., and Van Acker, R.C. 2008. Reconciling seed dispersal and seed bank observations to predict smooth brome (*Bromus inermis*) invasions of northern prairie. Invasive Plant Science and Management **1**:279-286.

Pärtel, M., Szava-Kovats, R. and Zobel, M. 2011. Dark diversity: shedding light on absent species. Trends in ecology & evolution **26**(3):124-128.

Paul, E.A., Myers, R.J.K. and Rice, W.A. 1971. Nitrogen fixation in grassland and associated cultivated ecosystems. Plant and Soil **35**(1):495-507.

Petherbridge, W.L. 2000. Sod salvage and minimal disturbance pipeline reclamation techniques: implications for native prairie restoration. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Pyke, D.A. 1990. Comparative demography of co-occurring introduced and native tussock grasses: persistence and potential expansion. Oecologia **82**(4):537-543.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Reed, A.W., Kaufman, G.A. and Kaufman, D.W. 2006. Effect of plant litter on seed predation in three prairie types. The American Midland Naturalist **155**(2):278-285.

Ren, L. and Bai, Y. 2016. Smoke and ash effects on seedling emergence from germinable soil seed bank in Fescue Prairie. Rangeland Ecology & Management **69**(6):499-507.

Renne, I.J. and Tracy, B.F. 2007. Disturbance persistence in managed grasslands: shifts in aboveground community structure and the weed seed bank. Plant Ecology **190**(1):71-80.

Rogler, G.A. and Lorenz, R.J. 1983. Crested Wheatgrass: Early History in the United States. Journal of Range Management **36**(1):91-93.

Rowland, J. 2008. Ecosystem impacts of historical shallow gas wells within the CFB Suffield National Wildlife Area. Report submitted February 2, 2008.

Russi, L., Cocks, P.S. and Roberts, E.H. 1992. Hard-seededness and seed bank dynamics of six pasture legumes. Seed Science Research **2**(4):231-241.

Samson, F. and Knopf, F. 1994. Prairie conservation in North America. BioScience **44**(6):418-421.

Samson, F.B., Knopf, F.L. and Ostlie, W.R. 2004. Great Plains ecosystems: past, present, and future. Wildlife Society Bulletin **32**(1):6-15.

Sanderson, M. A., S. C. Goslee, K. D. Klement, and Soder, K.J. 2007. Soil seed bank composition in pastures of diverse mixtures of temperate forages. Agronomy Journal **99**(6):1514-1520.

Savory, A. 1988. Holistic Resource Management. Island Press, Covelo, California.

Schröder, R. and Prasse, R. 2013. Do cultivated varieties of native plants have the ability to outperform their wild relatives? PLoS One **8**(8):e71066.

Shi, P., Xiao, J., Wang, Y.F. and Chen, L.D. 2014. The effects of pipeline construction disturbance on soil properties and restoration cycle. Environmental Monitoring and Assessment **186**(3):1825-1835.

Simmers, S.M., and Galatowitsch, S.M. 2010. Factors affecting revegetation of oil field access roads in semiarid grassland. Restoration Ecology **18**(s1):27-39.

Spehn, E.M., Scherer-Lorenzen, M., Schmid, B., Hector, A., Caldeira, M.C., Dimitrakopoulos, P.G., Finn, J.A., Jumpponen, A., O'donnovan, G., Pereira, J.S. and Schulze, E.D. 2002. The role of legumes as a component of biodiversity in a cross-European study of grassland biomass nitrogen. Oikos **98**(2):205-218.

Sprinkle, J.W. 2010. Bud bank density regulates invasion by exotic plants. M.Sc. Thesis, Oklahoma State University.

Soil Classification Working Group. 1998. The Canadian system of soil classification, Third Edition. Research Branch, Agriculture and Agri-Food Canada. NRC Research Press, Ottawa.

Soon, Y.K., Arshad, M.A., Rice, W.A. and Mills, P. 2000. Recovery of chemical and physical properties of boreal plain soils impacted by pipeline burial. Canadian Journal of Soil Science **80**(3):489-497.

Spribille, T., Tuovinen, V., Resl, P., Vanderpool, D., Wolinski, H., Aime, M.C., Schneider, K., Stabentheiner, E., Toome-Heller, M., Thor, G. and Mayrhofer, H. 2016. Basidiomycete yeasts in the cortex of ascomycete macrolichens. Science **353**(6298):488-492.

Stoa, T.E. 1933. Persistence of viability of sweet clover seed in a cultivated soil. Journal of the American Society of Agronomy **25**:177-81.

Suding, K.N. 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Annual Review of Ecology, Evolution, and Systematics **42**:465-487.

Tracy, B.F., and M.A. Sanderson. 2000. Seedbank diversity in grazing lands of the Northeast United States. Journal of Range Management **53**(1):114-118.

Thompson, K., Band, S.R., and Hodgson, J.G. 1993. Seed size and shape predict persistence in soil. Functional Ecology **7**(2):236-241.

Travnicek, A.J., Lym, R.G. and Prosser, C. 2005. Fall-prescribed burn and spring-applied herbicide effects on Canada thistle control and soil seedbank in a northern mixed-grass prairie. Rangeland Ecology & Management **58**(4):413-422.

Turkington, R.A., Cavers, P.B. and Rempel, E. 1978. The biology of Canadian weeds.: 29. *Melilotus alba* Desr. and *M. officinalis* (L.) Lam. Canadian Journal of Plant Science **58**(2):523-537.

Van Riper, L.C. and Larson, D.L. 2009. Role of invasive *Melilotus officinalis* in two native plant communities. Plant Ecology **200**(1):129-139.

Van Riper, L.C., Larson, D.L. and Larson, J.L. 2010. Nitrogen-limitation and invasive sweetclover impacts vary between two Great Plains plant communities. Biological Invasions **12**(8):2735-2749.

Vaness, B.M. and Wilson, S.D. 2007. Impact and management of crested wheatgrass (*Agropyron cristatum*) in the northern Great Plains. Canadian Journal of Plant Science **87**(5):1023-1028.

Viall, E.M., Gentry, L.F., Hopkins, D.G., Ganguli, A.C. and Stahl, P. 2014. Legacy effects of oil road reclamation on soil biology and plant community composition. Restoration Ecology **22**(5):625-632.

Wagner, M., Heinrich, W., and Jetschke, G. 2006. Seed bank assembly in an unmanaged ruderal grassland recovering from long-term exposure to industrial emissions. Acta Oecologica **30**:342-352.

Wainwright, C.E., Staples, T.L., Charles, L.S., Flanagan, T.C., Lai, H.R., Loy, X., Reynolds, V.A. and Mayfield, M.M. 2017. Links between community ecology theory and ecological restoration are on the rise. Journal of Applied Ecology 00:1-12.

Wali, M.K. 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. Plant and Soil **213**(1-2):195-220.

Wang, W., Liu, Y., Li, D., Hu, C. and Rao, B. 2009. Feasibility of cyanobacterial inoculation for biological soil crusts formation in desert area. Soil Biology and Biochemistry **41**(5):926-929.

Warren, R.J., Bahn, V. and Bradford, M.A. 2012. The interaction between propagule pressure, habitat suitability and density-dependent reproduction in species invasion. Oikos **121**(6):874-881.

White, S.R., Bork, E.W., Karst, J., and Cahill Jr., J.F. 2012. Similarity between grassland vegetation and seed bank shifts with altered precipitation and clipping, but not warming. Community Ecology **13**(2):129-136.

Willems, J.H. and Bik, L.P.M. 1998. Restoration of high species density in calcareous grassland: the role of seed rain and soil seed bank. Applied Vegetation Science **1**(1):91-100.

Williams, E.D. 1984. Changes during 3 years in the size and composition of the seed bank beneath a long-term pasture as influenced by defoliation and fertilizer regime. Journal of Applied Ecology **21**:603-615.

Willms, W.D., Ellert, B.H., Janzen, H.H. and Douwes, H. 2005. Evaluation of native and introduced grasses for reclamation and production. Rangeland Ecology and Management **58**(2):177-183.

Willms, W.D., and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Wilson, S.D. 1988. The effects of military tank traffic on prairie: a management model. Environmental Management **12**(3):397-403.

Wilson, S.D. 1989. The suppression of native prairie by alien species introduced for revegetation. Landscape and Urban Planning **17**(2):113-119.

Wilson, S.D. and Pärtel, M. 2003. Extirpation or coexistence? Management of a persistent introduced grass in a prairie restoration. Restoration Ecology **11**(4):410-416.

Wolf, J.J., Beatty, S.W., and Carey, G. 2008. Invasion by sweet clover (Melilotus) in montane grasslands, Rocky Mountain National Park. Annals of the Association of American Geographers **93**(3): 531-543.

Wolf, J.J., Beatty, S.W. and Seastedt, T.R. 2004. Soil characteristics of Rocky Mountain National Park grasslands invaded by *Melilotus officinalis* and *M. alba*. Journal of Biogeography **31**(3):415-424.

Xiao, J., Wang, Y.F., Shi, P., Yang, L. and Chen, L.D. 2014. Potential effects of large linear pipeline construction on soil and vegetation in ecologically fragile regions. Environmental Monitoring and Assessment **186**(11):8037-8048.

Yeomans, J.C. and Bremner, J.M. 1991. Carbon and nitrogen analysis of soils by automated combustion techniques. Communication in Soil Science and Plant Analysis **22**(9-10):843-850.

Zang, Z.Q., Shu, W.S., Lan, C.Y., and Wong, M.H. 2001. Soil seed bank as an input of seed source in revegetation of lead/zinc mine tailings. Restoration Ecology **9**(4):378-385.

6.9 Figures and Tables

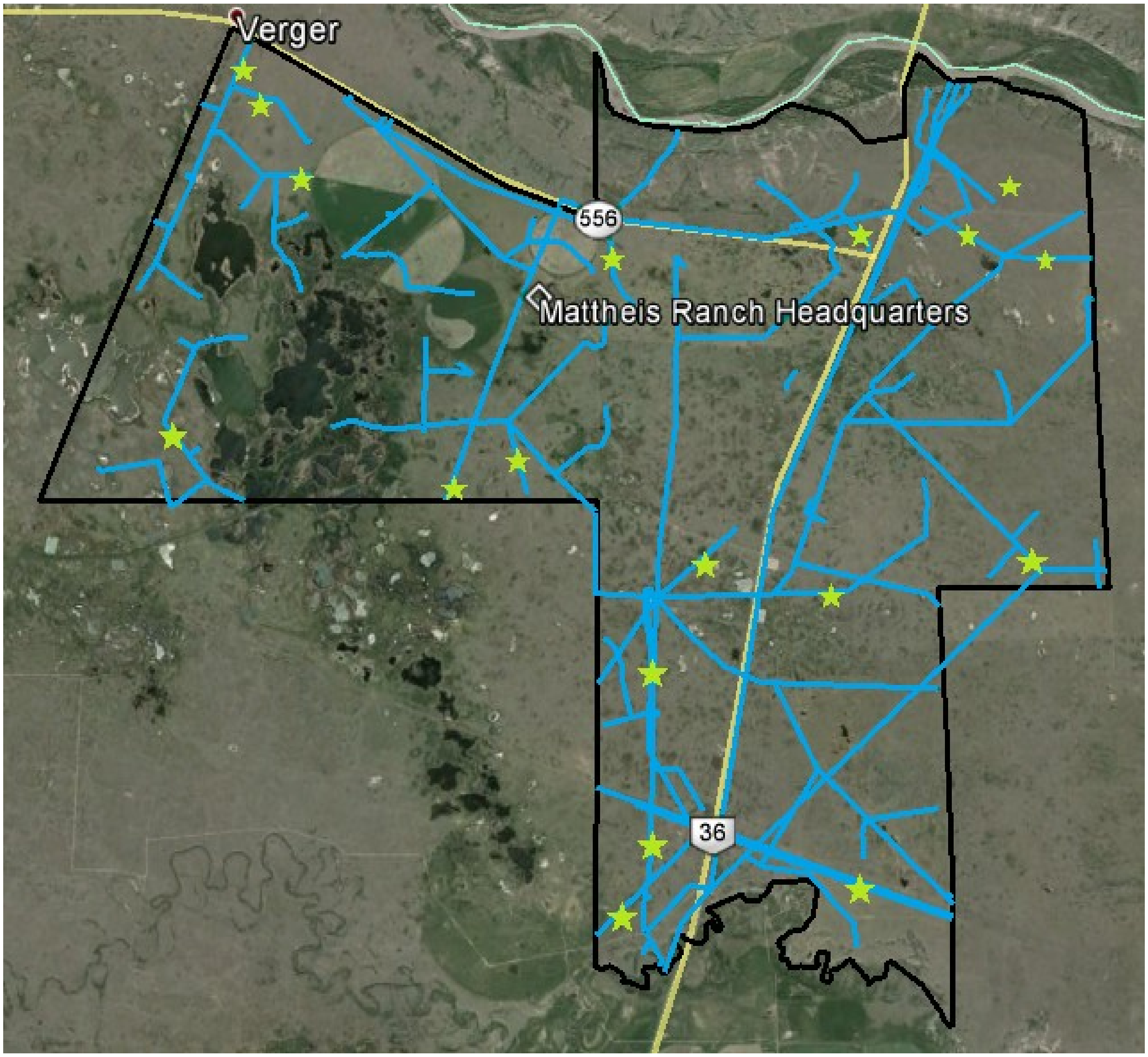


Figure 6.1. Map of the 18 study site locations (green stars) within the 5,200 ha Mattheis Research Ranch situated 40 km north of Brooks, Alberta on Highway 36, a component of the Rangeland Research Institute affiliated with the University of Alberta.

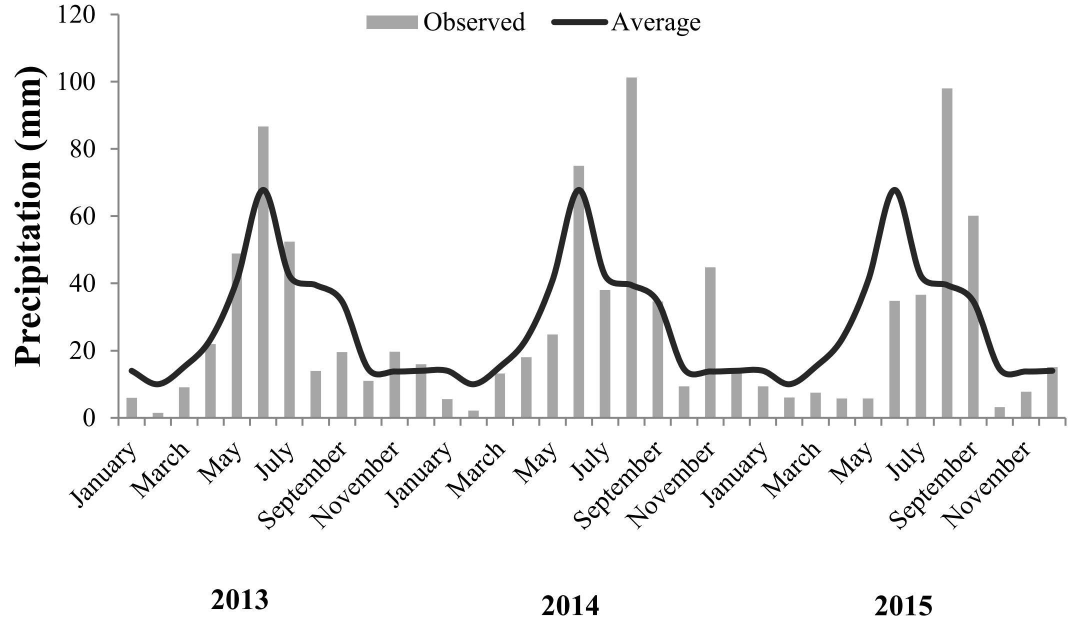


Figure 6.2. Monthly observed precipitation (mm) at the University of Alberta Mattheis Research Ranch between January of 2013 and December of 2015, inclusive (Alberta Agriculture and Forestry, 2016). Data were acquired from the Verger Monitoring Station, located NW of the Mattheis Ranch. The 30-year average for the area was obtained from readings taken at T22 – R14 – W4.

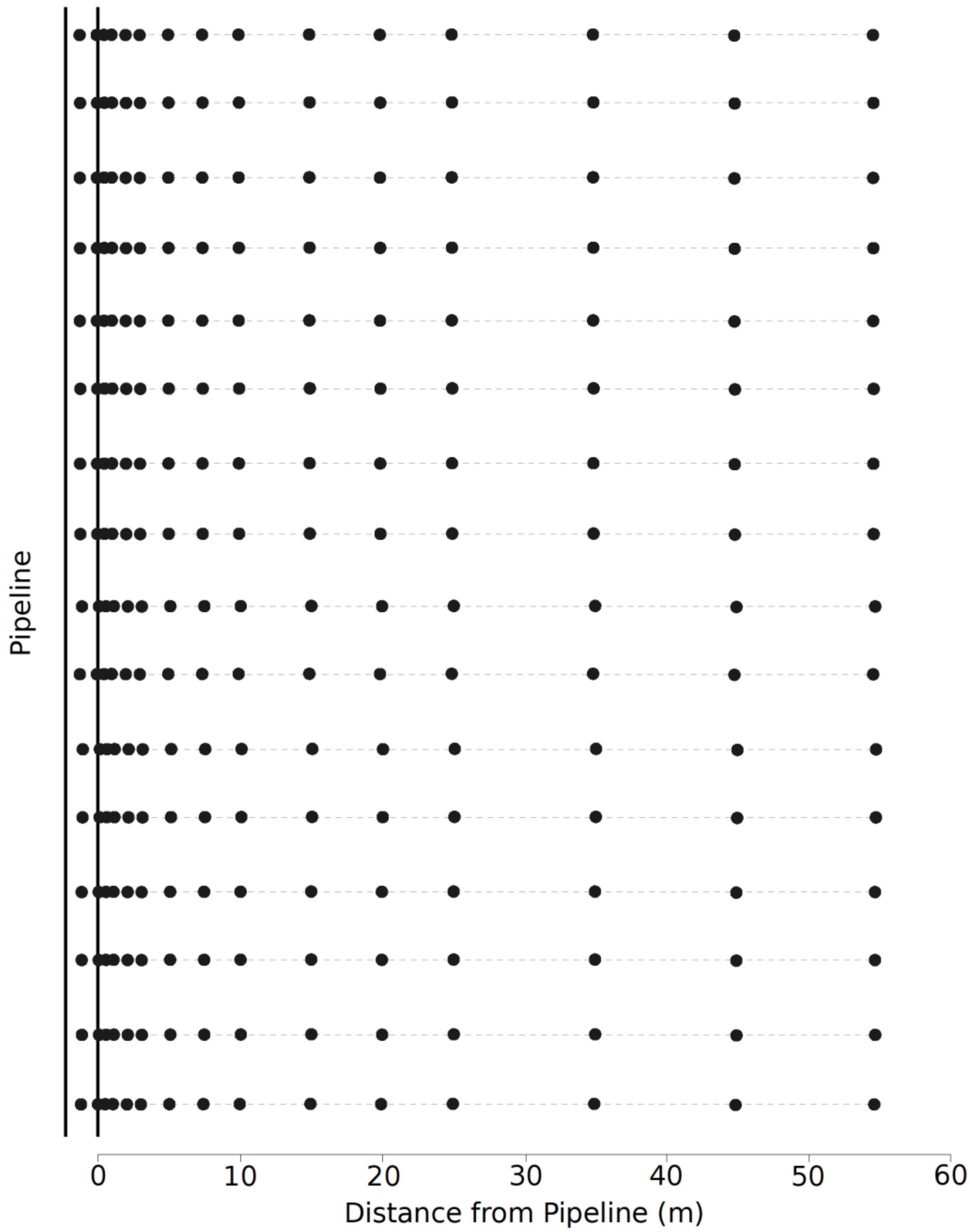


Figure 6.3. Seed bank sampling intensity along 55 m long transects placed perpendicular to the pipeline trench. Soil seed bank samples were drawn at the following distances: pipeline center, edge of soil trench, and 0.5, 1, 2, 3, 5, 7.5, 10, 15, 20, 25, 35, 45 and 55 m. Plant community foliar cover and basal cover (litter, exposed soil, etc.) were measured at the same points with a 0.25 m^2 quadrat with one exception; no cover was estimated at the pipeline trench edge as it would simultaneously describe cover on the trench and in the adjacent plant community.

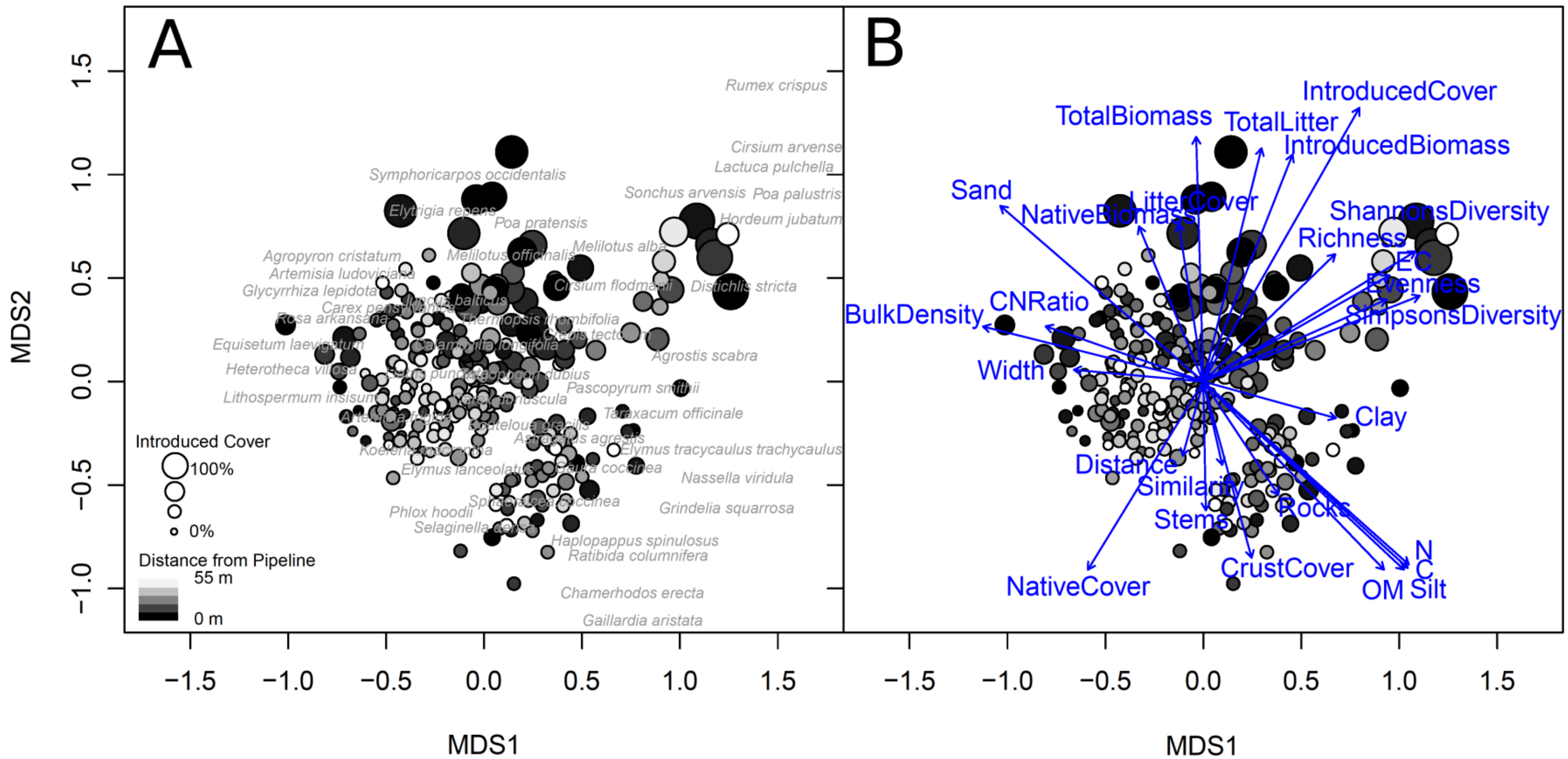


Figure 6.4. NMDS ordination biplots of plant community composition (stress = 0.23, dimensions = 2, distance = Bray-Curtis), including A) species with significant correlations to the axes ($P < 0.001$), and B) overlaid vectors of significant soil conditions, ground cover, and plant community cover attributes ($P < 0.05$). Larger symbols indicate greater introduced species cover, and darker coloured symbols indicate plant communities (i.e. plots) closer to pipeline disturbance, while lighter colours indicate plant communities further away that were likely undisturbed.

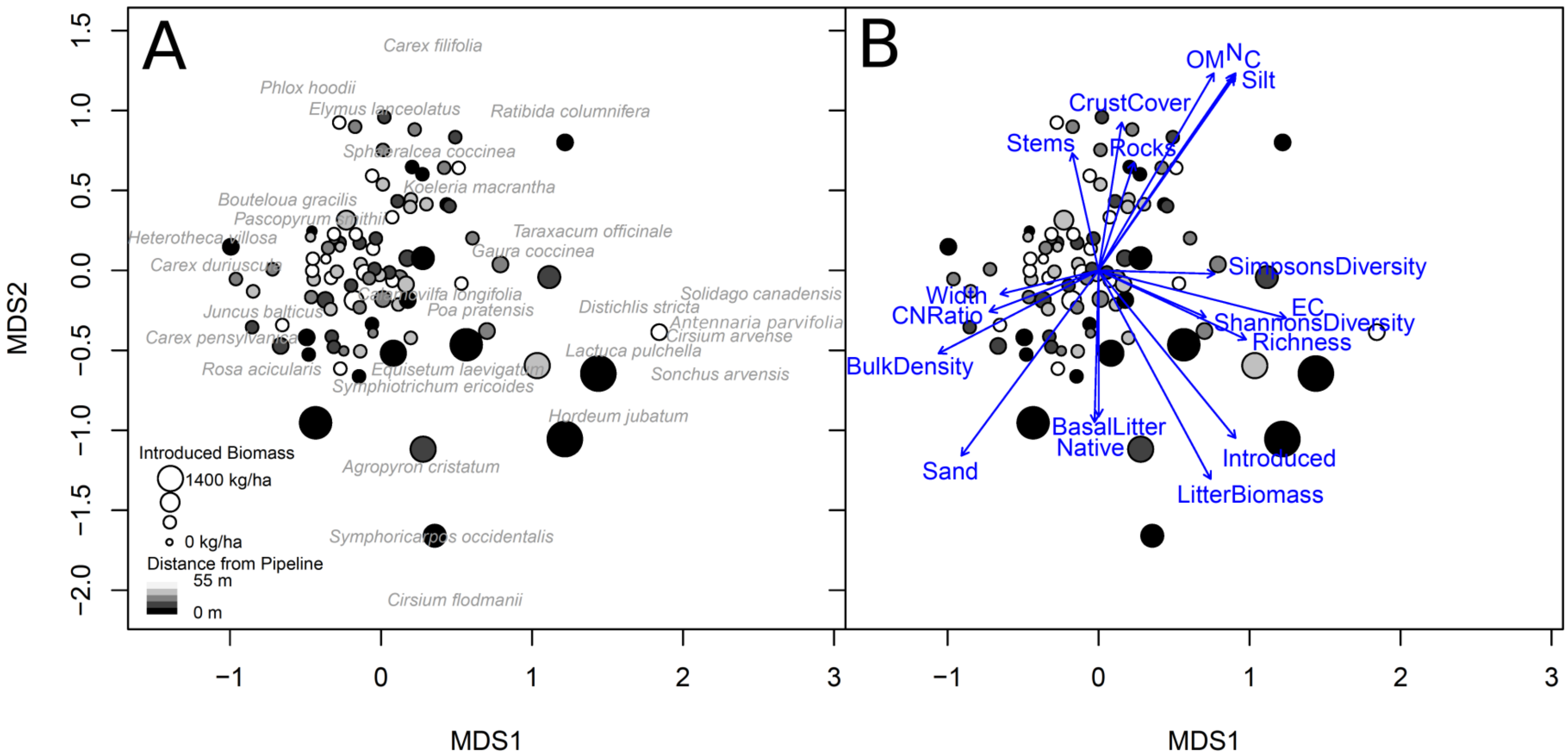


Figure 6.5. NMDS ordination biplot of plant community species biomass assessed at the species level (stress = 0.22, dimensions = 2, distance = Bray-Curtis), including A) species with significant correlations to the axes ($P < 0.001$), and B) overlaid vectors of significant soil conditions, ground cover and vegetation biomass attributes (i.e. richness, diversity, etc.) ($P < 0.05$). Larger symbols indicate greater introduced species biomass, and darker coloured symbols indicate plant communities (i.e. plots) closer to pipeline disturbance, while lighter colours indicate plant communities further away that were likely undisturbed.

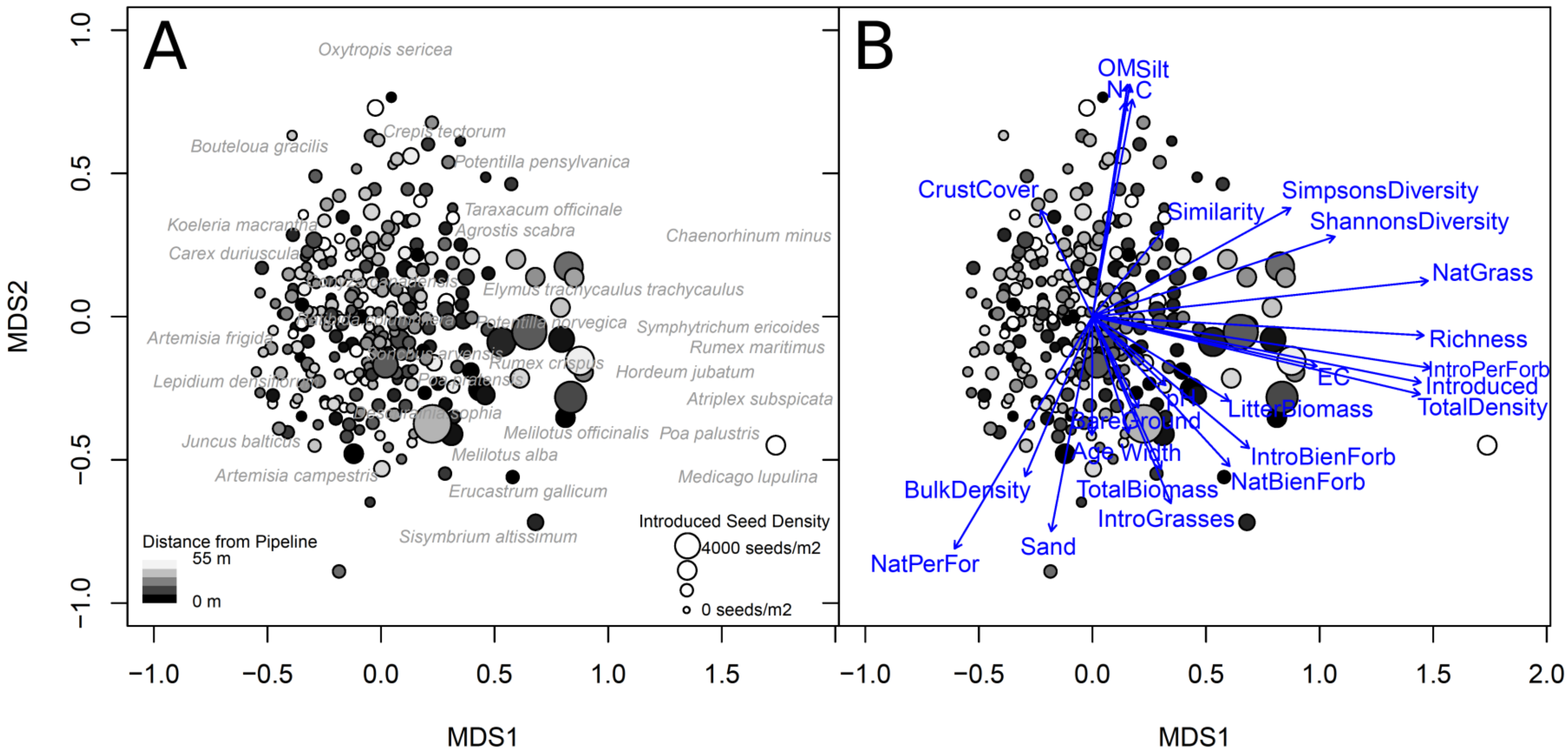


Figure 6.6. NMDS ordination biplot of seed bank composition (stress = 0.28, dimensions = 2, distance = Bray-Curtis), including A) species in the seed bank with significant correlations to the axes ($P < 0.001$), and B) overlaid vectors of significant soil conditions, ground cover, and seed bank community attributes ($P < 0.05$). Larger symbols indicate greater introduced species seed density, and darker coloured symbols indicate seed bank communities (i.e. plots) closer to pipeline disturbance, while lighter colours indicate communities further away that were likely undisturbed.

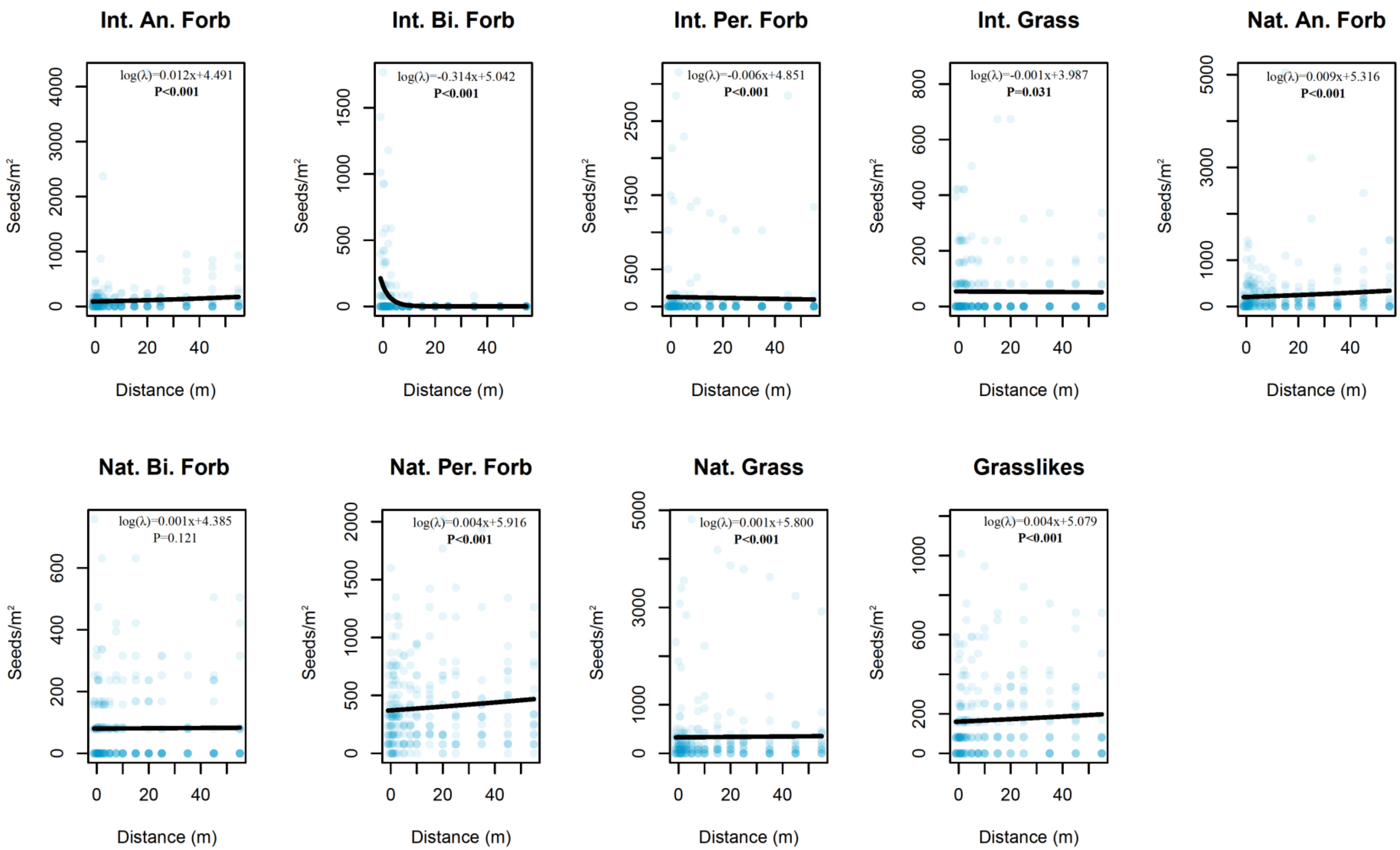


Figure 6.7. Poisson regressions of seed densities (seeds/m^2) for various plant lifeforms at increasing distance (m) from pipeline trench.

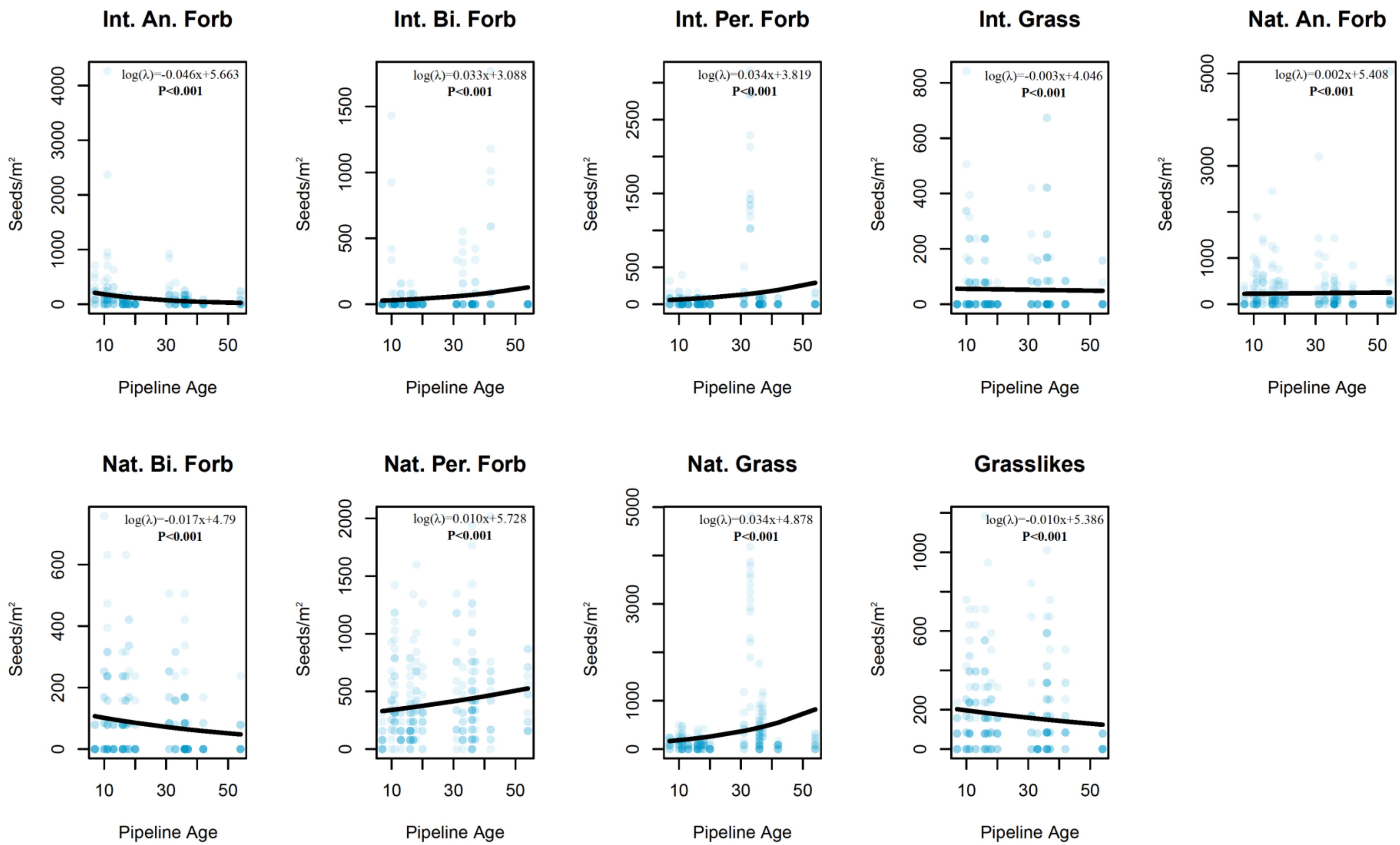


Figure 6.8. Poisson regressions of seed densities (seeds/m^2) for plant lifeforms in relation to various pipeline ages (years).

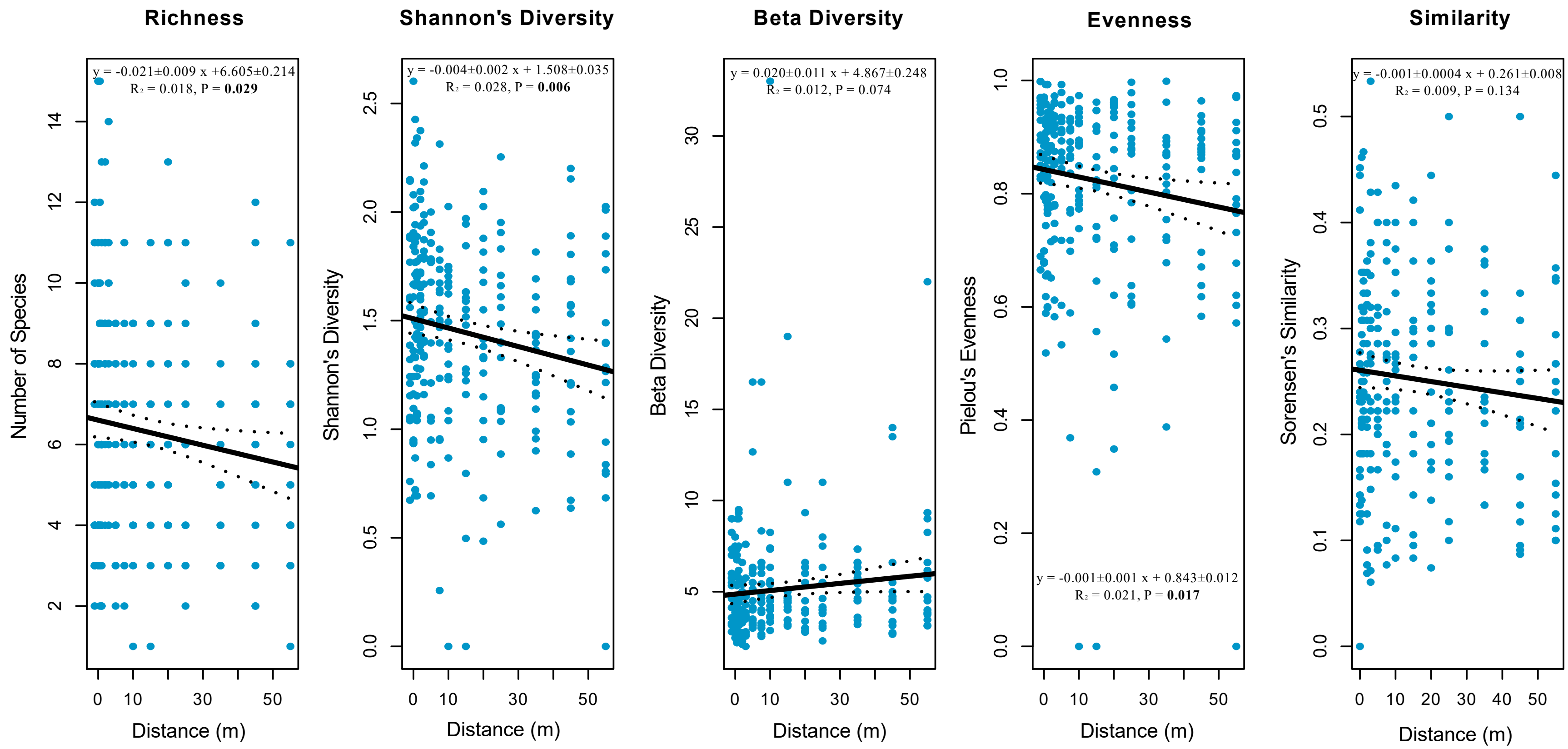


Figure 6.9. Linear regressions for indices of seed bank richness, Shannon's diversity, beta diversity, Pielou's evenness and Sorensen's similarity to the aboveground plant community, along increasing distances from pipeline disturbance, with 95% confidence intervals. Significant relationships have bolded P-values.

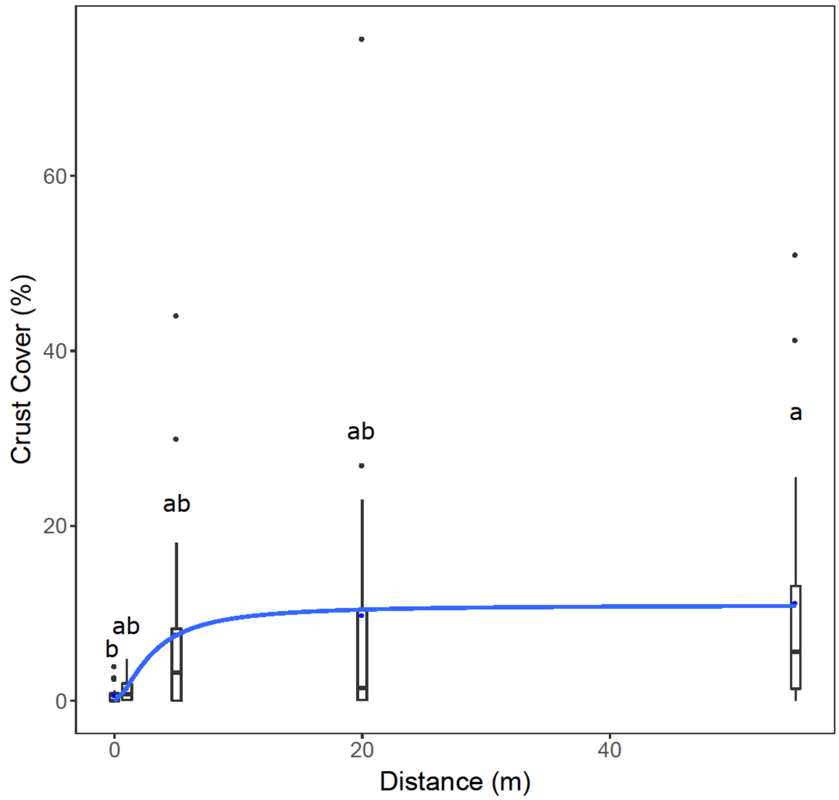


Figure 6.10. Pipeline impacts on biological soil crust cover within the proximity of disturbed trenches. Median crust cover significantly differed at all sampling distances (X^2 = 16.69, P = 0.002), each median is accompanied by the IQR range in boxplots. The smooth blue line represents the non-linear function for soil crust cover changes over distance (Θ_1 = 10.56 ± 1.19, t = 8.849, P < 0.001; Θ_2 = 2.57 ± 1.41, t = 1.823, P = 0.069; Θ_3 = -0.71 ± 0.34, t = -2.065, P = 0.040; R^2 = 0.084).

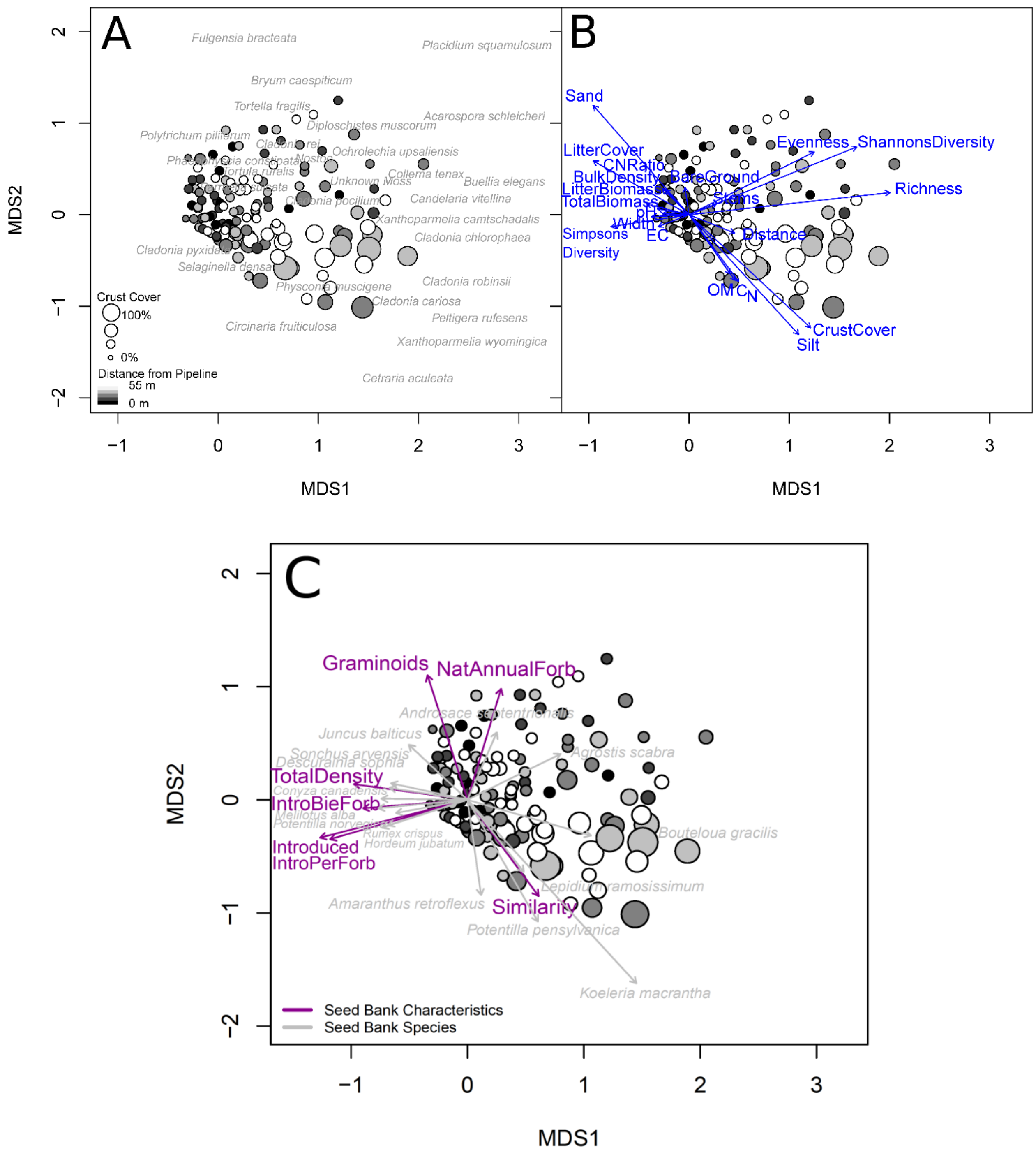


Figure 6.11. NMDS ordination biplot of biological soil crust composition (stress = 0.13, dimensions = 2, distance = Bray-Curtis) including lichens, mosses, and spike-mosses, and metrics associated with the resulting axes. Panel A includes significant biological soil crust species ($P < 0.05$). Panel B includes biplots of edaphic factors and soil crust indices ($P < 0.05$). Panel C includes significant ($P < 0.05$) seed bank composition characteristics (purple) and individual species from the seed bank (grey) as vectors. Larger symbols indicate greater biological soil crust cover, darker coloured symbols indicate plots closer to the pipeline trench, while lighter colours indicate plots further away that were likely undisturbed.

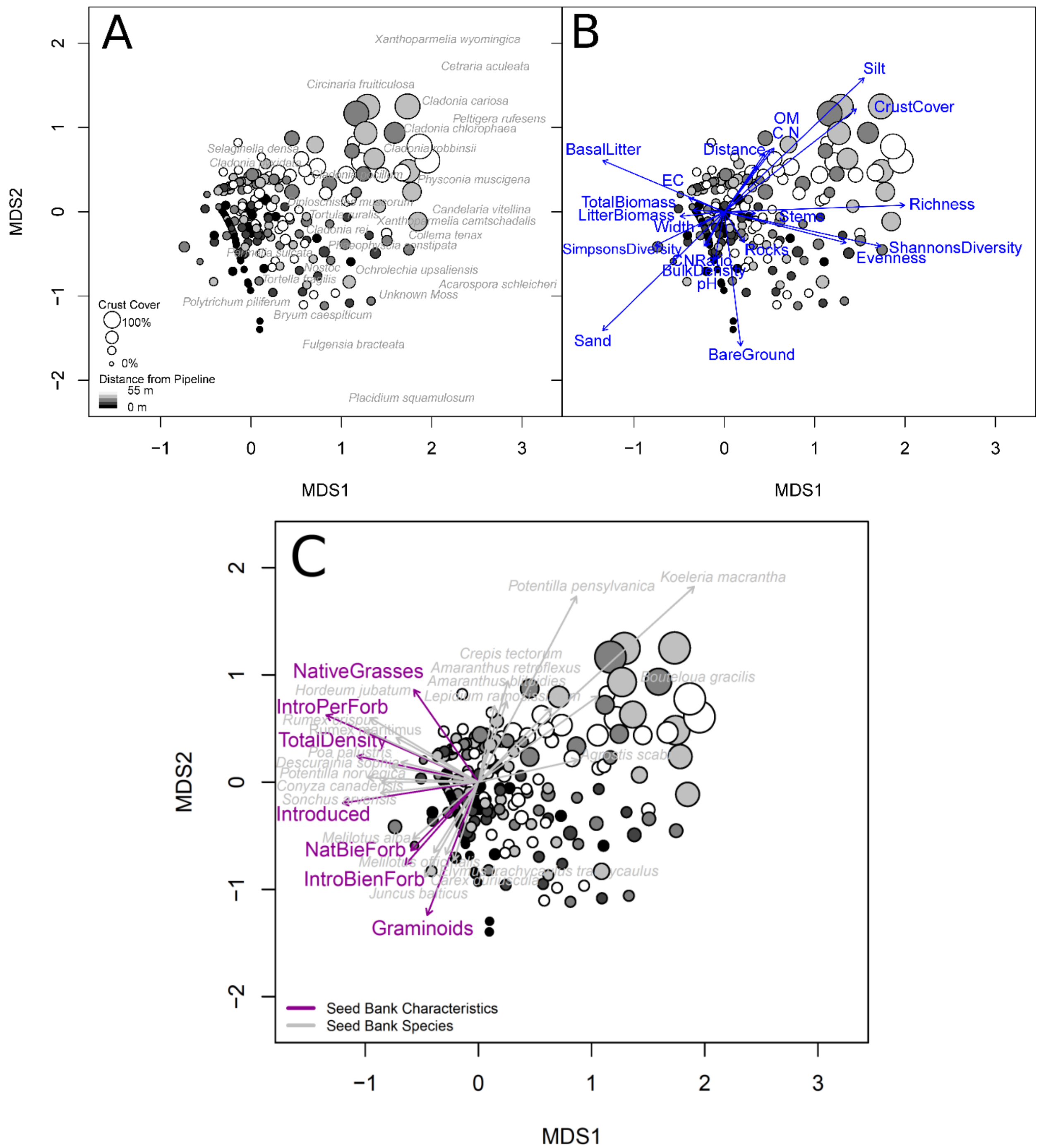


Figure 6.12. NMDS ordination biplot of biological soil crust composition and ground cover dynamics (stress = 0.14, dimensions = 2, distance = Bray-Curtis), including the proportion of soil exposure and litter cover, and all metrics significantly related to the resulting axes. Panel A includes significant biological soil crust species ($P < 0.05$). Panel B includes biplots of edaphic factors and soil crust indices ($P < 0.05$). Panel C includes significant ($P < 0.05$) seed bank composition characteristics (purple) and individual species from the seed bank (grey) as vectors. Larger symbols indicate greater biological soil crust cover, darker coloured symbols indicate plots closer to the pipeline trench, while lighter colours indicate plots further away that were likely undisturbed.

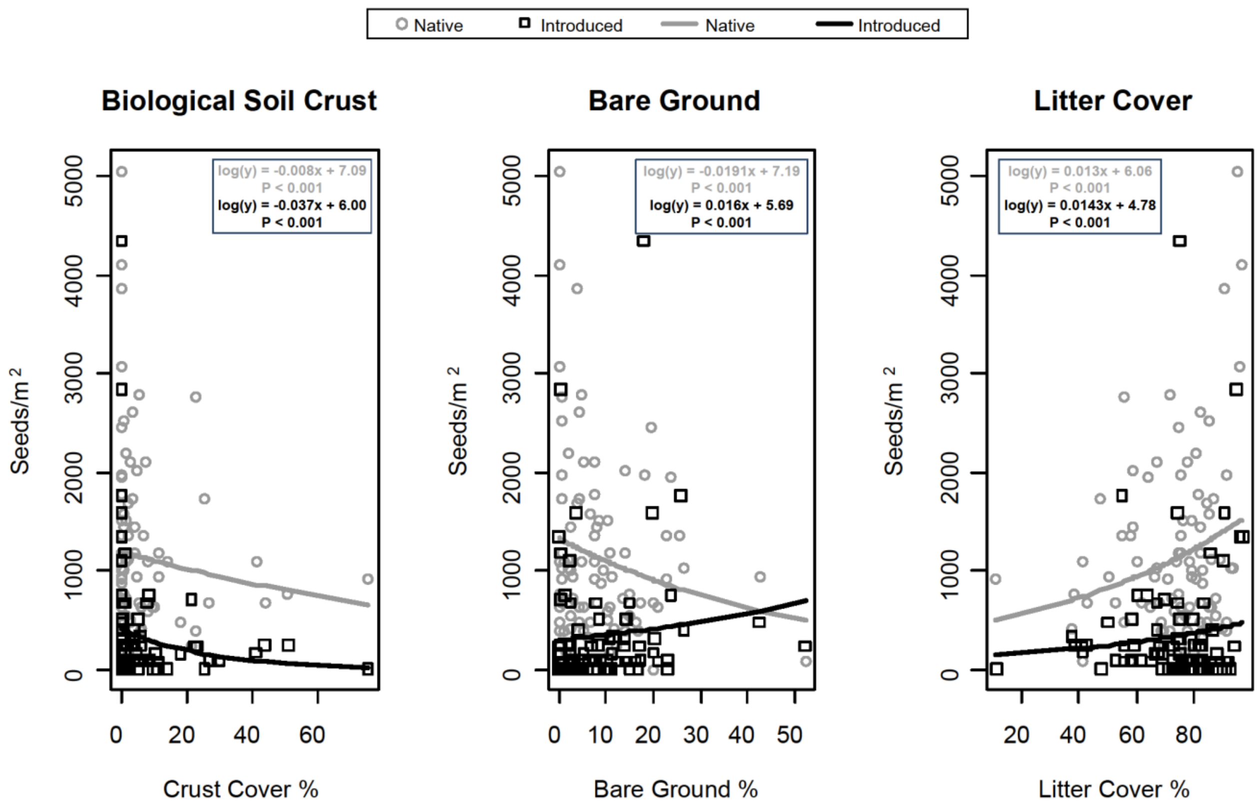


Figure 6.13. Poisson regressions describing the relationship between the total seed density of native (dark lines) and introduced (light lines) plant species in relation to increasing amounts of either A) surface biological soil crust, B) bare mineral soil, and C) litter cover.

Table 6.1. Relative rankings (R) of the top 15 species based on mean foliar cover for vegetation and mean seed density for the seed bank (± SE) stratified by samples from the native plant community (25 m to 55 m) and pipeline trench (center). Significant differences in foliar cover and seed density between the trench and native grassland for the top 15 species were tested using generalized linear mixed models (binomial distribution for cover and Poisson for seed density).

	Foliar Cover (%)					Seed Density (seeds/m^2)				
Species	**Native**	**R**	**Trench**	**R**	**P-value**	**Native**	**R**	**Trench**	**R**	**P-value**
Agropyron cristatum	0.01 (±0.01) b		3.07 (±1.89) a	10	**<0.001**	1.1 (±1.1)		0 (±0)		
Agrostis scabra	0.11 (±0.06)		0.31 (±0.17)			39.9 (±13.2) a	10	18.13 (±10.7) b		**<0.001**
Androsace septentrionalis	0.01 (±0.01)		0.04 (±0.01)			261.1 (±80.3) a	2	46.2 (±22.8) b	9	**<0.001**
Artemisia frigida	3.48 (±0.44)	7	3.22 (±0.70)	9	0.452	372.1 (±60.0) a	1	238.0 (±50.6) b	1	**<0.001**
Bouteloua gracilis	18.35 (±2.23) a	2	3.77 (±1.16) b	7	**0.001**	13.2 (±7.0)		14.0 (±14.0)		
Calamovilfa longifolia	28.88 (±4.29) a	1	13.95 (±3.49) b	1	**0.001**	2.3 (±1.6)		14.0 (±14.0)		
Campanula rotundifolia	0.06 (±0.05)		0.03 (±0.03)			17.0 (±7.3) a	13	0 (±0) b		**<0.001**
Carex duriuscula	3.69 (±0.68)	6	3.85 (±0.74)	6	0.396	123.7 (±27.5) a	4	36.0 (±11.7) b	12	**<0.001**
Carex pensylvanica	0.21 (±0.10)		0.19 (±0.14)			22.3 (±7.5) a	12	0 (±0) b		**<0.001**
Conyza canadensis	0.002 (±0.002)		0.06 (±0.04)			14.8 (±6.8) b	15	28.1 (±20.4) a	14	**<0.001**
Crepis tectorum	0.04 (±0.02)		0.06 (±0.03)			76.2 (±34.5) a	6	8.8 (±6.0) b		**<0.001**
Distichlis stricta	0.49 (±0.40)		1.66 (±0.71)	15	0.568	0 (±0)		4.7 (±4.7)		
Elymus lanceolatus	3.40 (±1.81)	8	3.42 (±1.85)	8	0.890	2.2 (±2.2)		4.4 (±4.4)		
Elymus trachycaulus sbsp. *Trachycaulus*	-		-			0 (±0) b		46.5 (±30.6) a	8	**<0.001**
Elytrigia repens	0.04 (±0.03)		1.73 (±0.59)	14	1.000	-		-		
Hesperostipa comata	8.27 (±1.36)	3	7.81 (±1.97)	3	0.811	2.3 (±1.6)		4.7 (±4.7)		
Heterotheca villosa	0.74 (±0.21)	13	0.59 (±0.25)		0.990	1.2 (±1.2)		0 (±0)		
Hordeum jubatum	0.33 (±0.33)		0.57 (±0.54)			182.0 (±180.8) a	3	123.1 (±109.3) b	2	**<0.001**
Juncus balticus	0.41 (±0.17)		1.17 (±0.63)			33.9 (±10.9) b	11	44.7 (±29.1) a	10	**<0.001**
Koeleria macrantha	5.24 (±1.25)	4	1.09 (±0.41)		0.983	74.3 (±39.3) a	7	14.0 (±10.2) b		**<0.001**
Lepidium densiflorum	0.004 (±0.003)		0.02 (±0.01)			44.6 (±17.3) b	8	59.4 (±25.9) a	7	**<0.001**
Melilotus alba	0.02 (±0.02) b		4.91 (±2.03) a	5	**<0.001**	0 (±0) b		97.4 (±53.7) a	3	**<0.001**
Melilotus officinalis	0.08 (±0.04) b		6.80 (±2.29) a	4	**<0.001**	1.1 (±1.1) b		82.5 (±42.7) a	4	**<0.001**
Nassela viridula	0.06 (±0.04) b		2.31 (±1.36) a	12	**<0.001**	0 (±0)		4.7 (±4.7)		
Pascopyrum smithii	4.36 (±2.02)	5	2.79 (±0.80)	11	0.211	3.4 (±2.4)		0 (±0)		
Poa pratensis	2.39 (±1.06) b	9	8.39 (±2.43) a	2	**0.007**	41.2 (±19.7) b	9	63.2 (±30.6) a	6	**<0.001**
Poa secunda	0.67 (±0.16)	14	0.09 (±0.07)		1.000	14.8 (±6.2) a	14	0 (±0) b		**<0.001**
Ratibida columnifera	0.07 (±0.05)		0.07 (±0.05)			2.3 (±1.6) b		42.1 (±42.1) a	11	**<0.001**
Rumex crispus	0.04 (±0.04)		0 (±0)			81.1 (±81.1)	5	80.7 (±58.2)	5	0.884
Sonchus arvensis	0.82 (±0.79)	12	2.06 (±1.82)	13	0.239	11.0 (±5.8)		13.7 (±7.5)		
Taraxacum officinale	0.92 (±0.41)	11	1.45 (±0.65)		0.802	7.9 (±3.7)		0 (±0)		
Thermopsis.rhombifolia	1.88 (±1.09)	10	1.13 (±0.79)		1.000	-		-		
Tragopogon dubius	0.62 (±0.12)	15	0.86 (±0.29)		1.000	9.2 (±3.0) b		32.7 (±18.2) a	13	**<0.001**
Typha latifolia	-		-			9.1 (±3.5) b		18.4 (±14.5) a	15	**<0.001**

R = relative rank

Table 6.2. Plant community and seed bank composition responses to various characteristics associated with pipelines installed within Mixedgrass prairie, as determined through perMANOVA (distance = Bray-Curtis, permutations = 999). Distance from pipeline, age of disturbance, and diameter of pipeline were blocked by site in the analysis. Bolded tests indicate those with $P < 0.05$.

	Plant Community				**Seed Bank**			
Factor	**Mean Square**	**F Model**	**R^2**	**P Value**	**Mean Square**	**F Model**	**R^2**	**P Value**
Age	1.23	7.52	0.03	**0.001**	1.53	5.12	0.02	**0.001**
Distance	1.42	8.68	0.03	**0.001**	0.75	2.53	0.01	**0.001**
Diameter	1.38	8.47	0.02	**0.001**	1.03	3.46	0.01	**0.001**
Interactions								
Age * Distance	0.23	1.41	0.00	**0.011**	0.46	1.53	0.01	**0.015**
Age * Diameter	2.28	13.96	0.05	**0.001**	1.73	5.79	0.02	**0.001**
Distance * Diameter	0.65	3.98	0.01	**0.001**	0.94	3.14	0.01	**0.001**
Age * Distance * Diameter	0.38	2.31	0.01	**0.001**	0.22	0.73	0.00	0.389

Table 6.3. Pairwise comparisons of either plant community or seed bank composition at different sampling distances away from the pipeline using perMANOVA (distance = Bray-Curtis, permutations = 999) blocked by 18 study sites. Contrasts were focused on the center of the pipeline and furthest sampling distance (55 m), to determine the extent of variance either away from, or towards, the pipeline trench, respectively.

	Plant Community				Seed Bank			
Distances Compared	**Mean Square**	**F Model**	**R^2**	**P Value**	**Mean Square**	**F Model**	**R^2**	**P Value**
Center vs. Edge	-	-	-	-	0.11	0.33	0.01	0.780
Center vs. 50 cm	0.10	0.38	0.01	**0.009**	0.25	0.73	0.02	0.376
Center vs. 1 m	0.28	1.01	0.03	**0.001**	0.27	0.77	0.02	0.332
Center vs. 2 m	0.45	1.84	0.05	**0.001**	0.37	1.17	0.03	0.141
Center vs. 3 m	0.66	2.89	0.08	**0.001**	0.33	1.01	0.03	0.286
Center vs. 5 m	0.81	3.76	0.10	**0.001**	0.26	0.78	0.02	0.396
Center vs. 7.5 m	0.86	3.90	0.10	**0.001**	0.26	0.78	0.02	0.431
Center vs. 10 m	0.85	3.92	0.10	**0.001**	0.37	1.13	0.03	0.067
Center vs. 15 m	1.08	5.12	0.13	**0.001**	0.45	1.38	0.04	**0.037**
Center vs. 20 m	1.18	5.71	0.14	**0.001**	0.50	1.56	0.04	**0.002**
Center vs. 25 m	1.22	1.22	5.86	**0.001**	0.64	1.98	0.05	**0.006**
Center vs. 35 m	1.18	5.63	0.14	**0.001**	0.43	1.33	0.04	**0.042**
Center vs. 45 m	1.05	4.97	0.13	**0.001**	0.47	1.40	0.04	0.067
Center vs. 55 m	1.07	4.88	0.13	**0.001**	0.47	1.42	0.04	0.073
Edge vs. 55 m	-	-	-	-	0.39	1.21	0.03	0.223
50 cm vs. 55 m	0.60	2.81	0.08	**0.001**	0.28	0.84	0.02	0.300
1 m vs. 55 m	0.33	1.69	0.05	**0.003**	0.25	0.75	0.02	0.487
2 m vs. 55m	0.21	1.10	0.03	**0.016**	0.19	0.59	0.02	0.676
3 m vs. 55 m	0.11	0.60	0.02	0.221	0.28	0.90	0.03	0.290
5 m vs. 55 m	0.12	0.68	0.02	0.190	0.27	0.85	0.02	0.287
7.5 m vs.55 m	0.05	0.30	0.01	0.674	0.17	0.53	0.02	0.809
10 m vs. 55 m	0.04	0.26	0.01	0.699	0.18	0.58	0.02	0.580
15 m vs. 55 m	0.07	0.43	0.01	0.205	0.08	0.25	0.01	0.965
20 m vs. 55 m	0.06	0.40	0.01	0.238	0.20	0.63	0.02	0.358
25 m vs. 55 m	0.05	0.35	0.01	0.409	0.22	0.69	0.02	0.384
35 m vs. 55 m	0.05	0.30	0.01	0.219	0.09	0.29	0.01	0.873
45 m vs. 55 m	0.07	0.41	0.01	0.152	0.10	0.30	0.01	0.881
Work Area[1]								
Trench vs. ROW	1.11	5.70	0.03	**0.001**	0.660	2.093	0.011	**0.003**
Trench vs. Undisturbed	1.79	10.24	0.10	**0.001**	1.001	3.186	0.029	**0.001**
ROW vs. Undisturbed	0.51	2.87	0.01	**0.001**	0.36	1.17	0.01	**0.044**

[1] Trench = Pipeline centre and edge; ROW = Right of Way = 0.5 m - 20 m, inclusive; Undisturbed = 25 m - 55 m, inclusive.

Table 6.4. Aboveground plant community and seed bank compositional responses to soil properties in Mixedgrass prairie, as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m) for this analysis.

	Plant Community				Seed Bank			
Soil Property	**Mean Square**	**F Model**	**R^2**	**P Value**	**Mean Square**	**F Model**	**R^2**	**P Value**
C	2.85	21.83	0.16	**0.001**	1.32	5.18	0.05	**0.001**
N	0.21	1.62	0.01	0.114	0.51	2.02	0.02	**0.011**
C:N Ratio	0.38	2.93	0.02	**0.015**	0.49	1.91	0.02	**0.027**
OM	0.17	1.27	0.01	0.236	0.65	2.57	0.02	**0.005**
EC	1.59	12.23	0.09	**0.001**	2.48	9.72	0.09	**0.001**
pH	0.90	6.91	0.05	**0.001**	0.80	3.12	0.03	**0.001**
Significant Interactions								
C * N	0.38	2.91	0.02	**0.008**	-	-	-	-
C * EC	0.30	2.28	0.02	**0.036**	-	-	-	-
C * pH	-	-	-	-	0.51	2.01	0.02	**0.009**
C:N Ratio * pH	-	-	-	-	0.50	1.97	0.01	**0.018**
N * C * OM	0.39	2.98	0.02	**0.011**	-	-	-	-
N * C:N Ratio * OM	-	-	-	-	0.53	2.07	0.02	**0.012**
N * C:N Ratio * pH	-	-	-	-	0.46	1.82	0.02	**0.031**
C:N Ratio * OM * pH	0.32	2.49	0.02	**0.021**	-	-	-	-
C * N * C:N Ratio * OM * pH * EC	-	-	-	-	0.48	1.87	0.02	**0.026**

Table 6.5. Aboveground plant community and seed bank compositional responses to ground cover and litter mass in Mixedgrass prairie, as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m).

	Plant Community				Seed Bank			
Ground Cover	**Mean Square**	**F Model**	**R^2**	**P Value**	**Mean Square**	**F Model**	**R^2**	**P Value**
Bare Ground	0.65	4.32	0.04	**0.002**	0.57	1.98	0.02	**0.018**
Litter Cov.	1.84	12.26	0.10	**0.001**	0.77	2.66	0.03	**0.003**
Litter Mass	1.21	8.07	0.07	**0.001**	1.01	3.49	0.03	**0.001**
Soil Crust	0.20	1.36	0.01	0.179	0.57	1.96	0.02	**0.020**
Stems	0.49	3.24	0.03	**0.004**	0.27	0.92	0.01	0.549
Significant Interactions								
Bare * Stems	-	-	-	-	0.49	1.68	0.02	**0.040**
Litter Mass * Soil Crust	-	-	-	-	0.52	1.80	0.02	**0.035**
Litter Cov. * Soil Crust	0.31	2.06	0.02	**0.050**	0.53	1.83	0.02	**0.030**
Litter Cov. * Stems	0.43	2.84	0.02	**0.009**	-	-	-	-
Soil Crust *Stems	-	-	-	-	0.55	1.91	0.02	**0.020**
Bare * Litter Cov. * Stems	-	-	-	-	0.53	1.81	0.02	**0.031**
Litter Cov. * Litter Mass * Stems	0.30	1.99	0.02	**0.050**	0.57	1.98	0.02	**0.016**

Table 6.6. Results of the indicator species analysis of aboveground vegetation cover assessed in relation to varying distances from pipeline disturbance.

Distances	Species	A	B	P Value
Center, 50 cm, 1 m	*Elytrigia repens*	0.69	0.39	**0.003**
Center, 50 cm, 1 m, 2 m	*Nassella viridula*	0.96	0.18	**0.007**
Center, 50 cm, 1 m, 2 m, 3 m	*Melilotus alba*	0.87	0.32	**0.001**
Center, 50 cm, 1 m, 2 m, 3 m, 5 m	*Melilotus officinalis*	0.94	0.51	**0.001**
Center, 50 cm, 1 m, 2 m, 3 m, 5 m, 7.5 m, 10 m	*Agropyron cristatum*	0.97	0.16	0.082
Center, 2 m, 5 m	*Festuca ovina*	1.00	0.09	0.097
Center, 2 m, 5 m, 15 m, 35 m	*Chenopodium album*	0.79	0.18	0.053
Center to 45 m	*Tragopogon dubius*	0.97	0.70	**0.026**
50 cm, 1m, 2 m, 3 m	*Elymus trachycaulus* ssp. *trachycaulus*	0.95	0.15	**0.024**
50 cm to 55 m	*Bouteloua gracilis*	0.98	0.94	**0.001**
1 m to 55 m	*Poa secunda*	0.96	0.54	**0.001**
2 m to 55 m	*Selaginella densa*	0.99	0.40	**0.003**
3 m	*Arnica*	0.93	0.11	**0.046**
5 m, 15 m, 25 m	*Orobanche fasciculate*	0.67	0.17	0.083
5 m, 7.5, 10 m, 15 m, 20, 25 m, 45 m	*Oenethera nuttallii*	0.93	0.15	0.057
Work Area				
Trench	*Astragalus cicer*	0.68	0.28	**0.003**
	Atriplex subspicata	0.78	0.11	**0.019**
	Bromus inermis ssp. *Inermis*	0.83	0.11	**0.029**
	Chenopodium album	0.67	0.28	**0.008**
	Cirsium flodmanii	0.71	0.22	**0.016**
	Cleome serrulata	1.00	0.06	0.075
	Conyza canadensis	0.74	0.11	**0.045**
	Eleocharis palustris	0.91	0.11	**0.006**
	Elymus junceus	0.88	0.06	0.089
	Elytrigia repens	0.84	0.44	**0.001**
	Festuca ovina	0.96	0.11	**0.008**
	Glycyrrhiza lepidota	0.57	0.44	**0.005**
	Kochia scoparia	1.00	0.06	0.060
	Lappula squarrosa	0.90	0.11	**0.015**
	Melilotus alba	0.73	0.39	**0.001**
	Melilotus officinalis	0.76	0.67	**0.001**
	Nassella viridula	0.82	0.28	**0.004**
	Salsola pestifer	0.77	0.11	**0.025**
	Schedonnardus paniculatus	0.81	0.11	**0.034**
	Shepherdia argentia	1.00	0.06	0.080
	Silene drumondii	0.80	0.06	0.073
	Sporololus cryptandrus	0.96	0.06	0.063
	Symphoricarpos occidentalis	0.55	0.17	0.074
ROW + Trench	*Agropyron cristatum*	1.00	0.14	**0.035**
ROW	*Elymus trachycaulus* ssp. *trachycaulus*	1.00	0.09	0.074
ROW + Undisturbed	*Selaginella densa*	0.99	0.36	**0.011**
Undisturbed + Trench	*Crepis tectorum*	0.86	0.16	0.083

[1] Trench = Pipeline centre and edge; ROW = Right of Way = 0.5 m - 20 m, inclusive; Undisturbed = 25 m - 55 m, inclusive.
A = Probability of occurring, B = Fidelity

Table 6.7. Results of the indicator species analysis relating aboveground plant community composition with varying pipeline diameters.

Diameter (mm)[1]	Species	A	B	P Value
60.3	*Androsace septentrionalis*	0.82	0.19	**0.006**
	Artemisia frigida	0.40	0.99	**0.015**
	Bouteloua gracilis	0.44	0.96	**0.001**
	Carex duriuscula	0.41	0.99	**0.003**
	Elymus trachycaulus ssp. *trachycaulus*	0.97	0.10	**0.026**
	Gaura coccinea	0.63	0.49	**0.001**
	Koeleria macrantha	0.46	0.84	**0.009**
	Nassella viridula	0.97	0.12	**0.011**
	Pascopyrum smithii	0.53	0.59	0.082
	Phlox hoodia	0.77	0.21	**0.003**
	Poa secunda	0.54	0.61	**0.004**
	Potentilla pensylvanica	0.70	0.12	**0.042**
	Selaginella densa	0.71	0.46	**0.001**
	Solidago missouriensis	0.76	0.18	**0.031**
	Symphyotrichum ericoides	0.58	0.34	**0.049**
88.9	*Agrostis scabra*	0.56	0.20	0.079
	Artemisia ludoviciana	0.61	0.33	**0.026**
	Astraglus pectinatus	1.00	0.04	0.085
	Carex pensylvanica	0.78	0.34	**0.001**
	Cirsium arvense	0.98	0.06	**0.033**
	Distichlis stricta	0.59	0.27	**0.017**
	Elymus lanceolatus	0.67	0.80	**0.001**
	Escobaria viviparia	0.63	0.13	0.072
	Glycyrrhiza lepidota	0.69	0.26	**0.007**
	Hordeum jubatum	0.99	0.24	**0.001**
	Juncus balticus	0.56	0.36	**0.049**
	Poa palustris	0.99	0.20	**0.001**
	Psoralea lanceolate	0.97	0.16	**0.001**
	Rumex crispus	1.00	0.03	0.095
	Salsola pestifer	0.77	0.09	**0.019**
	Schedonnardus paniculatus	0.86	0.10	**0.008**
	Sonchus arvensis	0.99	0.29	**0.001**
	Taraxacum officinale	0.57	0.70	**0.001**
	Thinopyrum intermedium	1.00	0.07	**0.005**
> 168.3	*Agropyrum cristatum*	0.95	0.45	**0.001**
	Bromus inermis ssp. *inermis*	0.98	0.14	**0.001**
	Chenopodium album	0.82	0.26	**0.001**
	Cirsium flodmanii	0.61	0.17	**0.046**
	Elytrigia repens	0.54	0.36	**0.007**
	Festuca ovina	0.79	0.05	0.088
	Heterotheca villosa	0.64	0.64	**0.001**
	Lithospermum insisum	0.53	0.81	**0.002**
	Opuntia polycantha	0.91	0.05	0.084
	Poa pratensis	0.51	0.62	**0.008**
	Sporobolus cryptandrus	0.97	0.05	**0.017**
	Symphoricarpos occidentalis	0.96	0.21	**0.001**
	Thermopsis rhombifolia	0.77	0.60	**0.001**

A = Probability of occurring, B = Fidelity

[1] >168.3 mm includes diameters up to 1067 mm

Table 6.8. Results of the indicator species analysis of aboveground plant community composition with varying age classes of pipelines.

Age Class	Species	A	B	P Value
0 to 10 yrs	*Bouteloua gracilis*	0.30	0.93	**0.014**
	Crepis tectorum	0.52	0.25	0.069
	Elymus trachycaulus ssp. *trachycaulus*	0.85	0.29	**0.002**
	Gaura coccinea	0.34	0.64	**0.029**
	Glycyrrhiza lepidota	0.53	0.36	**0.022**
	Nassella viridula	0.76	0.25	**0.011**
	Pascopyrum smithii	0.53	0.61	**0.023**
	Sisyrinchium montanum	0.72	0.11	0.065
	Taraxacum officinale	0.73	0.54	**0.001**
	Tragopogon dubius	0.46	0.96	**0.001**
11 to 20 yrs	*Artemisia campestris*	1.00	0.13	**0.031**
	Calamovilfa longifolia	0.28	0.88	0.059
	Campanula rotundifolia	1.00	0.16	**0.011**
	Carex duriuscula	0.35	1.00	**0.001**
	Carex pensylvanica	0.62	0.26	0.094
	Hesperostipa comate	0.27	0.96	0.086
	Juncus balticus	0.51	0.50	**0.015**
	Liatris punctate	0.54	0.26	0.094
	Poa secunda	0.36	0.63	0.053
	Psoralea lanceolate	1.00	0.11	**0.039**
	Vicia Americana	0.43	0.65	**0.009**
21 to 30 yrs	-			
31 to 40 yrs	*Elymus lanceolatus*	0.52	0.81	**0.003**
	Gaillardia aristata	1.00	0.06	0.061
	Grindella squarrosa	0.93	0.10	0.069
	Hordeum jubatum	1.00	0.21	**0.003**
	Poa palustris	0.99	0.17	**0.013**
	Ratibida columnifera	0.58	0.19	0.081
	Salsola pestifer	1.00	0.08	0.088
	Sonchus arvensis	0.92	0.17	**0.033**
41 to 50 yrs	*Artemisia frigida*	0.41	1.00	**0.001**
	Chenopodium album	0.71	0.50	**0.001**
	Cirsium flodmanii	0.73	0.36	**0.001**
	Dalea purpurea	0.58	0.29	**0.012**
	Distichlis stricta	0.56	0.57	**0.001**
	Erysimum inconspicuum	0.77	0.14	0.069
	Haplopappus spinulosus	0.40	0.50	**0.010**
	Lygodesmia juncea	0.58	0.71	**0.001**
	Melilotus alba	0.64	0.50	**0.001**
	Melilotus officinalis	0.48	0.57	**0.012**
51 to 60 yrs	*Agropyron cristatum*	0.84	0.57	**0.001**
	Astragalus agrestis	0.42	0.36	0.082
	Astragalus cicer	0.45	0.21	0.076
	Bromus inermis ssp. *inermis*	0.63	0.36	0.082
	Chenopodium pratericola	0.76	0.43	**0.002**
	Elytrigia repens	0.37	0.79	**0.001**
	Equisetum laevigatum	0.31	0.79	0.065
	Festuca ovina	0.77	0.14	**0.036**
	Koeleria macrantha	0.32	1.00	**0.012**
	Poa pratensis	0.64	1.00	**0.001**
	Rosa arkansana	0.34	0.71	**0.020**
	Symphoricarpos occidentalis	0.97	0.64	**0.001**
	Thermopsis rhombifolia	0.84	1.00	**0.001**

A = Probability of occurring, B = Fidelity

Table 6.9. Results of the indicator species analysis of aboveground plant community composition and various interactions of pipeline distance, age, and diameter.

Distance * Decade	**Species**	**A**	**B**	**P Value**
Trench = 0 to 10 yrs	*Conyza canadensis*	0.68	0.50	0.089
	Nassella viridula	0.63	1.00	**0.046**
	Shepherdia argentia	1.00	0.50	**0.020**
Trench = 31 to 40 yrs	*Kochia scoparia*	1.00	0.17	0.075
Trench = 41 to 50 yrs	*Cirsium flodmanii*	0.52	1.00	0.063
	Dalea purpurea	0.59	1.00	**0.030**
	Distichlis stricta	0.41	1.00	0.088
	Oxytropis sericea	0.89	1.00	**0.010**
	Polygonum aviculare	0.93	1.00	**0.009**
Trench = 51 to 60 yrs	*Agropyron cristatum*	0.52	1.00	0.079
	Bromus inermis ssp. *inermis*	0.72	1.00	**0.035**
	Descurainia sophia	0.70	1.00	**0.030**
	Festuca ovina	0.69	1.00	**0.027**
	Lappula squarosa	0.91	1.00	**0.013**
	Spaeralcea coccinea	0.61	1.00	**0.003**
	Symphoricarpos occidentalis	0.50	1.00	0.070
ROW = 0 to 10 yrs	*Elymus trachycaulus* ssp. *trachycaulus*	0.85	0.44	0.071
ROW = 41 to 50 yrs	*Artemisia frigida*	0.18	1.00	0.071
Undisturbed = 41 to 50 yrs	*Bouteloua gracilis*	0.18	1.00	**0.011**
	Calamovilfa longifolia	0.14	1.00	0.092
Undisturbed = 51 to 60 yrs	*Koeleria macrantha*	0.24	1.00	0.053
	Opuntia fragilis	1.00	0.25	**0.035**
	Thermopsis rhombifolia	0.42	1.00	0.055
Distance * Diameter (mm)				
Trench = 60.3	*Atriplex subspicata*	0.70	0.20	0.100
	Cleome serrulata	1.00	0.10	0.087
	Nassella viridula	0.78	0.40	**0.015**
	Shepherdia argentia	1.00	0.10	0.070
Trench = 88.9	*Cirsium arvense*	0.68	0.20	0.099
	Elymus junceus	0.88	0.20	**0.050**
	Kochia scoparia	1.00	0.20	**0.030**
	Lepidium densiflorum	0.55	0.40	**0.049**
Trench = > 168.3	*Bromus inermis* ssp. *inermis*	0.75	0.33	**0.027**
	Chenopodium album	0.52	0.67	**0.011**
	Festuca ovina	0.77	0.33	**0.026**
	Lappula squarrosa	0.74	0.33	**0.028**
	Oxytropis sericea	0.82	0.33	**0.021**
	Polygonum aviculare	0.80	0.33	**0.025**
	Sporobolus cryptandrus	0.98	0.33	**0.001**
Trench = 60.3 + > 168.3	*Astragalus cicer*	0.64	0.38	**0.042**
Trench = 88.9 + > 168.3	*Distichlis stricta*	0.52	0.50	0.055
	Salsola pestifer	0.81	0.25	0.057
Trench = All Diameters	*Glycyrrhiza lepidota*	0.59	0.44	0.056
	Melilotus alba	0.71	0.39	**0.035**
	Melilotus officinalis	0.76	0.67	**0.003**
Trench = 88.9 + > 168.3, ROW = > 168.3	*Symphoricarpos occidentalis*	0.98	0.31	**0.017**
Trench = All Diameters, ROW = 88.9 + > 168.3, Undisturbed = 60.3 + 88.9	*Artemisia ludoviciana*	0.95	0.31	0.093
Trench + ROW = >168.3	*Agropyron cristatum*	0.94	0.60	**0.001**

Trench + ROW = 60.3 + 88.9	*Crepis tectorum*	0.83	0.23	0.091
Trench + ROW = 60.3 + 88.9, Undisturbed = All Diameters	*Aster ericoides*	0.98	0.36	0.059
Trench = All Diameters, ROW = >168.3	*Elytrigia repens*	0.93	0.44	**0.006**
ROW + Undisturbed = All Diameters	*Taraxacum officinale*	0.99	0.56	**0.002**
ROW + Undisturbed = All Diameters, Trench = 60.3	*Poa secunda*	1.00	0.50	**0.023**
Undisturbed = 60.3 + 88.9, ROW = 60.3	*Selaginella densa*	0.88	0.45	**0.021**
All distances = 60.3 + >168.3, Trench = 88.9	*Lithospermum incisum*	0.91	0.72	**0.045**
All distances = 60.3, Trench = 88.9, Undisturbed = > 168.3	*Gaura coccinea*	0.87	0.48	**0.010**
All distances = 88.9 + >168.3, Trench = 60.3	*Poa pratensis*	0.92	0.62	**0.001**

[1] Trench = Pipeline centre and edge; ROW = Right of Way = 0.5 m - 20 m, inclusive; Undisturbed = 25 m - 55 m, inclusive.
A = Probability of occurring, B = Fidelity

Table 6.10. ANOVA summary statistics for plant community responses to pipeline disturbance and characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect.

	Native Cover		**Introduced Cover**		**Richness**		**Shannon's Diversity**		**Pielou's Evenness**		**Beta Diversity**	
Characteristic	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**
Age	1.43	0.254	0.32	0.584	0.68	0.42	0.591	0.458	0.08	0.779	0.08	0.776
Diameter (W)	0.41	0.674	0.77	0.483	0.02	0.984	1.15	0.345	0.19	0.823	0.18	0.836
Distance (D)	32.21	**<0.001**	30.41	**<0.001**	9.91	**<0.001**	3.92	**0.021**	3.49	**0.032**	8.68	**0.0002**
Texture (T)	0.28	0.603	0.25	0.624	0.22	0.643	0.865	0.369	0.96	0.343	0.02	0.896
W * D	5.29	**0.0004**	3.38	**0.010**	2.02	0.093	2.49	**0.044**	1.23	0.298	2.24	0.07
W * T	0.85	0.443	0.02	0.897	0.15	0.705	1.14	0.371	1.2	0.351	1.52	0.259
D * T	0.13	0.724	1.29	0.277	0.79	0.454	1.67	0.189	2.39	0.094	0.7	0.498
W * D * T	0.72	0.49	0.22	0.802	0.82	0.444	1.61	0.175	1.99	0.088	-	-

Transformations: square root (introduced cover), log (beta diversity)

Table 6.11. LS means (±SE) for plant community responses to pipeline disturbance and characteristics.

Characteristic	Levels	Native (%)	Introduced (%)	Richness	Shannon's Diversity	Pielou's Evenness	Beta Diversity
Distance	Trench	57.5 (±5.0) c	28.1 (±5.5) a	21.8 (±1.6) a	2.24 (±0.11) a	0.73 (±0.03) a	2.03 (±0.17) c
	ROW	77.7 (±3.9) b	16.1 (±4.5) b	18.6 (±1.2) b	2.05 (±0.08) b	0.71 (±0.02) a	2.40 (±0.10) b
	Undisturbed	86.6 (±4.0) a	5.6 (±4.6) c	17.1 (±1.3) c	1.85 (±0.09) c	0.66 (±0.02) b	2.63 (±0.12) a
Diameter * Distance	60.3 mm * Trench	55.7 (±5.1) bc	23.4 (±5.6) ab		2.12 (±0.11) ab		
	60.3 mm * ROW	79.9 (±4.0) ab	7.6 (±4.6) ab		2.00 (±0.09) ab		
	60.3 mm * Undisturbed	82.0 (±4.2) ab	2.9 (±4.8) b		1.92 (±0.09) ab		
	88.9 mm * Trench	67.0 (±7.0) bc	24.1 (±7.7) ab		2.26 (±0.15) a		
	88.9 mm * ROW	81.1 (±5.3) ab	14.6 (±6.1) ab		2.04 (±0.11) ab		
	88.9 mm * Undisturbed	85.3 (±5.6) ab	8.2 (±6.4) ab		1.86 (±0.12) b		
	≥168.3 * Trench	49.8 (±10.6) c	36.6 (±11.8) a		2.34 (±0.23) a		
	≥168.3 * ROW	72.1 (±8.6) b	26.2 (±10.0) ab		2.11 (±0.19) a		
	≥168.3 * Undisturbed	92.4 (±9.0) a	5.6 (±10.3) b		1.79 (±0.19) b		

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 6.12. ANOVA summary statistics for plant community dry biomass responses to pipeline disturbance characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect.

	Native Biomass		**Native Graminoids**		**Native Forbs**		**Introduced Biomass**		**Introduced Graminoids**		**Introduced Forbs**		**Total Biomass**		**Litter Biomass**	
Characteristic	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**
Age	0.22	0.649	0.002	0.966	0.12	0.738	0.13	0.72	0.06	0.81	0.01	0.938	0.44	0.521	0.07	0.802
Diameter (W)	0.03	0.967	0.32	0.735	0.4	0.681	0.2	0.824	0.48	0.63	0.08	0.922	0.02	0.976	1.67	0.229
Distance (D)	1.42	0.239	0.40	0.81	0.94	0.449	4.36	**0.004**	7.22	**0.0001**	1.27	0.293	4.94	**0.002**	2.6	**0.046**
Texture (T)	3.38	0.091	0.004	0.947	0.21	0.652	1.41	0.258	0.59	0.456	0.07	0.799	8.67	**0.012**	16.68	**0.002**
W * D	0.153	0.996	1.14	0.352	0.81	0.6	1.25	0.29	1.14	0.349	0.79	0.611	0.18	0.992	1.25	0.291
W * T	0.34	0.569	0.02	0.904	0.66	0.433	0.06	0.81	0.01	0.936	0.03	0.875	0.45	0.513	1.07	0.321
D * T	0.55	0.699	1.24	0.305	0.5	0.737	1.23	0.31	1.36	0.26	0.93	0.454	0.17	0.952	1.05	0.39
W * D * T	0.68	0.611	0.23	0.919	0.68	0.61	0.26	0.903	3.39	**0.015**	0.65	0.627	0.54	0.705	1.73	0.158

Transformations: square root (introduced, introduced grasses, total biomass), log (litter)

Table 6.13. LS means (±SE) dry biomass (kg/ha) responses to pipeline characteristics.

Characteristic	Levels	Introduced	Introduced Graminoids	Total Biomass	Litter Biomass
Distance (D)	0 m (Trench)	518.9 (±69.4) a	267.5 (±46.7) a	1709.0 (±126.7) a	1291.2 (±177.3) a
	1 m	171.2 (±69.4) b	109.8 (±46.7) b	1123.5 (±126.7) b	1002.3 (±177.3) ab
	5 m	96.7 (±69.4) b	61.5 (±46.7) b	966.5 (±126.7) b	742.0 (±177.3) b
	20 m	123.1 (±69.4) b	102.1 (±46.7) b	1111.0 (±126.7) b	918.3 (±177.3) ab
	55 m	89.8 (±69.4) b	68.8 (±46.7) b	964.5 (±126.7) b	763.7 (±177.3) b
Texture (T)	Loam			880.5 (±163.3) b	577.3 (±233.6) b
	Sandy Loam			1469.3 (±87.3) a	1309.7 (±124.8) a

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 6.14. Results of the indicator species analysis of seed bank composition in relation to varying distances from pipeline disturbance.

Distances	**Species**	**A**	**B**	**P Value**
Center, Edge, 50 cm, 2 m	*Elymus trachycaulus* ssp. *trachycaulus*	1.00	0.15	**0.010**
Center, Edge, 50 cm, 1 m, 2 m	*Melilotus alba*	0.92	0.26	**0.010**
Center, Edge, 50 cm, 1 m, 2 m, 3 m	*Melilotus officinalis*	0.98	0.25	**0.010**
Edge, 50 cm, 10 m	*Nassella viridula*	0.68	0.17	**0.049**
50 cm	*Lithospermum incisum*	1.00	0.11	0.063
50 cm, 1m, 2 m	*Schedonnardus paniculatus*	0.73	0.13	**0.040**
45 m	*Astragalus agrestis*	1.00	0.11	**0.044**
Work Area				
Trench	*Elymus trachycaulus* ssp. *trachycaulus*	0.94	0.19	**0.001**
	Elytrigia repens	0.81	0.56	**0.037**
	Melilotus alba	0.84	0.31	**0.001**
	Melilotus officinalis	0.80	0.31	**0.001**
	Nassella viridula	0.67	0.14	**0.020**
	Ratibida columnifera	0.65	0.08	0.091
	Sporobolus cryptandrus	0.65	0.08	0.068
ROW + Undisturbed	*Carex pensylvanica*	0.91	0.17	0.095
Undisturbed	*Astragalus agrestis*	1.00	0.03	0.080
	Crepis tectorum	0.68	0.26	**0.013**
	Erysimum capitatum	0.90	0.06	**0.027**
	Koeleria macrantha	0.68	0.26	0.085
	Thlaspi arvense	1.00	0.03	0.080

[1] Trench = Pipeline centre and edge; ROW = Right of Way = 0.5 m - 20 m, inclusive), Undisturbed = 25 m - 55 m, inclusive.

A = Probability of occurring, B = Fidelity

Table 6.15. Results of the indicator species analysis relating seed bank composition with varying pipeline diameters.

Diameter (mm)[1]	**Species**	**A**	**B**	**P Value**
60.3	*Arabis holboellii* ssp. *retrofracta*	0.83	0.11	**0.034**
	Amaranthus blitoidies	1.00	0.05	0.079
	Bouteloua gracilis	0.71	0.21	**0.006**
	Chenopodium album	1.00	0.05	0.093
	Euphorbia serpyllifolia	1.00	0.05	0.080
	Koeleria macrantha	0.54	0.32	**0.029**
	Lepidium densiflorum	0.55	0.37	**0.017**
	Poa secunda	0.81	0.13	**0.015**
88.9	*Conyza canadensis*	0.59	0.2	**0.015**
	Campanula rotundifolia	0.70	0.12	0.052
	Draba nemorosa	0.72	0.17	**0.007**
	Festuca ovina	0.92	0.07	**0.019**
	Hedeoma hispida	0.71	0.16	**0.008**
	Hordeum jubatum	0.98	0.21	**0.001**
	Juncus balticus	0.55	0.35	**0.024**
	Poa palustris	1.00	0.11	**0.005**
	Potentilla norvegica	0.94	0.16	**0.001**
	Rumex crispus	0.91	0.21	**0.001**
	Rumex maritimus	1.00	0.16	**0.001**
> 168.3	*Artemisia frigida*	0.42	0.96	**0.004**
	Calamovilfa longifolia	0.91	0.18	**0.001**
	Descurainia Sophia	0.78	0.24	**0.001**
	Distichlis stricta	1.00	0.07	**0.006**
	Heterotheca villosa	1.00	0.04	**0.026**
	Melilotus alba	0.72	0.2	**0.008**
	Melilotus officinalis	0.61	0.16	0.086
	Ratibida columnifera	0.75	0.09	**0.041**
	Sisymbrium altissimum	1.00	0.04	**0.034**
	Sporobolus cryptandrus	0.93	0.13	**0.001**

A = Probability of occurring, B = Fidelity

[1] >168.3 mm includes diameters up to 1067 mm

Table 6.16. Results of the indicator species analysis of seed bank composition with varying age classes of pipeline.

Age Class	Species	A	B	P Value
0 to 10 yrs	*Bouteloua gracilis*	0.48	0.23	0.097
	Crepis tectorum	0.63	0.43	**0.004**
	Elymus trachycaulus ssp. t*rachycaulus*	0.77	0.17	**0.015**
	Taraxacum officinale	0.57	0.30	**0.016**
	Tragopogon dubius	0.49	0.30	**0.025**
11 to 20 yrs	*Arabis holboellii* ssp. *retrofracta*	0.88	0.14	**0.035**
	Draba nemorosa	0.80	0.18	**0.029**
21 to 30 yrs	-			
31 to 40 yrs	*Agrostis scabra*	0.48	0.32	0.090
	Calamovilfa longifolia	0.78	0.12	0.068
	Hordeum jubatum	0.94	0.21	**0.006**
	Koeleria macrantha	0.70	0.39	**0.006**
	Poa palustris	1.00	0.09	0.055
	Potentilla norvegica	0.98	0.14	**0.009**
	Ratibida columnifera	0.80	0.11	0.083
	Rumex crispus	0.94	0.24	**0.002**
	Rumex maritimus	1.00	0.13	**0.015**
41 to 50 yrs	*Melilotus alba*	0.72	0.53	**0.001**
	Melilotus officinalis	0.73	0.47	**0.001**
	Typha latifolia	0.59	0.33	**0.006**
51 to 60 yrs	-			

A = Probability of occurring, B = Fidelity

Table 6.17. Results of the indicator species analysis of seed bank composition and various interactions of pipeline distance, age, and diameter.

Distance * Decade	**Species**	**A**	**B**	**P Value**
Trench = 0 to 10 yrs	*Elymus trachycaulus* ssp. *trachycaulus*	0.74	0.75	**0.002**
	Lepidium densiflorum	0.26	0.75	0.088
	Tragopogn dubius	0.44	0.50	**0.047**
Trench = 41 to 50 yrs	*Melilotus alba*	0.55	1.00	**0.009**
	Melilotus officinalis	0.48	1.00	**0.008**
Trench = 51 to 60 yrs	*Sporobolus cryptandrus*	0.85	1.00	**0.001**
	Descurainia sophia	0.46	0.50	0.078
Undisturbed = 0 to 10 yrs	*Calamagrostis montanensis*	1.00	0.13	0.088
	Cirsium flodmanii	1.00	0.13	0.086
	Crepis tectorum	0.49	0.50	0.060
Distance * Diameter (mm)				
Trench = 88.9	*Artemisia ludoviciana*	1.00	0.10	0.063
	Kochia scoparia	1.00	0.10	0.063
	Medicago lupulina	1.00	0.10	0.060
	Oxytropis sericea	1.00	0.10	**0.047**
	Poa palustris	0.59	0.20	0.055
	Potentilla gracilis	1.00	0.10	0.063
Trench = > 168.3	*Distichlis stricta*	0.62	0.17	0.059
	Erysimum inconspicuum	0.66	0.33	**0.002**
	Melilotus alba	0.61	0.50	**0.003**
	Ratibida columnifera	0.83	0.33	**0.001**
	Sporobolus cryptandrus	0.69	0.50	**0.001**
Trench = 60.3 + > 168.3	*Elymus trachycaulus* ssp. *trachycaulus*	0.95	0.27	**0.004**
Trench = All Diameters	*Melilotus officinalis*	0.77	0.31	**0.008**
Trench = All Diameters, ROW = 60.3 + > 168.3, Undisturbed = 60.3 + > 168.3	*Lepidium densiflorum*	0.95	0.35	**0.011**
Trench = 60.3 + > 168.3, ROW = > 168.3	*Nassella viridula*	0.80	0.15	0.084
Trench + ROW = 88.9	*Hedeoma hispida*	0.72	0.22	**0.045**
ROW = > 168.3	*Calamovilfa longifolia*	0.77	0.26	**0.016**
All distances = 60.3, Undisturbed + ROW = 88.9	*Koeleria macrantha*	0.98	0.30	**0.011**
All distances = 60.3, Trench = 88.9, Undisturbed = > 168.3	*Bouteloua gracilis*	0.93	0.19	0.070
Undisturbed = 88.9 mm	*Erysimum capitatum*	0.81	0.15	**0.023**
All Distances + Diameters Except Trench = > 168.3	*Androsace septentrionalis*	1.00	0.53	**0.038**

[1] Trench = Pipeline centre and edge; ROW = Right of Way = 0.5 m - 20 m, inclusive; Undisturbed = 25 m - 55 m, inclusive.

Table 6.18. LS means (±SE) for seed density (seeds/m^2) of plant groups of major life histories and origins. Significant differences in seed density among pipeline diameter classes were determined using ANOVA and Kruskal-Wallis tests. Lower case letters denote significant differences ($P < 0.05$).

	Distance		
Life History	**60.3 mm**	**88.9 mm**	**≥168.3**
Introduced Annual Forbs (X^2=0.319, P=0.853)	131.7 (±49.1)	73.5 (±69.5)	94.1 (±89.7)
Introduced Biennial Forbs (X^2=4.502, P=0.105)	33.4 (±33.9)	38.9 (±48.0)	147.8 (±62.0)
Introduced Perennial Forbs (X^2=7.303, **P=0.026**)	23.1 (±124.7) b	356.2 (±176.4) a	38.7 (±227.7) ab
Introduced Grasses (X^2=2.067, P=0.356)	47.4 (±22.1)	71.9 (±31.2)	40.6 (±40.3)
Native Annual Forbs (X^2=0.682, P=0.711)	10.1 (±1.3)	11.0 (±1.9)	11.5 (±2.4)
Native Biennial Forbs (X^2=9.159, **P=0.010**)	102.2 (±21.6) a	47.2 (30.6) b	66.5 (±39.5) ab
Native Perennial Forbs (F=0.626, P=0.548)	131.6 (±49.1)	73.5 (±69.5)	94.1 (±89.7)
Native Grasses (X^2=1.069, P=0.586)	186.7 (±222.5)	734.0 (±314.6)	167.7 (±406.1)
Native Graminoids (F=0.321, P=0.731)	10.4 (±1.4)	10.3 (±2.0)	8.1 (±2.5)

Table 6.19. ANOVA summary statistics for seed density (seeds/m^2) responses to pipeline disturbance characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect.

Characteristic	**Native**		**Native Graminoids**		**Native Forbs**		**Introduced**		**Introduced Graminoids**		**Introduced Forbs**		**Total**	
	F Value	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**
Age	0.06	0.808	2.17	0.166	0.6	0.453	0.37	0.555	0.002	0.963	0.37	0.555	0.83	0.379
Diameter (W)	0.46	0.64	0.73	0.502	1.53	0.253	0.35	0.71	0.28	0.761	0.46	0.642	0.21	0.809
Distance (D)	1.77	0.172	2.16	0.118	1.11	0.33	0.41	0.666	0.05	0.948	2.58	0.078	2.01	0.136
Texture (T)	3.71	0.075	0.004	0.951	8.98	**0.009**	0.0003	0.986	1.34	0.268	0.02	0.891	1.49	0.243
W * D	0.08	0.988	2.15	0.075	1.97	0.1	2.02	0.092	1.37	0.245	2.54	**0.041**	1.2	0.311
W * T	0.43	0.523	0.25	0.636	1.71	0.212	0.34	0.572	0.01	0.907	0.34	0.572	0.82	0.382
D * T	0.76	0.471	4.3	**0.015**	0.18	0.836	1.72	0.181	2.33	0.099	0.29	0.746	0.83	0.439
W * D * T	0.01	0.99	0.07	0.933	0.88	0.417	0.25	0.78	0.96	0.383	0.02	0.981	0.28	0.755

Transformations: square root (native, native graminoids), log (native forbs, introduced, introduced graminoids, introduced forbs, total density)

Table 6.20. LS mean (±SE) for seed density (seeds/m^2) and diversity responses to pipeline disturbance and characteristics.

Characteristic	Levels	Native Graminoids	Native Forbs	Introduced Forbs	Beta Diversity
Distance	Trench				5.06 (±0.52) ab
	ROW				5.02 (±0.25) b
	Undisturbed				5.53 (±0.37) a
Texture	Loam		455.3 (±132.3) b		
	Sandy Loam		788.1 (±70.7) a		
Diameter * Distance	60.3 mm * Trench			169.5 (±169.0) ab	
	60.3 mm * ROW			186.2 (±148.3) ab	
	60.3 mm * Undisturbed			202.0 (±156.0) ab	
	88.9 mm * Trench			484.2 (±239.0) ab	
	88.9 mm * ROW			461.9 (±209.7) ab	
	88.9 mm * Undisturbed			475.7 (±220.7) ab	
	≥168.3 * Trench			417.5 (±308.6) a	
	≥168.3 * ROW			145.4 (±270.8) b	
	≥168.3 * Undisturbed			516.6 (±284.9) a	
Distance * Texture	Trench * Loam	155.3 (±377.8) b			
	ROW * Loam	336.4 (±366.6) ab			
	Undisturbed * Loam	430.6 (±370.6) a			
	Trench * Sand	287.8 (±201.9) ab			
	ROW * Sand	356.6 (±195.9) ab			
	Undisturbed * Sand	315.5 (±198.1) ab			

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 6.21. ANOVA summary statistics of diversity indices describing seed bank responses to pipeline disturbance and characteristics. Distance, pipeline diameter, and texture were analyzed as categorical fixed effects and age as a continuous fixed effect.

	Richness		**Shannon's Diversity**		**Pielou's Evenness**		**Beta Diversity**		**Sorenson's Similarity**	
Characteristic	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**	**F Value**	**P value**
Age	0.01	0.931	0.69	0.423	1.68	0.219	1.30	0.277	0.61	0.450
Diameter (W)	0.11	0.897	0.07	0.933	0.04	0.959	1.50	0.252	1.48	0.254
Distance (D)	2.75	0.065	1.88	0.155	1.56	0.212	3.32	**0.038**	1.73	0.18
Texture (T)	0.78	0.394	0.11	0.743	1.33	0.268	2.38	0.14	3.88	0.062
W * D	0.297	0.879	0.15	0.96	0.27	0.896	0.38	0.823	0.80	0.525
W * T	0.27	0.614	0.01	0.932	0.71	0.412	0.54	0.469	1.66	0.212
D * T	1.40	0.249	1.55	0.214	0.31	0.733	1.57	0.209	1.72	0.181
W * D * T	0.93	0.398	1.63	0.197	1.07	0.346	0.96	0.386	1.18	0.309

Transformations: square root (richness), box-cox (beta diversity), x^3 (Pielou's evenness).

Table 6.22. Biological soil crust compositional responses to various pipeline disturbances in the Mixedgrass prairie, as determined through perMANOVA (distance = Bray-Curtis, permutations = 999). Distance from pipeline, age of disturbance, and diameters of pipeline were analysed in a perMANOVA blocked by site.

Factor	**Mean Square**	**F Model**	**R^2**	**P Value**
Age	0.12	0.71	0.01	**0.001**
Distance	1.37	7.86	0.08	**0.001**
Diameter	0.57	3.25	0.03	**0.001**
Interactions				
Age * Distance	0.00	0.00	0.00	0.998
Age * Diameter	0.32	1.81	0.02	**0.001**
Distance * Diameter	0.13	0.76	0.01	**0.004**
Age * Distance * Diameter	0.18	1.05	0.01	**0.044**

Table 6.23. Pairwise comparisons of biological soil crust composition at different sampling distances away from pipelines using perMANOVA (distance = Bray-Curtis, permutations = 999), blocked by site (n=18).

Distance[1]	Mean Square	F Model	R^2	P Value
Center vs. 1 m	0.11	1.28	0.04	**0.037**
Center vs. 5 m	0.68	4.57	0.12	**0.001**
Center vs. 20 m	0.60	4.07	0.11	**0.001**
Center vs. 55 m	1.55	10.46	0.24	**0.001**
1 m vs. 55 m	1.01	5.89	0.15	**0.001**
5 m vs. 55 m	0.21	0.90	0.03	0.106
20 m vs. 55 m	0.29	1.27	0.04	**0.009**
Work Area				
Trench vs. ROW	0.58	3.51	0.05	**0.001**
Trench vs. Undisturbed	1.50	10.16	0.23	**0.001**
ROW vs. Undisturbed	0.57	2.80	0.04	**0.003**

[1] Trench = Pipeline centre and edge; ROW = Right of Way = 0.5 m - 20 m, inclusive; Undisturbed = 25 m - 55 m, inclusive.

Table 6.24. Biological soil crust indicators associated with different pipeline disturbance treatments in the Mixedgrass prairie along a subset of sampling distances observed (pipeline center trench, 1 m, 5 m, 20 m, 55 m), pipeline age classes, diameters, and interactions thereof. Only those indicators with P < 0.10 are shown.

Distance	Species/Assemblage	A	B	P Value
1 m	Nostoc[2]	0.71	0.33	**0.004**
1 m, 5 m, 20 m, 55 m	*Cladonia pyxidata*	0.98	0.64	**0.006**
	Phaeophyscia constipate	0.98	0.33	0.095
	Selaginella densa	0.98	0.50	**0.048**
5 m, 20 m, 55 m	*Cladonia pocillum*	0.97	0.35	**0.008**
	Physconia muscigena	0.93	0.20	0.104
	Tortula ruralis	0.98	0.24	0.077
5 m, 55 m	*Diploschistes muscorum*	0.88	0.25	0.074
20 m, 55 m	*Cladonia cariosa*	1.00	0.14	0.093
Age Class (years)				
0 to 10 yrs	*Cladonia pocillum*	0.64	0.60	0.067
	Selaginella densa	0.52	1.00	0.092
21 to 30 yrs, 41 to 50 yrs	*Tortella fragilis*	0.99	0.20	0.073
Pipeline Diameter (mm)				
60.3, 88.9	*Xanthoparmelia camtschadalis*	0.94	0.31	0.082
60.3, > 168.3	*Ochrolechia upsaliensis*	1.00	0.22	0.094
88.9, > 168.3	*Polytrichum piliferum*	1.00	0.45	**0.013**
> 168.3	*Fulgensia bracteata*	0.83	0.20	**0.020**
Distance * Decade				
Undisturbed = 0 to 10 yrs	*Cladonia pocillum*	1.00	0.71	0.100
Distance[1] * Diameter (mm)				
Trench = 88.9, ROW + Undisturbed = All Diameters	*Cladonia pyxidata*	0.98	0.64	0.078
ROW =88.9, Undisturbed = 60.3 + 88.9	*Phaeophyscia constipate*	0.86	0.50	**0.041**

A = Probability of occurring, B = Fidelity

[1] Trench = Pipeline centre and edge; ROW = Right of Way = 0.5 m - 20 m, inclusive; Undisturbed = 25 m - 55 m, inclusive.

[2] Non-lichenized commune of cyanobacteria.

Table 6.25. Biological soil crust compositional responses (including lichen) to soil properties in Mixedgrass prairie as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m) from which soil was sampled. Lichen composition was also analysed separately.

	Soil Crust				**Lichens Only**			
Soil Property	**Mean Square**	**F Model**	**R^2**	**P Value**	**Mean Square**	**F Model**	**R^2**	**P Value**
C	2.23	14.89	0.13	**0.001**	0.91	12.15	0.09	**0.001**
N	0.28	1.88	0.02	0.122	0.22	3.00	0.02	0.059
C:N Ratio	0.05	0.31	0.00	0.906	0.02	0.22	0.00	0.901
OM	0.05	0.33	0.00	0.855	0.05	0.71	0.01	0.535
EC	0.84	5.60	0.05	**0.004**	0.48	6.46	0.05	**0.008**
pH	0.53	3.58	0.03	**0.028**	0.25	3.41	0.03	**0.046**
Significant Interactions								
C * N	-	-	-	-	0.27	3.62	0.03	**0.026**
C * pH	0.45	2.98	0.03	**0.003**	-	-	-	-

Table 6.26. Biological soil crust compositional responses to ground cover and litter mass in Mixedgrass prairie, as well as their interactions, as determined through perMANOVA (distance metric = Bray-Curtis, permutations = 999). Data were subset from 5 sampling distances along the pipeline disturbance (trench centre, 1 m, 5m, 20 m, and 55 m). Detailed ground cover was ocularly assessed and fallen litter was weighed.

Ground Cover	**Mean Square**	**F Model**	**R^2**	**P Value**
Bare Ground	1.09	10.70	0.06	**0.001**
Litter Cov.	4.92	48.09	0.29	**0.001**
Litter Mass	0.17	1.65	0.01	0.402
Stems	0.88	8.65	0.05	**0.001**
Significant Interactions				
Litter Cov. * Litter Mass	0.47	4.62	0.03	**0.020**
Litter Mass * Stems	0.50	4.87	0.03	**0.027**
Bare * Litter Cov. * Litter Mass * Stems	0.35	3.39	0.02	**0.029**

Table 6.27. Coefficients for mixed effects models of soil properties and their relationship to distance from the center trench, pipeline diameter and age, as well as soil texture.

Soil Property	Factor	β-Estimate	SE	DF	t-value	P-value
Carbon (%)	Intercept	1.634	0.125	14.49	13.054	**<0.001**
	Distance	0.0013	0.0010	71	1.243	0.218
	Diameter	-0.0004	0.0002	14	-1.516	0.152
	Age	-0.0048	0.0043	14	-1.113	0.248
	Ecosite Texture	0.479	0.069	14	7.016	**<0.001**
Nitrogen (%)	Intercept	0.131	0.012	14.48	10.745	**<0.001**
	Distance	0.0002	0.0001	71	2.104	**0.039**
	Diameter	-0.00003	0.00002	14	-1.334	0.203
	Age	-0.00034	0.00042	14	-0.815	0.429
	Ecosite Texture	0.046	0.007	14	6.899	**<0.001**
C:N Ratio*	Intercept	2.576	0.064	15.41	40.004	**<0.001**
	Distance	-0.0022	0.0009	71	-2.573	**0.012**
	Diameter	0.0003	0.0001	14	2.381	**0.032**
	Age	-0.0007	0.0022	14	-0.320	0.754
	Ecosite Texture	-0.072	0.035	14	-2.090	0.055
OM (%)	Intercept	3.058	0.234	14.6	13.094	**<0.001**
	Distance	-0.0006	0.0021	71	-0.278	0.781
	Diameter	-0.0001	0.0005	14	-0.186	0.855
	Age	-0.0119	0.0080	14	-1.488	0.159
	Ecosite Texture	0.846	0.127	14	6.657	**<0.001**
EC*	Intercept	4.642	0.428	14.14	10.856	**<0.001**
	Distance	-0.0075	0.0018	71	-4.068	**<0.001**
	Diameter	-0.0001	0.0008	14	-0.160	0.875
	Age	0.0074	0.0140	14	0.507	0.620
	Ecosite Texture	0.061	0.235	14	0.260	0.799
pH*	Intercept	1.892	0.017	16.22	109.754	**<0.001**
	Distance	-0.0007	0.0003	71	-2.523	**0.014**
	Diameter	-0.00001	-0.00003	14	0.433	0.672
	Age	0.00031	0.00058	14	0.544	0.595
	Ecosite Texture	0.009	0.009	14	0.996	0.336
Bulk Density	Intercept	0.959	0.032	14.32	29.906	**<0.001**
	Distance	-0.0002	0.0002	71	-0.906	0.368
	Diameter	0.0001	0.0001	14	1.275	0.223
	Age	0.0009	0.0011	14	0.815	0.429
	Ecosite Texture	-0.032	0.018	14	-1.851	0.085
Root Density	Intercept	0.0039	0.0009	14.7	4.435	**0.001**
	Distance	0.000	0.000	71.5	0.205	0.838
	Diameter	0.000	0.000	14	-0.171	0.867
	Age	0.000	0.000	14	0.751	0.465
	Ecosite Texture	-0.0023	0.0005	14	-4.936	**<0.001**

Soil Property ~ Distance + Diameter + Age + Ecosite Texture + (1|Site)
*Analysis of C:N Ratio, EC, and pH are based on log transformed data. Coefficients are derived from transformed data.

Table 6.28. ANOVA tests for legacy effects on soil properties at increasing distance from pipeline trenches where distance and ecosite texture are fixed factors.

Soil Property	Factor	F Value	P Value
Carbon	Distance (D)	0.79	0.534
(%)	Ecosite Texture (T)	49.58	**<0.001**
	D * T	1.52	0.207
Nitrogen	D	1.33	0.270
(%)	T	50.98	**<0.001**
	D * T	0.78	0.541
C:N Ratio*	D	1.13	0.349
	T	5.08	**0.039**
	D * T	0.23	0.922
OM	D	3.65	**0.010**
(%)	T	47.60	**<0.001**
	D * T	1.11	0.362
EC*	D	9.12	**<0.001**
	T	0.06	0.812
	D * T	1.99	0.106
pH*	D	16.01	**<0.001**
	T	0.83	0.376
	D * T	0.93	0.451
Bulk Density	D	1.56	0.196
	T	4.64	**0.047**
	D * T	0.56	0.692
Root Density	D	1.01	0.408
	T	50.30	**<0.001**
	D * T	0.08	0.986

Soil Property ~ Distance * Ecosite Texture + (1|Site)
Log transformed: C:N Ratio and EC.

Table 6.29. LS means (± SE) of soil properties along 18 pipelines at 5 sampling distances and among differing soil textures.

Factor		C (%)	N (%)	C:N	OM (%)	EC	pH	Bulk Density (g/cm3)	Root Density (g/cm3)
Distance	Trench				2.6 (±0.2) ab	210.3 (±128.3) a	7.1 (±0.1) a		
	1				2.7 (±0.2) ab	208.5 (±128.3) a	6.8 (±0.1) a		
	5				2.9 (±0.2) ab	155.4 (±128.3) b	6.4 (±0.1) b		
	20				3.0 (±0.2) a	119.4 (±128.3) b	6.4 (±0.1) b		
	55				2.5 (±0.2) b	181.5 (±128.3) b	6.5 (±0.1) b		
Ecosite Texture	Loam	2.0 (±0.1) a	0.17 (±0.01) a	12.4 (±1.0) b	3.6 (±0.2) a			0.95 (±0.03) b	0.0020 (±0.0008) b
	Sandy Loam	1.0 (±0.1) b	0.07 (±0.01) b	14.3 (±0.5) a	1.9 (±0.1) b			1.03 (±0.02) a	0.0068 (±0.0004) a

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Study III

Chapter 7

Recruitment potential of agronomic, escaped-agronomic, and native legumes from an artificial seed bank

7.1 Abstract

Legume species with diverse functional roles (native, agronomic, and escaped agronomics with invasive properties) were seeded into native and tame pastures in the Dry Mixedgrass prairie (DMG) and Central Parkland (CP) natural subregions in Alberta to monitor their recruitment from an artificial seed bank in established grasslands. During the first year of observation litter and vegetation height were manipulated to alter the plant community structure and microsite to resemble various states of disturbance. Each legume species performed differently, where potentially invasive species like *Melilotus officinalis* had good germination during the first year, but low survival and additional recruitment in later years, while *Astragalus cicer* demonstrated high dormancy and individuals emerged gradually from the seed bank over 3 years. Species were responsive to the microsites created through disturbance treatments and associated shifts in plant communities. Within native DMG, bare soil benefited the development of introduced forage legumes like *M. officinalis* and *Medicago sativa*. *Dalea purpurea* seedlings tended to become taller in microsites with high cover from established *Astragalus* in both natural regions.

7.2 Introduction

Establishment of new individuals or populations of plants typically starts from seed, while the success of recruited seedlings is highly regulated by the microenvironment in which they emerge and competition from established vegetation, acting as an environmental filter in community assembly (Booth and Swanton 2002). The latter in turn, are typically altered by ongoing disturbance regimes. In western Canadian prairie grasslands, dominant species tend to be perennial graminoids (Poaceae and Cyperaceae) and perennial forbs from diverse taxa. Dense foliage, litter, and limited niche space often limit the recruitment potential of annual ruderals, ephemeral species, and perennials that propagate through vegetative means once established (Coffin and Laurenroth 1989; Ma et al. 2010; Sanderson et al. 2014), such as grasses. In western Canadian grasslands, legumes are typically desirable herbaceous plants in both native, semi-native, and seeded (referred to as 'tame' from here on) grassland for their nitrogen fixation potential from *Rhizobia* within root nodules, which benefit the plant community by increasing available nitrogen (Freedman 2010). Legumes, inherent in high forage quality, benefit both domestic livestock (Ledgard and Steele 1992) and wild ungulates (Semiadil et al. 1995), and contribute to floristic diversity including after disturbance (Bork et al. 2002), thereby supporting overall biodiversity (i.e. pollinators and other arthropods, etc.) (Woodcock et al. 2014). Native grasslands in Alberta generally support a high diversity of native legumes (major genera including *Astragalus*, *Dalea*, *Oxytropis*, *Pediomelum, and Vicia*) from the Parkland to Dry Mixedgrass ecoregions; while semi-native and tame grasslands exhibit dominance of seeded or voluntary forage legumes like alfalfa (*Medicago* spp.) and clovers (*Trifolium* spp.).

Some legumes deter herbivory through toxic alkaloids and thereby aid in their conservation (Smolenski et al. 1981); otherwise, the highly palatable nature of legumes would leave them susceptible to overutilization. Repeated defoliation can reduce legume reproductive potential (i.e. less seed production or vegetative propagation), depleting energy stores and productivity, and in some species leading to their removal from the community (Smith et al. 1988). Within rangelands, legumes are also susceptible to other management factors such as broad-leaf herbicides, which directly limit legume productivity and survival through incidental foliar contact with non-target species during weed control (Grekul and Bork 2007; Bork et al. 2007). Moreover, residual herbicides also influence legume survival and recovery indirectly through their residual nature (Miller et al. 2015). Rejuvenation of pastures through fertilizer has been shown to lead to overall reductions in legume biomass and cover (Aydin and Uzun 2005, Lardner et al. 2001). Sustainable legume populations require the recruitment of new individuals from seed to improve community genetic diversity and buffer the community against acute (i.e. herbicide) and prolonged (i.e. grazing regime, climatic variability, etc.) disturbance. Grazing-tolerant legumes, like white clover (*Trifolium repens*), volunteer from the seed bank when soil is disturbed (Barret and Silander 1992), and it is possible that other agronomic species could similarly benefit from low intensity disturbances.

Palatable introduced legume species have been commonly included in forage (Sleugh et al. 2000) and reclamation mixes (Gardiner 1993; Simmers and Galatowitsch 2010). As a result, some species have emerged as invasives propagating outside of their ‘intended environments’ like seeded pastures, ditches, well sites, and pipelines through the formation of persistent seedbanks (Klemow and Raynal 1981), where they compete with established vegetation. In some cases, introduced species may be seeded into native rangelands to improve forage production (Mortenson et al. 2005), but then encroach into adjacent areas. For species like sweet clover (*Melilotus* spp.) and cicer milkvetch (*Astragalus cicer*) there is limited recognition of their role as invasive species despite their potential deleterious effects on native grassland communities. For example, sweet clover exhibits invasive properties in resource-limited environments and is known to exploit disturbed areas such as roadsides and pipelines (Wolf et al. 2008). Furthermore, sweet clover is structurally taller, shading native vegetation, and its leaf litter creates a nitrogen enriched microsite that facilitates ongoing invasion of exotic species (Van Riper and Larson 2009). Unlike sweet clover, cicer milkvetch emerges stochastically within native grassland, likely from pats dispersed by livestock (Willms et al. 1995) and exploits mesic range sites and disturbance (personal observation). Like sweet clover, cicer milk vetch structurally differs from native grassland vegetation having numerous prostrate branching stems, which would similarly suppress nearby native vegetation.

Micro-environmental conditions are likely a factor in legume recruitment, as the biophysical characteristics of the niche space needed to induce germination, thereby allowing seedling establishment. Ground cover characteristics like the proportion of exposed bare soil, litter cover, biological soil crust, or plant cover (i.e. basal area occupied by bunch grass crowns, etc.) regulate soil temperature, light availability, and soil moisture (Li et al. 2005; Facelli and Pickett 1991). Disturbances like grazing that influence vegetation structure and litter abundance (Adams et al. 2005), as well as plant community composition, can create the niche space and suppress competition.

Mechanisms regarding the recruitment and persistence of legumes in natural environments are poorly understood and understudied, particularly for native species. In agro-pastoral systems, the mechanisms regulating forage legume populations are better understood and therefore serve as a source of information to better understand grasslands (Barret and Silander 1992; Groya and Sheaffer 1981). Legume establishment and persistence is often influenced by soil fertility, where high soil nitrogen availability limits the competitiveness of newly established legumes (Turnbull et al. 2005). Soil resources are influenced by the functional traits of established plant species, which can influence a community's susceptibility to invasion by legumes (Turnbull et al. 2005). In pastures, sod seeded legumes are typically suppressed by competitive grasses and their establishment is improved when management activity suppresses competitive vegetation, the latter of which could be achieved through grazing and herbicide application (Groya and Sheaffer 1981; Kunelius and Campbell 1984).

Recruitment potential of legumes from a seed bank is limited in-part due their seed biology. Legume seeds have thick indurate seed coats, which aid in physical dormancy but are a barrier to water absorption, and hence are often described as impermeable (Acharya 2006; Baskin et al. 2000; Russi et al. 1992; Tracy and Sanderson 2000). Given appropriate conditions for germination, non-permeable seed coats could delay germination; seed coats require degradation through mechanisms like cold stratification, physical or chemical scarification, heating, and aging (Acharya 2006; Baskin et al. 2000). These factors contribute to persistence in the seed bank and potentially conflate legume seed density in grassland seed banks. Legume germination is also influenced by seed size and the depth of seeds in the soil profile, where larger seeds of a species have a higher probability of recruiting and positional depths up to a few centimeters below the soil surface can yield better germination (Townsend 1972).

The seed bank studies described in previous chapters characterized the abundance of germinable legume seeds in a unit of soil. However, within naturally-occurring perennial pasture one would not expect germination in the green house to reflect recruitment in a natural environment. Consequently, this study follows-up on these results with a more detailed mechanistic evaluation of legume demography within experimental plots wherein microsite characteristics are manipulated (through simulated grazing and litter removal) to potentially alter the conditions for legume seed germination and recruitment. Individual seedling emergence, survival, and persistence of individuals were monitored over 3 years to quantify the recruitment potential of 6 legume species commonly found across Alberta grasslands.

Plantings were further conducted in each of two types of grasslands (native and tame grassland), at each of two natural subregions (Central Parkland and Dry Mixedgrass Prairie). The specific objectives of this study were to 1) quantify legume emergence and survival in response to varied defoliation of established vegetation and litter removal treatment, 2) evaluate legume recruitment potential over time within different grassland environments, and 3) interpret legume establishment success in light of changes in the environment and overlying plant community. I hypothesized that defoliation would favour short statured legumes resistant to grazing such as white clover (Briske, 1996; and Brummer and Moore, 2000). At the seed level, heterogeneity in the plot's amount of bare ground, soil crust, cover of sod and bunch grasses, could affect which seeds germinate and recruit into the plant community. Therefore, I further hypothesized that the microenvironment within plots will be significantly affected by clipping and litter removal, whereby 1) clipping improves light availability and creates higher temperatures; 2) litter removal will improve light availability at the soil surface and increased soil surface temperatures; 3) ambient litter depths should have a cooler soil surface and higher soil moisture; and 4) when clipping and litter removal are combined the environment will have the highest light availability, soil surface temperature and lower soil moisture.

Overall, recruitment and survival metrics were measured to identify relative competitive abilities of the selected species in different grasslands and microsites and provide insight into legume population dynamics. This has implications for management attempting to improve pasture or grassland for restoration or rejuvenation (i.e. improving floral diversity or attracting pollinators), or with improved forage quality in mind. The inclusion of potentially invasive legumes should provide insight into processes that promote or limit their establishment in native grassland.

7.3 Methods

7.3.1 Study Locations

Two experimental sites were seeded at each of two locations within perennial pasture, including the Roy Berg Kinsella Research Ranch [Central Parkland (CP)] and Mattheis Research Ranch [Dry Mixedgrass Prairie (DMG)] (Fig. 7.1) in spring of 2014, and monitored for three growing seasons. At each location treatments were conducted in both native (N) and tame (T) grasslands (Fig 7.2). Long-term mean precipitation at the CP and DMG locations were 411 and 330 mm, respectively, while mean annual temperatures were 1.2ºC and 10.9 ºC, respectively; note that average precipitation was observed during the initial establishment year (2014), proceeded by a spring drought in 2015 (Fig. 7.3). Native grasslands were dominated by later seral grasses typical of their natural subregion, with *Hesperostipa comata, Pascopyrum smithii, Koeleria macrantha,* and *Bouteloua gracilis* dominating the DMG-N site and *Festuca hallii* with *Hesperostipa curtiseta* dominating the CP-N sites. Tame grasslands at both the DMG and CP locations consisted of *Bromus inermis* and *Medicago sativa*. Pivot irrigation occurred just adjacent to the DMG-T site, and therefore sub-irrigation may have influenced vegetation at this site. Soils in the more northern CP sites were Orthic Black Chernozems with favorable organic matter (Table E.1). Soils at the southern DMG sites were Orthic Brown Chernozems, with soil textures generally similar at all four sites being classified as sandy-clay-loams on loamy ecosites (Table E.1). Other minor differences were apparent in soil characteristics, including soil organic matter (OM), nitrogen (N), pH and electrical conductivity (EC) (Table E.1).

7.3.2 Legume Species

Six legume species were examined with diverse functional roles in Alberta's grasslands. White clover (*Trifolium repens*) and alfalfa (*Medicago sativa*) were selected for their importance as desirable forage legumes in tame pastures. Alfalfa is more sensitive to grazing pressure than white clover and tends to be more abundant in newly seeded high-performance pastures. The escaped agronomic legumes cicer milkvetch (*Astragalus cicer*) and sweet clover (*Melilotus officinalis*) were included as they have demonstrated invasibility on the two Research Ranches under investigation. Finally, two native palatable legume species, American vetchling (*Vicia americana*) and purple prairie clover (*Dalea purpurea*), were also included; these native legume species occur at both Research Ranches although purple prairie clover is more abundant in Alberta's DMG prairie.

7.3.3 Germination Trial

Seed was sourced from BrettYoung™ and represented native seed that would be used for revegetation of reclaimed disturbance and prairie restorations in Alberta. We specified that seed should not be inoculated, coated, or scarified. *Astragalus cicer* seed was acquired from Agriculture and Agri-Food Canada after a shipment of treated seed and came with a cultivar description, Oxley II. The germination potential of all species was initially tested in a preliminary trial where 5 sets of 100 seeds per species were germinated over two months on moist filter paper in a sealed plastic container in darkness. A second trial under similar conditions and replication included seeds glued with white Elmer's glue to plastic toothpicks (10 seeds per toothpick) to determine if glue inhibited germination. Seeds were checked periodically (daily initially, weekly after germination slowed) for germination and counted, then discarded. After two months the remaining seeds were counted and checked; if they had hard seed coats they were counted as 'hard' seeds (i.e. viable but dormant), while if they had degraded (usually from mold) and failed to germinate this was recorded as well.

7.3.4 Field Experiment Seeding Legumes into Pasture

7.3.4.1 Treatments

Soil moisture, soil temperature, and light availability were manipulated through two disturbance treatments in a factorial design: defoliation (simulated grazing via clipping), litter removal (gentle raking of litter prior to seeding), both defoliation and litter removal, and no treatment. Disturbance treatments were set up as whole plots (1 m x 6 m in size) and were then randomly seeded with the six legume species into 1 x 1 m subplots in a factorial, split-plot design, with four replications of each whole plot at each site and location (Fig. E.1, E.2.). Defoliation (+D) was applied every 3 weeks during the first growing season to 5 cm height commencing in late May (CP) and early June (DMG), with a final clip done at the end of the growing season (in September) within all plots to 5 cm height in 2014 and 2015 to remove excess live biomass and standing litter; the latter treatment also defoliated any emergent seedlings. Litter was gently raked during plot establishment to remove fallen and standing litter resulting in two treatment levels: ambient litter (+L) and removed litter (-L). Disturbance to soil surface and biological crusts were minimized during litter removal, as the goal of this treatment was to reduce litter to levels that may occur when ongoing defoliation removes standing biomass and reduces litter accumulation.

Plots were seeded in late May 2014, with 40 seeds of each species installed per plot in an 8 x 5 grid, distributed across a 0.25 m^2 area placed inside the center of the 1 m x 1 m subplot. To facilitate planting, seeds were glued to toothpicks with white Elmer's glue and then inserted just below the soil surface. Germination, survival, and persistence were then monitored every three weeks (starting just before clipping treatments began) throughout the growing season (May to August) and less regularly in autumn (September and October). At the end of 2014, 2015, and 2016 (late August or early September) and prior to year-end defoliation, the height and growth stage were determined for each emergent seedling. Growth stages were assessed based on a method developed for alfalfa by Fick and Mueller (1989), with additional early stages added to describe seedling development (Table E.2.). It should be noted that all legumes chosen experience epigeal germination, except for *Vicia,* which has hypogeal germination and therefore *Vicia*'s cotyledons remain underground and were not visible during staging. For the second (2015) and third (2016) growing season, established plants and seedlings were clipped and assessed for bud production, flowering, and seed production; however, these data were sparse due to few established and surviving plants, and thus will not be presented.

7.3.4.2 Characterization of Microsite Conditions and Plant Competition

Soil surface temperature, light availability, and soil moisture were measured approximately weekly during June and July of 2014, and roughly every three weeks through to October 2014, and again from May through July in 2015. Soil surface temperature was measured with an infrared laser thermometer aimed below the surface litter or in a representative area. Light availability was measured on days with uniform sky conditions with a fish-eye light meter placed at the soil surface and under any existing litter and vegetation within a representative area of the plot; readings were taken on photon flux density (PFD). For each microsite variable measured, repeated subsamples (n=2) were taken and averaged before analysis. Soil moisture (%) was measured using a time domain reflectometer (TDR) with 10 cm probes. Additionally, in July of 2014 and 2015, the soil surface litter depth and standing vegetation height were measured at random points within the plot.

Because treatments were expected to alter overstory plant species composition, foliar cover was estimated in July 2014 and 2015 (i.e. at peak growth) using a 0.25 m^2 frame centered around the grid of seedlings. Ground cover of bare soil, lichens, litter, and plant crowns was also estimated. To quantify microsite effects at the seed level, we made replicate grids on paper representing detailed seed placement and drew polygons around seeds near bunch grasses and forbs with basal crowns (i.e. alfalfa) and identified the neighboring plants by species. Polygons were also drawn around patches of bare soil and species that adhered to the soil surface (i.e. *Antennaria* spp. or *Selaginella densa*). Seeds were classified as being seeded into crowns, immediately adjacent caespitose species, in sod, or in bare soil. These data were intended for analysis of potential competitive effects on seedling establishment, and the influence of microsite heterogeneity at the seed level, however, they will not be presented here.

7.3.5 Formulas and Definitions

Germination refers to the number of seedlings or recent emergents of seeded species observed during a growing season, expressed as the proportion of germinants that emerged from the original seed bank (i.e. 40 seeds per plot). Recruitment refers to the total number of seeded individuals alive and observed during a given growing season. In the first year, recruitment was equal to germination. In the second and third growing season, however, recruitment included new germinants in addition to any survivors from previous years. Recruitment is therefore the proportion of observed individuals out of the original seed bank. Mortality was only analyzed for the turn-over between the first and second growing season due to limited data availability for the third year. Mortality was defined as the number of observed survivors from the second year divided by the number of germinants from the first year. Finally, we calculated the number of individuals required to survive one time-step (i.e. 1-year) with 95% confidence.

$$P = X^n \quad \text{or} \quad n = \frac{logP}{logX}$$

Where P = 1-0.95 (95% probability), X = 1-survival rate between time-step, and n = the number of individuals. Note that this formula created infinite values when 0 individuals survived, and therefore these values were excluded from the analysis.

7.4 Statistical Analysis

The experiment was designed as a factorial, randomized split block with defoliation and litter removal treated as factorial main plots, and species (n=6) nested within main plots. Using defoliation and litter removal as factorial effects allow for their relative importance and any interactions among them to be determined. Data were analyzed separately for each location (DMG or CP) and site (N or T) therein, because of distinct *a-priori* differences in climate, soil conditions and vegetation composition, making the probability of site specific effects high, which in turn, would have necessitated further analysis by site anyway. Within each site, defoliation (D), litter removal (L) and species (S) were analyzed as fixed effects, while replicates were considered random effects. In cases where the height and growth stage of individual seedlings were assessed, plot was considered random. For repeatedly measured characteristics like soil surface temperature, PFD, and soil moisture sampling, time was analyzed as a random effect with mixed models to assess generalized differences in microsite conditions in relation to the treatments. Demographic variables (germination, recruitment, mortality, seedling height, seedling growth stage, etc.) and other overstory vegetation factors (litter depth, foliar cover and indices of diversity) were compared among treatments and species using analysis of variance (ANOVA) with type III sums of squares in R software (R Core Team 2017) with *lme4*, a package for linear mixed effects models (Bates et al. 2015). When residuals were normally distributed, data were analyzed using linear mixed-effects models (LMM) with the function *lmer*. Data transformations (square-root and log) were applied when necessary and indicated in the tables of results below. Contrasts (Tukey HSD corrected) were conducted to further identify significant effects within any 2-way interactions with least square (LS) means; 3-way interactions were not explored in detail. All data presented are least-square (LS) means for non-transformed data to maintain interpretation.

Limited germination and recruitment was observed in 2015 and 2016. Residuals of mixed models were visually checked for normality and these variables were transformed with log (x + 0.01) and analyzed with LMMs. Growth stage data assessed in mixed models failed normality tests, though residuals visually appeared to be normally distributed, and were therefore analyzed with LMMs. Because data from subsequent years were limited, mortality rate and the number of individuals required to survive a time-step (annual, between growing seasons) were analyzed between the first and second growing seasons. Mortality was analyzed using generalized linear mixed models (GLMM) using *glmer* in the *lme4* package set to a binomial distribution due to inflation around 100% mortality. The number of individuals required to survive 1 time-step were grouped by species with the treatment effects ignored due to the low survival overall within species and compared among species using a non-parametric Kruskal-Wallis test with the *agricolae* package in R (De Mendiburu 2017); contrasts were Bonferroni corrected. When species altogether failed to survive they were also dropped from the analysis, at some experimental sites there were entire species that failed to survive.

Shifts in plant community composition under litter removal, defoliation, and their interaction were tested using permutational analysis of variance (perMANOVA) set to the Bray-Curtis index of similarity, with replicate blocks within each site used as a random factor. This test used the *adonis* function in the *vegan* package (Oksanen et al. 2017). When an interaction was detected 2-way comparisons with perMANOVA were conducted for all contrasts. Patterns in plant community data from each experimental location were evaluated individually using non-metric multidimensional scaling (NMDS) ordination with the *metaMDS* function in *vegan* using a Bray-Curtis distance metric, and 999 permutations (Oksanen et al. 2017). Solutions were limited to 2 dimensions to maintain interpretive quality of the results. Plant community cover and ground cover metrics, plant community indices (richness, Shannon's diversity, and Pielou's evenness), seedling characteristics by species (height, stage, germination, mortality), and microenvironment measurements (temperature, light availability, and soil moisture) were included as vectors in associated NMDS biplots when identified as significant ($P < 0.05$)

using the *envfit* function in *vegan* (Oksanen et al. 2017). Individual species included in ordination graphs were limited to significance levels of P < 0.05.

7.5 Results

7.5.1 Germination Trial

Initial germination tests on moist filter paper showed that the glue coating did not inhibit the probability of germination (P = 0.587), nor effect the abundance of hard, ungerminated seeds (P = 0.410) or the probability of seed degradation (P = 0.832) (Table 7.1). Different species exhibited unique germination potentials (P < 0.001, Table 7.1), with *Dalea, Medicago,* and *Melilotus* having high germination potential (93.3 to 95.5 ± 1.3 % 1 SE), while *Astragalus* and *Vicia* had low potential (77.4 and 74.4 ± 1. 3% respectively); *Trifolium* had an intermediate probability of germinating (Table 7.2). After a 2-month germination period, *Vicia* retained the most 'hard' (dormant) seeds (19.8 ± 1.0 %), with *Dalea, Medicago,* and *Melilotus* had the lowest dormancy potential (Table 7.2). *Astragalus* seeds were the most effected by degradation (largely mold) (13.7 ±1.0 %). Germination did not differ with the interaction of species and glue treatment (P = 0.488) (Fig. 7.4) There were significant interactions between species and glue exposure for the proportion of hard seeds and degraded seeds observed (P ≤ 0.003); where *Vicia* with glue had the most dormant seeds remaining and *Melilotus* had relatively higher proportions of dormant seeds with glue (Table 7.2). *Melilotus* seeds were less likely to degrade when coated in glue, while *Medicago* and *Trifolium* had an increased probability of degrading with glue (Table 7.2).

7.5.2 Field Experiment

7.5.2.1 Microsite and Overstory Plant Community Competition

Both the defoliation and litter removal treatments reduced surface litter depths at all four sites (P < 0.05), and there was an interaction between litter removal and defoliation at all sites (Table 7.3). Litter depths were consistently highest in control plots and lowest in plots that had litter removal - with or without defoliation (Table 7.4). Litter removal also consistently increased the proportion of bare soil exposed (P < 0.001) (Table 7.3 and Table 7.4). During the initial establishment year, live vegetation height was reduced from the defoliation treatment ($P \leq 0.016$). At the DMG location, initial litter removal reduced subsequent live vegetation height within the native site ($P < 0.001$), while in the DMG-T site there was a significant interaction between litter removal and defoliation ($P = 0.002$). The latter resulted in taller vegetation in control plots, while ambient litter with defoliation had the shortest vegetation (Table 7.3 and Table 7.4). Although no additional defoliation 'treatments' occurred during the growing season in 2015, carry over effects of defoliation from 2014 were reflected in standing vegetation heights ($P \leq 0.045$), except for the CP-N site, where only litter removal the previous year resulted in shorter vegetation ($P = 0.046$) (Table 7.3 and Table 7.4).

Litter removal and defoliation were intended to alter soil surface temperature, soil moisture, and the light available to seedlings (photon flux density). Soil surface temperature was consistently increased by defoliation in 2014 ($P < 0.001$). Within the DMG-N site, litter removal also increased soil temperatures ($P < 0.001$), and in the CP-T site, control treatments had the lowest soil surface temperature compared to plots that had treatments (defoliation and/or litter removal) applied in any combination ($P < 0.001$) (Table 7.3 and Table 7.4). Defoliation led to carry over effects on soil surface temperature into 2015 at the DMG-N site ($P = 0.015$), resulting in higher soil surface temperature. PFD was strongly influenced by all treatments at all sites in 2014 ($Ps \leq 0.037$), where litter removal, defoliation, and the combination thereof, resulted in the highest PFD available to seedlings (Table 7.3 and Table 7.4). Carry over environmental effects on PFD in 2015 resulted primarily from litter removal treatments ($P \leq 0.031$), resulting in greater light availability. In the DMG, defoliation treatment also resulted in carry over (into 2015) of greater light availability ($P \leq 0.007$) (Table 7.3 and Table 7.4). In 2014, there were limited significant effects on soil moisture: in the DMG-T site, defoliation increased moisture ($P < 0.001$) while in the CP-N site, defoliation decreased moisture ($P = 0.022$) and also interacted with litter removal to further decrease soil moisture ($P = 0.048$) (Table 7.3 and Table 7.4). In 2015, soil moisture increased with litter removal at the CP-N site ($P = 0.043$) and decreased at the CP-T site ($P = 0.009$), while in the DMG-T site, control plots had the lowest soil moisture while ambient litter with defoliation improved soil moisture ($P = 0.023$) (Table 7.3 and Table 7.4).

Plant community composition during the first year (2014) was affected by the treatments imposed, with the DMG-N site exhibiting strong community shifts from litter removal (P = 0.001), defoliation (P = 0.001), and their interaction (P = 0.026) (Table 7.5). Unlike the DMG-N site, the DMG-T community did not demonstrate significant shifts with treatment. For the CP-N site, litter removal (P = 0.010) and defoliation (P = 0.001) altered the community, while the CP-T grassland community was only affected by defoliation (P = 0.023) (Table 7.5). In both native grasslands, litter removal had a strong effect on native grass cover (P ≤ 0.006), which decreased in the DMG-N but increased in the CP-N (Table 7.6). In contrast to the latter, litter removal was associated with reduced native forb cover in the CP-N site (P < 0.001). Defoliation decreased the foliar cover of native grasses (P < 0.001) and introduced forages (P < 0.001) at the respective DMG sites, while foliar cover of all major vegetation groups was unaffected by defoliation at the CP sites (Table 7.6). Plant species richness was affected by litter removal at both native study sites (P ≤ 0.034), resulting in increased richness within the DMG-N site, while litter removal decreased plant richness of native fescue prairie in the CP (Table 7.6 and 7.7). Community richness, diversity, and evenness in the DMG-T site were affected by litter removal and defoliation, with richness, diversity, and evenness each decreasing with litter removal, and defoliation increasing richness, diversity, and evenness (Table 7.7).

7.5.2.2 Dry Mixedgrass Native Site (DMG-N)

Germination during the first year within the DMG-N site was influenced by the interaction between litter removal and defoliation treatments (P = 0.049), and by species identity (P < 0.001) (Table 7.8). Overall, the combination of litter removal and defoliation (-L+D) improved germination in the DMG-N site (to 24.8 ± 2.6 %), while defoliation without litter removal (+L+D) resulted in the lowest germination (16.4 ± 2.6 %) (Table 7.9). The introduced legumes *Medicago*, *Melilotus*, and *Trifolium* had the highest germination rate in the field during the first year ranging from 25.2 to 30.8 ± 2.8 %, while the germination of native *Dalea* and *Vicia,* in addition to introduced *Astragalus*, had lower germination, at a level of about half the other introduced species, ranging from 13.1 to 14.4 ± 2.8 % (Table 7.9). Germination and total recruitment from the original seed bank was relatively low during the second and third years, and combined with high mortality, overall low recruitment (survivors plus new germination) was observed the second and third year. Recruitment after the first year was only influenced by species ($P \leq 0.001$), with *Astragalus* having the highest recruitment in the second and third year (Table 7.8 and 7.9). During the second year, *Vicia* had the second highest germination with 1.1 ± 0.6 %, which combined with winter carry over, led to 4.1 ± 0.6 % of the *Vicia* initially seeded leading to seedling recruitment (Table 7.9).

Mortality rates following the first year's recruitment in DMG-N was affected by litter manipulation ($P = 0.027$), the interaction of litter manipulation and defoliation treatments ($P = 0.001$), legume species ($P = 0.026$), the interaction of species and litter manipulation ($P = 0.034$), and a three-way interaction between all treatments and species ($P = 0.021$) (Table 7.10). Mortality rates were highest in DMG-N when the grassland was defoliated and left with an intact litter layer, while defoliated plots with reduced litter had the lowest mortality rate (Table 7.11). The introduced forage species *Medicago, Melilotus*, and *Trifolium* had mortality rates exceeding 99 %, while *Vicia* had the lowest mortality rate at 83.3 ± 2.9 % (Table 7.11). *Dalea* seedlings had higher mortality when there was ambient litter (98.3 ± 4.4 %) and lower mortality when litter was reduced (91.8 ± 4.1 %) (Table 7.11). The DMG-N site was the only grassland found to contain significant differences among species in the number of individuals required to survive one time-step (Fig. 7.5); *Trifolium* required the most individuals (>50) while *Astragalus* and *Vicia* required the least (~10). *Medicago* had no survivors and was thus unable to contribute to the analysis, although it could be interpreted as an infinite value, or at a minimum, a much larger value than the number of individuals seeded.

Seedling height in the DMG-N site was influenced by an interaction between species and litter removal ($P < 0.022$), and heights were reduced under defoliation ($P < 0.001$) (Table 7.12 and Table 7.13). *Vicia* seedlings were the tallest at 5.74 ± 0.24 cm, while *Astragalus* and *Trifolium* seedlings were the shortest at < 1 cm (Table 7.13). The mean growth stages of germinants did not advance much beyond small seedlings at stage 3 (with 2 or more leaves) and remained < 5 cm tall, while introduced forages

Astragalus, Medicago, and *Trifolium* did not advance much beyond stage 2 - defined as having a single true leaf present (Table E.2 and Table 7.13).

Mixed models used to relate germination rates to overstory vegetation characteristics in the DMG-N site demonstrated significant associations of first year germination for *Medicago, Melilotus, Trifolium,* and *Vicia* (Table 7.14). *Medicago* germination was positively associated with introduced forage cover ($P = 0.002$). *Melilotus* germination was positively associated with native grass cover, plant species richness, and Pielou's evenness, while germination of this same species was negatively associated with Shannon's diversity, litter cover, litter depth, and *Selaginella densa* cover ($P \leq 0.021$). *Trifolium* germination was positively associated with native grass cover, native forb cover, lichen cover, and *Selaginella densa* cover, while it was negatively associated with bare soil exposure ($P \leq 0.042$). *Vicia* germination was positively associated with Shannon's diversity, while germination was negatively associated with plant species richness and Pielou's evenness ($P \leq 0.034$).

NMDS ordination of plant community composition at the native Dry Mixedgrass (DMG-N) location in 2014 (stress = 0.27, dimensions = 2, distance = Bray-Curtis) further identified legume seedling responses relative to changes in the overlying plant community (Fig. 7.6). *Dalea* seedlings were generally tallest in plots that were defoliated and had high cover of native perennial forbs like scarlet mallow (*Sphlaeracea coccinea*) and purple milkvetch (*Astragalus agrestis*). In plots where litter was reduced but plants were not defoliated (-L-D) there was greater bare soil cover, and this was associated with the forage legume seedlings of *Medicago* being tall and *Melilotus* advancing to later developmental stages. Where litter removal had occurred, there was greater light availability (PFD), which was associated with greater plant community richness, diversity, native forbs, introduced grasses, lichen, and *Selaginella* cover, but this was not associated with legume demographics or vigor during the initial year.

7.5.2.3 Dry Mixedgrass Tame Site (DMG-T)

During the first year at the DMG-T site, germination was generally affected by litter removal ($P = 0.041$), and legume species also had distinct germination rates ($P < 0.001$), with a species interaction with defoliation treatments ($P = 0.007$) (Table 7.8). Within the DMG-T site, litter removal improved overall germination from 11.2 to 16.6 ± 1.5 %. The introduced legumes *Medicago*, *Melilotus*, and *Trifolium* had the highest germination rate in the first year, ranging from 17.3 to 22.7 ± 2.4 %, while the germination of native *Dalea* and *Vicia,* in addition to introduced *Astragalus,* germinated at a level near half of the other introduced species, ranging from 8.1 to 8.8 ± 2.4 % (Table 7.9). The interaction of legume species and defoliation resulted in three-fold greater *Trifolium* germination when the standing vegetation was not defoliated, while *Medicago* germination (26.9 ± 3.4%) was highest when plots were defoliated (Table 7.9). Germination and total recruitment from the original seed bank was low during the second and third year at this site. During the second growing season germination differed only by species ($P < 0.001$), where *Astragalus* germinated the most, with trace (<1%) numbers of *Dalea, Medicago,* and *Vicia* emerging (Table 7.8 and Table 7.9). Second year recruitment showed that *Medicago* was the most abundant, while no *Melilotus* or *Trifolium* were recruited (Table 7.9). There were no significant effects on germination or recruitment in the third year. Mortality rates between the first and second growing season differed only by species ($P < 0.001$): *Medicago* (87.8 ± 2.9 %) had the lowest mortality rate followed by *Astragalus* (91.4 ± 2.9 %) (Table 7.5 and Table 7.6). Legume seedling height and stage during the first year did not differ by species or the other treatments ($P > 0.05$) (Table 7.12).

Mixed models used to relate germination rates to overstory plant community characteristics in the DMG-T site revealed first year germination was affected for *Astragalus, Medicago,* and *Melilotus* (Table 7.15). *Astragalus* germination was negatively associated with introduced forage cover ($P = 0.044$). *Medicago* germination was positively associated with introduced forage cover and Pielou's evenness, while litter depth was negatively associated with germination ($P \leq 0.001$). *Melilotus* was also negatively related to introduced ruderal forb cover ($P = 0.020$).

NMDS ordination of plant community composition at the tame Dry Mixedgrass (DMG-T) location in 2014 (stress = 0.21, dimensions = 2, distance = Bray-Curtis) shows that higher soil surface temperatures and light availability (PFD) were associated with the litter removal and defoliation treatment (-L+D) which was correlated with greater richness and introduced ruderal forbs like Russian thistle (*Salsola tragus*) and wild buckwheat (*Polygonum convolvulus*) (Fig. 7.7). Plots dominated by established

Medicago plants were associated with greater community evenness and greater soil surface area occupied by vegetative stems, shoots, and crowns. Recruited *Medicago* seedlings were tallest in plots with high diversity and intermediate between plots dominated by established *Medicago* and introduced ruderals. Other legumes were not significantly associated with plant community composition.

7.5.2.4 Central Parkland Native Site (CP-N)

Germination during the first year at the CP-N site only varied by species ($P < 0.001$, Table 7.8), with native *Dalea* and introduced *Medicago* and *Melilotus* exhibiting the highest germination rates, followed by intermediate germination by *Astragalus* and *Trifolium,* while *Vicia* had the lowest probability of germinating (Table 7.9). Germination and recruitment during the second growing season again differed by species ($P < 0.001$), with *Astragalus* having the greatest germination corresponding with the highest recruitment. *Medicago* and *Melilotus* did not germinate during the second year, but existing seedlings did carry over from the first year. New individuals of *Vicia* germinated at the second highest level in year two and accounted for all of the second year recruitment in this species. In the third year, plots with reduced litter had higher legume germination and overall recruitment ($P \leq 0.037$), and legume species interacted with litter removal, as did the combination of defoliation and litter removal ($P \leq 0.02$) (Table 7.8 and 7.9). *Astragalus* had the highest germination and recruitment during the third year, which was improved by litter removal and defoliation treatments conducted early in the study (in 2014, the first year). Seedling mortality rates between the first and second growing season did not differ among species or treatments (Table 7.10).

Both seedling height and growth stage differed between species within the CP-N site ($P \leq 0.002$). While the native legume *Vicia* was the tallest, introduced *Medicago* and *Trifolium* were the shortest, and all the latter introduced species also exhibited the least development, typically failing to reach growth stage 3 (Table E.2, 7.12, and 7.13).

Mixed models used to relate germination rates to overstory characteristics in the CP-N site found first year germination was affected for *Medicago, Melilotus,* and *Vicia* (Table 7.16). *Medicago* germination was positively associated with plant species richness and Pielou's evenness, while germination was negatively influenced by Shannon's diversity ($P \leq 0.042$). *Melilotus* germination was positively associated with Shannon's diversity of vegetation and litter depth, while germination was negatively associated with native forb cover, plant species richness, Pielou's evenness, bare soil cover, litter cover, and lichen cover ($P \leq 0.038$). *Vicia* germination was negatively associated with native forb cover ($P = 0.039$).

NMDS ordination of plant community composition at the native Central Parkland (DMG-T) location in 2014 (stress = 0.26, dimensions = 2, distance = Bray-Curtis) showed that warmer soil temperatures and light availability were associated with greater native grass cover from caespitose species like blue grama (*Bouteloua gracilis*), western porcupine-grass (*Hesperostipa curtiseta*), and Junegrass (*Koeleria macrantha*) (Fig. 7.8). There was also strong divergence in composition from litter removal (-L), which was also associated with greater native grass cover. *Dalea* was responsive to plant community composition, preferentially emerging where there was a greater cover of introduced ruderal forbs, greater litter depth, and established native legumes in the plant community like golden buffalo bean (*Thermopsis rhombifolia*) and purple milkvetch (*Astragalus agrestis*). In this microsite, *Dalea* seedlings had advanced development (stage) and reached greater height. *Melilotus* mortality rate over the first winter was associated with litter removal and a high cover of plains rough fescue (*Festuca hallii*), northern wheatgrass (*Elymus lanceolatus*), and American vetch (*Vicia americana*).

7.5.2.5 Central Parkland Tame Site (CP-T)

Germination during the first year at the CP-T site was affected by defoliation ($P < 0.001$) and differed among species ($P < 0.001$) (Table 7.8). Defoliated plots had markedly lower first-year germination at 21.6 ± 1.7 % when compared to non-defoliated plots at 30.1 ± 1.7 % (Table 7.9). *Medicago* and *Trifolium* had the highest germination at 42.8 and 36.7 ± 2.7 %, respectively, *Dalea* had an intermediate germination rate. In contrast, *Astragalus*, *Melilotus*, and *Vicia* had lower germination ranging from 12.7 to 18.3 ± 2.7 % (Table 7.9). Germination the following year differed by species ($P = 0.001$), where *Astragalus* had the highest germination and *Medicago* the lowest (Table 7.9). Recruitment in the second year did not differ among any factors. Germination and recruitment differed by species in the third year ($P \leq 0.021$), and germination further differed with the interaction of litter reduced and defoliation ($P = 0.046$) (Table 7.8), where overall germination was highest in plots defoliated with litter removal 2014 while no germination occurred in plots that had ambient litter and defoliation (Table 7.9). Most new germination was attributed to *Astragalus* at 1.4 ± 0.3 %, while no new germination occurred from *Dalea* or *Melilotus*. *Medicago* had the highest third year recruitment at 3.6 ± 0.9 %, indicating improved survival from previous years, while no recruits from *Melilotus* were detected (Table 7.9).

Mortality between the first and second year at the CP-T site differed by species ($P < 0.001$), and species interacted with both litter ($P < 0.001$) and defoliation ($P = 0.001$) (Table 7.10). Mortality rates were highest for *Trifolium* at 97.1 ± 4.9 % and lowest for *Vicia* and *Melilotus* at 86.8 to 88.8 ± 5.0 % (Table 7.11). *Melilotus* had higher mortality when there was ambient litter at 96.7 ± 7.4 % compared to 81.0 ± 6.9 % when litter was removed. Although nonsignificant, there was a trend for *Vicia* to have higher mortality when litter was removed at 91.1 ± 7.4 % compared to 82.5 ± 6.9 % when there was ambient litter. Defoliation in the CP-T site resulted in lower mortality for *Astragalus* and *Vicia*, but not any other legume species (Table 7.11).

Seedling height during the first year in the CP-T site differed by species, and species interacted with litter removal ($P < 0.001$) (Table 7.12). *Melilotus* and *Vicia* were the tallest seedlings at 7.17 ± 0.65 cm and 6.56 ± 0.63 cm, respectively, while *Trifolium* seedlings only achieved 0.74 ± 0.58 cm (Table 7.13). *Melilotus* seedlings were significantly taller (by 2-fold) when litter was removed. At the CP-T site first year seedlings typically advanced to at least growth stage 3, with the exception of *Trifolium* (Table E.2 and Table 7.13). After the second growing season (2015), significant differences in height and growth stage were evident among species ($P = 0.001$); however, due to small sample sizes numerous coefficients were dropped from mixed models (Table 7.12). *Melilotus* was the tallest species and was one of the few to achieve flowering and seed production (Table 7.13)

Mixed models used to link germination rates to overstory characteristics in the CP-T site found first year germination was affected for *Astragalus, Melilotus,* and *Trifolium* (Table 7.17). *Astragalus* germination was negatively influenced by bare soil exposure and litter cover ($P \leq 0.023$). *Melilotus* was positively associated with introduced forage cover and negatively associated with native grass cover ($P \leq 0.025$). *Trifolium* germination was positively associated with introduced forage cover, bare soil, and litter depth, while native grass cover was negatively associated with *Trifolium* germination ($P \leq 0.029$).

NMDS ordination of plant community composition at the tame Central Parkland (CP-T) location in 2014 (stress = 0.23, dimensions = 2, distance = Bray-Curtis) showed that the treatments did not cause significant shift in plant communities with the treatments, and there was no relationship with microclimate. Germination of *Melilotus* and *Vicia* were associated with plots with higher species richness and diversity attributed to native and introduced forbs, and their germination was negatively associated with introduced grass cover – primarily that of smooth brome (*Bromus inermis*).

7.6 Discussion

7.6.1 Germination Trial

Legume species planted in perennial grasslands exhibited unique responses regarding the probability of germinating and emerging, as well as surviving over time, with defoliation and litter removal further influencing these responses. Overall *in-situ* field observations of germination and recruitment did not mimic rates of germination observed during germination tests on moist filter paper; however, we can conclude that the addition of glue to facilitate uniform planting was an unlikely inhibitor of germination in the field based on lab tests. There were a handful of interactions between species and glue treatment that indicate the forage legumes *Medicago* and *Trifolium* could have experienced greater degradation due to the glue coating. In the soil where seeds are more moisture limited, it is possible that glue was less likely to contribute to pathogens degrading the seed. Germination in the field was more likely limited by environmental factors including moisture and light availability, which in turn, were manipulated by initially defoliating and removing litter from plots. Germination tests showed that seed used for *Dalea, Medicago*, and *Melilotus* had good germination potential with very little dormant ('hard') or degraded seed remaining, while *Vicia* had the highest proportion of seeds remaining dormant. *Vicia* and *Astragalus* had a similar probability of germinating, however *Astragalus* had the highest probability of seeds degrading but also retained about ~9% hard seed at the end of the trial.

7.6.2 Field Trial

7.6.2.1 Microenvironment

Litter removal reduced standing and fallen litter, increased bare soil exposure, improved the light available to seedlings, and warmed the soil – conditions expected to improve recruitment of legumes (Barret and Silander 1992; Davis and Pelsor 2001; Groya and Sheaffer 1981; Kunelius and Campbell 1984). Native grassland communities in the DMG and CP responded significantly and differently to litter removal, with total native grass cover decreasing in the DMG with litter removal and plant species richness increasing, while the native CP site exhibited an opposing response with improved native grass cover and decreased richness. In contrast, native grass cover in the CP increased with litter removal and was attributed to increases in *Festuca hallii* and *Elymus lanceolatus*, which are rhizomatous decreasers in fescue prairie. Litter reduction likely stimulated tillering (Willms et al. 1986; Deutsch et al. 2010b), increasing cover of fescue grassland decreasers. Tame grassland communities were generally more resistant to compositional change when litter was removed, in the Parkland Deutsch (2010b) found that tillering of forages [which could translate into cover in our case] was unaffected by litter removal or addition. Changes in the microenvironment and competitive vegetation were linked to germination and recruitment of legume seedlings. When litter removal influenced overall germination (e.g. DMG-T year 1 and CP-N year 3) it had a positive effect, however this effect was seldom significant. During the first year litter manipulation had no influence on seedling development (height or stage). For native DMG ambient litter was associated with higher overwinter mortality.

Litter serves important ecological functions in grasslands, including building soil organic matter, increasing water infiltration and preventing run-off, shading the soil surface and thereby reducing evaporation (Facelli and Pickett 1991; Naeth et al. 1991), and is therefore a key indicator of rangeland health (Adams et al. 2005). Litter also plays a significant role in seed bank formation (Chapters 5 and 6; Willms and Quinton) and seedling establishment (Jensen and Gutekunst 2003; Loydi et al. 2013). Litter was expected to influence soil moisture and influence the recruitment of legumes via this mechanism (Loydi et al. 2013). However, litter removal was not associated with differences in soil moisture in during the establishment year (2014), this could be attributed to high precipitation (Fig. 7.3). The following year carry over of reduced litter treatment was associated with soil moisture differences in the DMG, this could have been attributed to the spring drought (Fig. 7.3). Abundant litter can shield seeds from granivores like rodents (Reed et al. 2006), evidence of granivory was observed but inadequate data was acquired to link this to treatments.

In the current study, defoliation was expected to improve germination and recruitment by reducing the amount of light intercepted by competitive established vegetation (Williams et al. 2007). Clipping biomass and/or removing litter from plots resulted in thinner litter layers at all sites, likely by reducing standing litter and biomass that could become fallen litter but did not typically alter bare soil exposure. Vegetation height was consistently reduced and this resulted in greater light availability to seedlings, which in turn, increased soil surface temperatures. Defoliation during the first year effectively reduced total native grass and introduced forage cover at the native and tame sites, respectively, within the DMG natural subregion, while foliar cover of dominant vegetation was resistant in the CP. There were instances where defoliation had negative effects resulting in shorter statured seedlings at the native DMG site. Ultimately, the summer defoliation treatments did not influence mortality in this experiment, though other experiments have found that defoliation can reduce seedling mortality when seeded into established grasslands (Williams et al. 2007). Manipulation of litter and defoliation frequently interacted possibly because both these factors influence (independently and via interactions) microsite environmental conditions, including photo flux density, bare ground, soil surface temperature. Soil moisture differences may be obscured by the variability over time between sites.

7.6.2.2 Legume Species

Strong differences among legume species germination, survival and recruitment were exhibited at all sites. This was likely attributed to a species biology, competitiveness, and tolerance of stresses imposed by treatments. Legume species classified a native, agronomic, and escaped agronomics also behaved dissimilar within general functional classifications.

Despite its importance in native prairie grassland, low establishment of *Dalea* was observed. Low *Dalea purpurea* recruitment in competitive forage swards has been demonstrated in other studies (Mischolz et al. 2013). For native species like *Dalea purpurea*, the importance of using locally sourced native seeds has been demonstrated to improve the success of establishment resulting from genetic differences in populations (Gustafson et al. 2002; Gustafson et al. 2005; Stewart 2006). Our seed source was from BrettYoung™ and was intended to provide *Dalea* seed in restoration and reclamation efforts and was unlikely to be locally sourced. Thus, these results may have implications for restoration and reclamation efforts in the province attempting to establish native legumes in disturbed grasslands. Different seed sources of *Dalea* can also differ in their reproductive potential (Gustafson et al. 2002). In the current study, with over three years of monitoring established *Dalea* plants never entered reproductive life stages. Lauenroth and Alder (2008) described the demographics of *Dalea purpurea* in southern (Kansas) Mixedgrass prairie inside permanent plots, where the probability of first year survival was 29.6%, life expectancy was 1.37 years, and the maximum life-span was 7 years. Similarly, we found high mortality for *Dalea* following the first year, leading to very short-lived seedlings. In native Dry Mixedgrass prairie, higher mortality rates were observed when the ambient litter layer was present. Davis and Pelsor (2001) found that *Dalea purpurea* establishment benefited from disturbance treatments that removed competitive biomass (weeds). In established native plant communities *Dalea* also exhibited better fitness (height and development) in plots occupied by native forb increasers (e.g. *Sphaeralcea coccinea, Achillea millefolium*) including native legumes. Metrics of *Dalea* recruitment and fitness were associated with *Astragalus agrestis* in both natural regions and *Thermopsis rhombifolia* in CP-N. This is somewhat contrary to theory that suggests legumes are more competitive in nitrogen limited environments and establishment can be inhibited by high soil nitrogen (Aydin and Uzun 2005) or other legumes (Turnbull et al. 2005). Perhaps *Dalea* had similar niche requirements (i.e. ratios of soil resources (Tilman 1985)) or benefited from the relationship biochemically, note that taller seedlings often occurred in association with established native legumes and establishing seedlings could have been responding to the available nitrogen.

In native grasslands, *Vicia* continued to emerge during the following years (2015 to 2016). *Vicia* had the largest seeds and based on germination tests, we expected *Vicia* to have the highest dormancy rate. Under field conditions, especially during 2015 with below average spring moisture (Figure 7.3), available soil moisture for seed imbibition could have contributed to low emergence and high seed dormancy. Thompson et al. (1993) described a closely related species common to woodlands, *Vicia cracca*, as having short-lived seeds that could account for limited germination in later years. *Vicia* emergence in native grasslands was associated with competitive vegetation attributes. In DMG-N, *Vicia* emergence was positively associated with Shannon's diversity and negatively associated with total species richness and community evenness; while emergence was negatively associated with native forb cover in CP-N. Species richness of established communities has been linked to resistance of invasion and the recruitment of novel species (Tillman 1997), which could be inhibitory to *Vicia* establishment in native grassland. Although our experiment was not designed to measure seed predation by granivores, birds, or degradation by soil microfauna, we suspect *Vicia*'s large seeds could have made it more vulnerable, especially if soil movement or precipitation exposed the seed. At one site (CP-N) within a few plots (~3) granivory by ground squirrels on *Vicia* seed was detected, where toothpicks were plucked from the ground and seeds had been visibly consumed. In prairie ecosystems, granivory is selective and larger seeds are more susceptible, especially overwinter, which can influence the composition of plant communities (Howe and Brown 2001). Under field conditions, larger seeds are less likely to enter the soil seed bank compared to smaller seeds that are more easily buried, enter cracks in soil, or become enclosed in surface litter, thereby placing the seed below the soil surface and possibly prolonging seed persistence (Thompson et al. 1993). For a palatable legume decreaser abundant in both native and tame grasslands in the Northern Great Plains, this species is understudied, and its ecology is largely unknown relative to other native legumes (Gunn 1965; Gunn 1970). More research is required as this species is likely desirable for restoration and reclamation.

Medicago had lower mortality rates in tame grasslands and individuals persisted resulting in relatively higher *Medicago* recruitment during 2015 and 2016. Within native grasslands, *Medicago* mortality was very high and followed by limited to no recruitment the following years. The microenvironment or plant community in native grassland may be deleterious to *Medicago* seedlings and likely suppresses individuals that attempt to establish. Mixed models showed that *Medicago* germination in native DMG was improved when there was higher introduced forage cover. Bagavathiannan et al. (2011) also found low establishment of *Medicago* in established grassland, but postulated low densities were compensated for by the production of shoots later on. Inter-seeding *Medicago* into mixed grasslands may be a desirable range transformation that can improve plant community biomass, forage quality (including native plants), soil fertility, and can persist long-term [decades] (Mortenson et al. 2005). Although *Medicago* is introduced it has limited invasibility in native grassland, however *Medicago* can become problematic to natural habitats along transportation corridors (Bagavathiannan et al. 2010; Hansen and Clavenger 2005). In tame grasslands in the current study, *Medicago* germination was also positively associated with introduced forage cover in communities with high evenness, while higher litter loads supressed germination. While not tested here, it is possible that seedling establishment of *Medicago* may benefit from having neighbors of the same species, similar to that found by Wagg et al. (2015) for *Trifolium pratense*, perhaps through soil conditioning via biochemical cues or microbiologicals required for N fixation.

Trifolium was a forage legume expected to benefit from our treatments, which reduced litter and the competitiveness of established vegetation (Barret and Silander 1992). During the first year germination was about 17.8 to 36.7 % depending on sites, and contradicting our original hypothesis, *Trifolium* emergence was 3-fold higher in non-defoliated plots in tame DMG grassland. However, mortality rates over winter were very high eliminating most *Trifolium* seedlings, with limited germination after 2014. In tame pasture, *T. repens* tends to increase with grazing pressure and its short stature and stoloniferous growth habit make this species grazing tolerant (Turkington and Burdon 1983). We expected *Trifolium* establishment from treatments that created vegetation structure and the reduced litter layers typically observed in moderately to heavily grazed tame pasture. In native DMG, *Trifolium* emergence was positively associated cover from native grasses, forbs and soil crust components like lichen and *Selaginella densa* and had a strong negative association with bare soil. This suggests *Trifolium* seedlings at the microclimate level required some refugia and areas of bare soil in native DMG were unsuitable for germinants. Emergence of *Trifolium* in CP-T differed, positively associated with bare soil, litter depth and forage cover, this could be attributed to higher moisture in the Parkland. In Chapter 5, we found a germinable seed bank dense with *Trifolium* spp. (both *T. hybridum* and *T. repens*). This study indicates that germination of *Trifolium* can occur in competitive grassland, but it does not successfully recruit. This result could have been exacerbated by the spring drought in 2015, as *Trifolium* seedlings and clones can become damaged by oxidative stress when water is limited (Vaseya et al. 2012). Low to little germination of *Trifolium* in later years could be caused by seed degradation (Russi et al. 1992), failure to become incorporated into the artificial soil seed bank, and other loses (i.e. predation).

Astragalus continued to germinate through the second and third year at all locations, while other species demonstrated little to no emergence. This has been observed in other research that describes *A. cicer* as a legume with high seed dormancy and slow establishment (Acharya et al. 2006). Scarification is often recommended to improve the germination potential of *A. cicer* sown into pastures (Miklas et al. 1987), and freeze-thaw cycles can improve germination of this species as it can be beneficial over mechanical scarification which damages embryos (Acharya et al. 1993). Freeze-thaw cycles during spring 2015 and 2016 likely aided in further germination of *Astragalus* from the planted seed bank. While germination was consistently higher for *Astragalus* during the following years, it was also much higher in native grasslands than tame grasslands. *Astragalus* was selected as a potentially invasive legume in native grasslands and its previously documented affinity for emergence in native prairie is concerning (Carlyle, unpublished data). In native DMG prairie, *Astragalus* also had a relatively low mortality rate, comparable to native *Vicia*. These features could aid in a slow, steady invasion in native grassland. This differs from *Melilotus* that relies on high propagule pressure and its biennial life cycle to quickly invade suitable habitats.

Melilotus was expected to establish in environments where it exhibits invasiveness, like native grassland (Van Riper and Larson 2009; Wolf et al. 2008); however, poor establishment was observed in later years despite the relatively advanced seedling development to stage 3 (i.e. at least 2 or more leaves) during the first year. Unlike the other legumes examined, *Melilotus* is a biennial (occasionally triennial), and therefore must complete its life cycle relatively quickly in order to ensure propagation of the population. While not tested directly here, high seedling mortality due to winter weather or drought the following spring (Fig. 7.3) could have exacerbated the lack of development (i.e. roots) in this species.

Compared to other legumes, *Melilotus* exhibited stronger associations with microsite, shaped by plant community characteristics. In native Dry Mixedgrass prairie, *Melilotus* germination was more likely when there was higher native grass cover, plant community evenness, and overall community richness, while germination rates were reduced when litter was high in cover and thick in depth. *Selaginella densa* (which is a major component of DMG biological crusts) inhibited germination as well. Van Riper and Larson (2009) found that *Melilotus* is a relatively poor competitor in established native mixed grassland [*Pascopyrum smithii* dominated] when compared to plant communities with sparse vegetation, and *Melilotus* cover was positively associated with native cover when vegetation was sparse. In native fescue grassland, *Melilotus* was also sensitive to microsite, but had some divergent responses. Germination was positively associated with diversity and litter depth, while native forbs, species richness, evenness, bare ground, litter cover, and lichen cover were negatively related to germination. In both the native sites, biological crust components emerged as important ground cover characteristics that inhibited *Melilotus* germination. Biological crusts are a functionally important community layer in native grasslands that can form a barrier for seed entry into the seed bank, influencing seedling recruitment into the plant community (Johansen 1993; Li et al. 2005), and it is likely that soil crusts can also form a barrier to emergent seedlings as well. Conversely, Delach and Kimmerer (2002) found that moss turfs of *Polytichum piliferum* (occurs in Alberta's prairie soil crusts, see Chapter 6) benefited the establishment and survival period of *Melilotus* seedlings on disturbance, likely through the creation of a cooler microclimate. Litter depth and cover had different roles in suppressing or enhancing *Melilotus* germination in DMG and CP native grasslands. Abundant ground cover, which can be altered through disturbance (including grazing management), is an important component of native grassland communities regulating *Melilotus* establishment and subsequent invasion. Components of community diversity in native grasslands also influenced *Melilotus* germination. Seedling recruitment was much greater when plant community evenness was high in DMG but much lower when evenness was high in CP, discussions of plant community evenness and their resistance to invasion are disparate and the findings are often species specific (Mattingly et al. 2007; Wilsey and Polley 202) and the identity of dominant plants may be more important (Emery and Gross 2007). In tame grasslands, *Melilotus* germination was less responsive to ground cover characteristics where germination was reduced with high introduced ruderal forb cover in DMG-T, and improved by forage cover in CP-T, while being reduced by native grasses in CP-T.

7.6.3 Comments on Legume Recruitment Overall

During the trial, high seedling mortality rates were observed. Legume species exhibited different mortality rates at all sites except for the native CP fescue prairie site. In native DMG, introduced *Medicago*, *Melilotus*, and *Trifolium* had the highest mortality rates (>99 %), while 5 to 17 % of *Astragalus*, *Dalea*, and *Vicia* were retained. In tame grasslands, greater retention of *Medicago* (8 to 12%) and *Melilotus* (11%) was evident in the CP. Given the critical nature of first year establishment, our sample size of legume seeds (40 per plot) may have been too small to accurately assess subsequent losses in year 2. For invasive species like *Melilotus* with a biennial life strategy, high propagule densities are likely required for establishment of new populations.

Legume seeds were planted shallow to resemble seeds recently incorporated into a soil surface seed bank, however seeds often require good contact with soil to ensure imbibition and some species could have benefited from deeper seeding depths (Townsend 1972). The microenvironment created by treatments, such as litter removal and defoliation could also have created a niche that enabled other persistent seeds to emerge or created a place for species that form transient seed banks (e.g. grasses) to emerge from ongoing seed rain (Bullock et al. 1994). Unfortunately, we did not record the emergence of *in-situ* recruitment of emergent forbs and grasses, this would have provided additional insight into competitive influences on legume seedling recruitment when niches are created. Here, time was a limiting factor.

Legume seeds planted were also free of scarification or inoculant which could have aided in release from dormancy (Acharya 2006; Baskin et al. 2000) and improved nodulation, increasing plant phytomass (Tlusty et al. 2004), which could have aided in plant development and competitiveness during recruitment. Further, the microsymbiont required for inoculation can exhibit species specificity and these relationships are not fully understood (Graham 2005), but we can assume when a novel legume species is introduced to a community its symbionts may be absent. Native *Dalea purpurea* can become nodulated by a variety of commercially available strains and forms associations with multiple *Rhizobium* species (Graham 2005; Tlusty et al. 2004), while *Vicia* may require more specificity (Graham 2005). Note that metrics of *Dalea* recruitment were associated with *Astragalus agrestis* at both native grasslands, this could indicate a mutual microsymbiont. In previously cultivated pastures (and perhaps less diverse pastures dominated by tame forages), microsymbionts required for nodulation of native legumes could be limited and vary with time since last disturbance (lower with recent disturbance and old-fields) (Larson and Siemann 1998). *Trifolium* exhibited low recruitment and no *Trifolium* was observed within the established vegetation, recruitment can be improved with suitable *Rhizobia* (Hale et al. 1979); thus, suitability of microsymbionts likely limited establishment of agronomic species as well. Further, microsymbiont promiscuity of invasive legumes can aid in establishment when introduced into novel communities (Klock et al. 2015); however, it is unclear if this mechanism benefited our potential invaders in this study. Further, successful recruitment of legumes, especially novel species, could have been further limited by mycorrhizal symbionts in the soil. However, an experiment that reduced mycorrhizal populations with fungicide found that native *Dalea purpurea* was unaffected by the treatment (Hartnett et al. 1994). Overall, plant-soil feed back influences the establishment and persistence of species and legacy effects from the plant composition of previous states can influence intra and interspecific competition (Voorde et al. 2011). However, we did not examine soil characteristics at a plot level and initially did not recognize the potential importance of microbial and fungal communities in regulating legume survival and competitiveness.

7.6.4 Comments on Methods

It is important to note that observed legume germination, recruitment, survival, etc. in the field, may have been influenced by the frequency of observations and possibly environmental influences not measured here (i.e. granivory, frost, herbivory, etc.). Evidence of granivory from rodents and seedling herbivory from arthropods was observed, but not accounted for. Post-winter, both tame grasslands had evidence of new mole activity (i.e. soil deposited on the surface from mole hills and tunnels) which indicated an active subnivean zone which could have lead to opportunistic foraging of seeds. In a seed addition [including legumes *Medicago sativa* and *Melilotus officinalis*] experiment by MacDougall and Wilson (2007), where 1.2 million seeds were broadcast, poor recruitment resulted in the termination of the study at 12 weeks and this was attributed to high herbivory [the herbivore was largely unknown, but small mammals were suspected]. MacDougall and Wilson (2007) also suspected inadequate propagule pressure may have played a role, for context, in our experiment only ~15,360 seeds were planted. In addition, we did not make an effort to check for dormant seeds. With forethought during experimental establishment, our understanding of seed dormancy could have been improved by setting up a seed burial study (Van Assche et al. 2003) at each experimental field site. It is also possible that our seeding rate may have been too low to observe true demographic patterns, as many plants require high propagule densities to ensure the successful establishment and survival of a population through to the reproductive life phases of the next generation (Lockwood et al. 2005).

7.7 Conclusions and Management Implications

A primary conclusion of this artificial seed bank study is that the mortality of legume seedlings in established competitive pasture vegetation is very high, and the recruitment of many individuals is required to ensure that a single individual survives more than one annual growth cycle, let alone becoming mature and reaching a reproductive age itself. This is important to consider when interpreting seed bank composition data, particularly where recruitment in a greenhouse lacks environmental stresses created at a microsite level and dominant competitive native grasses or perennial forages. Nevertheless, this study provided substantial insight into the seedling recruitment demographics of a variety of legume species across a range of growing environments. We found legumes have strong species-specific responses to the microenvironment and competitive stresses, and some legumes like *Dalea* and *Medicago* demonstrated positive relationships during early establishment with other established legumes. Legumes like *Trifolium* and *Melilotus* were influenced by community structure and ground cover in stressful microclimates like DMG.

7.8 Literature Cited

Acharya, S.N., Kastelic, J.P., Beauchemin, K.A., and Messenger, D.F. 2006. A review of research progress on cicer milkvetch (*Astragalus cicer* L.). Canadian Journal of Plant Science **86**(1):49-62.

Acharya, S.N., Kokko, E.G. and Fraser, J. 1993. Storage duration and freeze-thaw effects on germination and emergence of cicer milkvetch (*Astragalus cicer*) seeds. Journal of Seed Technology **17**(2):9-21.

Adams, B. W., Ehlert, G., Stone, C., Lawrence, D., Alexander, M., Willoughby, M., Hincz, C., Moisey, D., Burkinshaw, A., Carlson, J. and France, K. 2005. Rangeland health assessment for grassland, forest and tame pasture. Public Lands and Forests Division, Alberta Sustainable Resource Development.

Alberta Agriculture and Forestry. 2016. AgroClimatic Information Service. Accessed September 2017 for the Verger, AB and Kinsella, AB weather stations. http://agriculture.alberta.ca/acis/alberta-weather-data-viewer.jsp

Aydin, I., and Uzun, F.2005. Nitrogen and Phosphorus fertilization of rangelands affects yield, forage quality and the botanical composition. European Journal of Agronomy **23**(1):8-14.

Bagavathiannan, M.V., Gulden, R.H., Begg, G.S. and Van Acker, R.C. 2010. The demography of feral alfalfa (*Medicago sativa* L.) populations occurring in roadside habitats in Southern Manitoba, Canada: implications for novel trait confinement. Environmental Science and Pollution Research **17**(8):1448-1459.

Bagavathiannan, M.V., Gulden, R.H. and Van Acker, R.C. 2011. The ability of alfalfa (*Medicago sativa*) to establish in a seminatural habitat under different seed dispersal times and disturbance. Weed Science **59**(3):314-320.

Bagavathiannan, M.V. and Van Acker, R.C. 2009. The biology and ecology of feral alfalfa (*Medicago sativa* L.) and its implications for novel trait confinement in North America. Critical Reviews in Plant Sciences **28**(1-2):69-87.

Barret, J.P. and Silander Jr., J.A. 1992. Seedling recruitment limitation in white clover (*Trifolium repens*; Leguminosae). American Journal of Botany **79**(6):643-649.

Baskin. J.M., Baskin, C.C., and Li, X. 2000. Taxonomy, anatomy and evolution of physical dormancy in seeds. Plant Species Biology **15**:139-152.

Bates, D., Maechler, M., Bolker, B., and Walker, S. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software **67**(1):1-48. doi:10.18637/jss.v067.i01.

Booth, B.D., and Swanton, C.J. 2002. 50th anniversary—invited article: Assembly theory applied to weed communities. Weed Science **50**:2-13.

Bork, E. W., Grekul, C. W., and De Bruijn, S. L. 2007. Extended pasture forage sward responses to Canada thistle (*Cirsium arvense*) control using herbicides and fertilization. Crop Protection **26**(10):1546-1555.

Briske, D.D. 1996. Strategies of plant survival in grazed systems: a functional interpretation. The Ecology and Management of Grazing Systems. pp 37-67.

Bullock, J.M., Hill, B.C., Dale, M.P. and Silvertown, J. 1994. An experimental study of the effects of sheep grazing on vegetation change in a species-poor grassland and the role of seedling recruitment into gaps. Journal of Applied Ecology **31**(3):493-507.

Caswell, H. 1989. Matrix population models: construction, analysis, and interpretation. Sinauer Associates Inc., Sunderland, Massachusetts.

Clements, D.R., Krannitz, P G., and Gillespie, S.M. 2007. Seed bank responses to grazing history by invasive and native plant species in a semi-desert shrub-steppe environment. Northwest Science **81**(1):37-49.

Coffin, D.P. and Laurenroth, W.K. 1989. Spatial and temporal variation in the seed bank of a semiarid grassland. American Journal of Botany **76**(1):53-58.

Davis, M.A. and Pelsor, M. 2001. Experimental support for a resource-based mechanistic model of invasibility. Ecology Letters **4**(5):421-428.

De Mendiburu, F. 2017. agricolae: Statistical Procedures for Agricultural Research. R package version 1.2-6.
https://CRAN.R-project.org/package=agricolae

De Wet, J.M. and Harlan, J.R. 1975. Weeds and domesticates: evolution in the man-made habitat. Economic Botany **29**(2):99-108.
Delach, A. and Kimmerer, R.W. 2002. The effect of *Polytrichum piliferum* on seed germination and establishment on iron mine tailings in New York. The Bryologist **105**(2):249-255.

Deutsch, E. S., Bork, E. W. and Willms, W. D. 2010a. Separation of grassland litter and ecosite influences on seasonal soil moisture and plant growth dynamics. Plant Ecology **209**(1):135-145.

Deutsch, E. S., Bork, E. W. and Willms, W. D. 2010b. Soil moisture and plant growth responses to litter and defoliation impacts in Parkland grasslands. Agriculture, Ecosystems & Environment **135**(1):1-9.

Donohue, K., Dorn, L., Griffith, C., Kim, E., Aguilera, A., Polisetty, C.R. and Schmitt, J. 2005. The evolutionary ecology of seed germination of *Arabidopsis thaliana*: variable natural selection on germination timing. Evolution **59**(4):758-770.

Emery, S.M. and Gross, K.L. 2007. Dominant species identity, not community evenness, regulates invasion in experimental grassland plant communities. Ecology **88**(4):954-964.

Eriksson, A. and Eriksson, O. 1997. Seedling recruitment in semi-natural pastures: the effects of disturbance, seed size, phenology and seed bank. Nordic Journal of Botany **17**(5):469-480.

Fick, G.W, and S.C. Mueller. 1989. Alfalfa quality, maturity, and mean stage of development. Cornell Coop. Extension Information Bulletin 217. Cornell University, Ithaca, NY.

Freedman, B. 2010. *Environmental Science: A Canadian Prospective.* Pearson Canada Inc., Toronto: pp 60-61.

Gardiner, D.T. 1993. Revegetation status of reclaimed abandoned mined land in western North Dakota. Arid Land Research and Management **7**(1):79-84.

Gioria, M., Jarosik, V., and Pysek, P. 2014. Impact of invasions by alien plants on soil seed bank communities: emerging patterns. Evolution and Systematics **16**:132-142.

Graham, P.H. 2005. Practices and issues in the inoculation of prairie legumes used in revegetation and restoration. Ecological Restoration **23**(3):187-195.

Grekul, C.W. and Bork, E.W. 2007. Fertilization augments Canada thistle (*Cirsium arvense* L. Scop) control in temperate pastures with herbicides. Crop Protection **26**(4):668-676.

Groya, F.L. and Sheaffer, C.C. 1981. Establishment of sod-seeded alfalfa at various levels of soil moisture and grass competition. Agronomy Journal **73**(3):560-565.

Gunn, C.R. 1965. The *Vicia americana* complex (Leguminosae). Retrospective Theses and Dissertations, Iowa State University. Paper 4086.

Gunn, C.R. 1970. Genus *Vicia* with notes about tribe Vicieae (Fabaceae) in Mexico and Central America. Technical Bulletin 1601, United States Department of Agriculture.

Gustafson, D.J., Gibson, D.J. and Nickrent, D.L. 2002. Genetic diversity and competitive abilities of *Dalea purpurea* (Fabaceae) from remnant and restored grasslands. International Journal of Plant Sciences **163**(6):979-990.

Gustafson, D.J., Gibson, D.J. and Nickrent, D.L. 2005. Using local seeds in prairie restoration—data support the paradigm. Native Plants Journal **6**(1):25-28.

Hale, C.N., Lowther, W.L. and Lloyd, J.M. 1979. Effect of inoculant formulation on survival of *Rhizobium trifolii* and the establishment of oversown white clover (*Trifolium repens*). New Zealand Journal of Experimental Agriculture **7**(3):311-314.

Hansen, M.J. and Clevenger, A.P. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation **125**(2):249-259.

Harker, K.N., Baron, V.S., Chanasyk, D.S., Naeth, M.A., and Stevenson, F.C. 2000. Grazing intensity effects on weed populations in annual and perennial pasture systems. Weed Science **48**(2):231-238.

Hartnett, D.C., Samenus, R.J., Fischer, L.E. and Hetrick, B.A.D. 1994. Plant demographic responses to mycorrhizal symbiosis in tallgrass prairie. Oecologia **99**(1-2):21-26.

Howe, H.F. and Brown, J.S. 2001. The ghost of granivory past. Ecology Letters **4**(4):371-378.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological monographs **54**(2):187-211.

Jensen, K. and Gutekunst, K. 2003. Effects of litter on establishment of grassland plant species: the role of seed size and successional status. Basic and Applied Ecology **4**(6):579-587.

Johansen, J.R. 1993. Cryptogamic crusts of semiarid and arid lands of North America. Journal of Phycology **29**:140-147.

Johnston, A., Smoliak, S. and P.W. Stringer. 1969. Viable seed populations in Alberta prairie topsoils. Canadian Journal of Plant Science **49**(1):75-82.

Katepa-Mupondwa, F., Singh, A., Smith Jr, S.R. and McCaughey, W.P. 2002. Grazing tolerance of alfalfa (*Medicago* spp.) under continuous and rotational stocking systems in pure stands and in mixture with meadow bromegrass (*Bromus riparius* Rehm. syn. *B. biebersteinii* Roem & Schult). Canadian Journal of Plant Science **82**(2):337-347.

Klemow, K.M., and Raynal, D.J. 1981. Population ecology of *Melilotus alba* in a limestone quarry. The Journal of Ecology **69**(1):33-44.

Klock, M.M., Barrett, L.G., Thrall, P.H. and Harms, K.E. 2015. Host promiscuity in symbiont associations can influence exotic legume establishment and colonization of novel ranges. Diversity and Distributions **21**(10):1193-1203.

Kunelius, H.T. and Campbell, A.J. 1984. Performance of sod-seeded temperate legumes in grass dominant swards. Canadian Journal of Plant Science **64**(3):643-650.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2001. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on botanical composition. Canadian Journal of Plant Science **81**(4):673-683.

Larson, J.L. and Siemann, E. 1998. Legumes may be symbiont-limited during old-field succession. The American midland naturalist **140**(1):90-95.

Lauenroth, W.K. and Adler, P.B. 2008. Demography of perennial grassland plants: survival, life expectancy and life span. Journal of Ecology **96**(5):1023-1032.

Ledgard, S.F. and Steele, K.W. 1992. Biological nitrogen fixation in mixed legume/grass pastures. Plant and Soil **141**:137-153.

Li, X. Jia, X., Long, L., and Zerbe, S. 2005. Effects of biological soil crusts on seed bank, germination and establishment of two annual plant species in the Tengger Desert (N China). Plant and Soil **277**(1):375-385.

Lockwood, J.L., Cassey, P. and Blackburn, T. 2005. The role of propagule pressure in explaining species invasions. Trends in Ecology & Evolution **20**(5):.223-228.

Loydi, A., Eckstein, R.L., Otte, A. and Donath, T.W. 2013. Effects of litter on seedling establishment in natural and semi-natural grasslands: a meta-analysis. Journal of Ecology **101**(2):454-464.

Ma, M., Zhou, X., and Du, G. 2010. Role of soil seed bank along a disturbance gradient in an alpine meadow on the Tibet Plateau. Flora-Morphology, Distribution, Functional Ecology of Plants **205**(2):128-134.

MacDougall, A.S. and Wilson, S.D. 2007. Herbivory limits recruitment in an old-field seed addition experiment. Ecology **88**(5):1105-1111.

Mattingly, W.B., Hewlate, R. and Reynolds, H.L. 2007. Species evenness and invasion resistance of experimental grassland communities. Oikos **116**(7):1164-1170.

McClay, A. 2012. Revising Alberta's Provincial Weeds List: Experiences and Lessons Learned. Weeds Across Borders Conference, At Cancún, Quintana Roo, Mexico.

Miklas, P.N., Townsend, C.E. and Ladd, S.L. 1987. Seed coat anatomy and the scarification of cicer milkvetch seed. Crop Science **27**(4):766-772.

Miller, A.J., Bork, E.W., Hall, L.M., and Summers, B. 2015. Long-term forage dynamics in pastures sprayed with residual broadleaf herbicide: A test of legume recovery. Canadian Journal of Plant Science **95**(1):43-53.

Mischkolz, J.M., Schellenberg, M.P. and Lamb, E.G. 2013. Early productivity and crude protein content of establishing forage swards composed of combinations of native grass and legume species in mixed-grassland ecoregions. Canadian Journal of Plant Science **93**(3):445-454.

Mortenson, M.C., Schuman, G.E., Ingram, L.J., Nayigihugu, V. and Hess, B.W. 2005. Forage production and quality of a mixed-grass rangeland interseeded with *Medicago sativa* ssp. *falcata*. Rangeland Ecology & Management **58**(5):505-513.

Naeth, M.A. Bailey, A.W., Pluth, D.J., Chanasyk, D.S., and Hardin, R.T. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland. Journal of Range Management **44**(1):7-12.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, K., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson. G.L., Solymos, P., M., Stevens, M.H., Szoecs, E., and Wagner, H. 2017. vegan: Community ecology package. R package version 2.4-4. https://CRAN.R-project.org/package=veg

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Reed, A.W., Kaufman, G.A. and Kaufman, D.W. 2006. Effect of plant litter on seed predation in three prairie types. The American Midland Naturalist **155**(2):278-285.

Rees, M., 1996. Evolutionary ecology of seed dormancy and seed size. Philosophical Transactions: Biological Sciences **351**(1345):1299-1308.

Russi, L., Cocks, P.S. and Roberts, E.H. 1992. Hard-seededness and seed bank dynamics of six pasture legumes. Seed Science Research **2**(4):231-241.

Sanderson, M. A., Goslee, S. C., Klement, K. D., and Soder, K.J. 2007. Soil seed bank composition in pastures of diverse mixtures of temperate forages. Agronomy Journal **99**:1514-1520.

Semiadil, G., Barry, T.N., Muir, P.D. and Hodgson, J. 1995. Dietary preferences of sambar (*Cervus unicolor*) and red deer (*Cervus elaphus*) offered browse, forage legume and grass species. The Journal of Agricultural Science **125**(1):99-107.

Simmers, S.M., and Galatowitsch, S.M. 2010. Factors affecting revegetation of oil field access roads in semiarid grassland. Restoration Ecology **18**(s1):27-39.

Sleugh, B., Moore, K.J., George, J.R. and Brummer, E.C. 2000. Binary legume–grass mixtures improve forage yield, quality, and seasonal distribution. Agronomy Journal **92**(1):24-29.

Smith, S.R., Bouton, J.H. and Hoveland, C.S. 1988. Alfalfa persistence and regrowth potential under continuous grazing. Agronomy Journal **81**:960-965.

Smolenski, S.J., Kinghorn, A.D. and Balandrin, M.F. 1981. Toxic constituents of legume forage plants. Economic Botany **35**(3):321-355.

Spehn, E.M., Scherer-Lorenzen, M., Schmid, B., Hector, A., Caldeira, M.C., Dimitrakopoulos, P.G., Finn, J.A., Jumpponen, A., O'donnovan, G., Pereira, J.S. and Schulze, E.D. 2002. The role of legumes as a component of biodiversity in a cross-European study of grassland biomass nitrogen. Oikos **98**(2):205-218.

Stewart, W.L. 2006. The effects of remnant seed source size on plant performance in a prairie restoration. M. Sc. Thesis, Eastern Illinois University, Department of Biological Sciences, Charleston, Illinois.

Thompson, K., Band, S.R., and Hodgson, J.G. 1993. Seed size and shape predict persistence in soil. Functional Ecology **7**(2):236-241.

Tilman, D. 1985. The resource-ratio hypothesis of plant succession. The American Naturalist **125**(6):827-852.

Tilman, D. 1997. Community invasibility, recruitment limitation, and grassland biodiversity. Ecology **78**(1):81-92.

Tlusty, B., Grossman, J.M. and Graham, P.H. 2004. Selection of rhizobia for prairie legumes used in restoration and reconstruction programs in Minnesota. Canadian Journal of Microbiology **50**(11):977-983.

Townsend, C.E. 1972. Influence of seed size and depth of planting on seedling emergence of two milkvetch species. Agronomy Journal **64**(5):627-630.

Tracy, B.F., and Sanderson, M.A. 2000. Seedbank diversity in grazing lands of the Northeast United States. Journal of Range Management **53**(1):114-118.

Turnbull, L.A., Rahm, S., Baudois, O., Eichenberger-Glinz, S., Wacker, L. and Schmid, B. 2005. Experimental invasion by legumes reveals non-random assembly rules in grassland communities. Journal of Ecology **93**(6):1062-1070.

Turkington, R. and Burdon, J.J. 1983. The biology of Canadian weeds: 57. *Trifolium repens* L. Canadian Journal of Plant Science **63**(1):243-266.

Van Assche, J.A., Debucquoy, K.L. and Rommens, W.A. 2003. Seasonal cycles in the germination capacity of buried seeds of some Leguminosae (Fabaceae). New Phytologist **158**(2):315-323.

Van Riper, L.C. and Larson, D.L. 2009. Role of invasive *Melilotus officinalis* in two native plant communities. Plant Ecology **200**(1):129-139.

Van Riper, L.C., Larson, D.L. and Larson, J.L. 2010. Nitrogen-limitation and invasive sweetclover impacts vary between two Great Plains plant communities. Biological Invasions **12**(8):2735-2749.

Vaseva, I., Akiscan, Y., Simova-Stoilova, L., Kostadinova, A., Nenkova, R., Anders, I., Feller, U. and Demirevska, K. 2012. Antioxidant response to drought in red and white clover. Acta Physiologiae Plantarum **34**(5):1689-1699.

Van de Voorde, T.F., van der Putten, W.H. and Martijn Bezemer, T. 2011. Intra-and interspecific plant–soil interactions, soil legacies and priority effects during old-field succession. Journal of Ecology **99**(4):945-953.

Wagg, C., Boller, B., Schneider, S., Widmer, F. and van der Heijden, M.G. 2015. Intraspecific and intergenerational differences in plant–soil feedbacks. Oikos **124(**8):.994-1004.

White, S.R., Bork, E.W., Karst, J., and Cahill Jr., J.F. 2012. Similarity between grassland vegetation and seed bank shifts with altered precipitation and clipping, but not warming. Community Ecology **13**(2):129-136.

Williams, D.W., Jackson, L.L. and Smith, D.D. 2007. Effects of frequent mowing on survival and persistence of forbs seeded into a species-poor grassland. Restoration Ecology **15**(1):24-33.

Wilsey, B.J. and Polley, H.W. 2002. Reductions in grassland species evenness increase dicot seedling invasion and spittle bug infestation. Ecology Letters **5**(5):676-684.

Willms, W.D., Acharya, S.N. and Rode, L.M. 1995. Feasibility of using cattle to disperse cicer milkvetch (*Astragalus cicer* L.) seed in pastures. Canadian Journal of Animal Science **75**(1):173-175.

Willms, W.D., and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Willms, W.D., Smoliak, S. and Bailey, A.W. 1986. Herbage production following litter removal on Alberta native grasslands. Journal of Range Management **39**(6):536-540.

Wolf, J.J., Beatty, S.W., and Carey, G. 2008. Invasion by sweet clover (*Melilotus*) in montane grasslands, Rocky Mountain National Park. Annals of the Association of American Geographers **93**(3): 531-543.

Wolf, J.J., Beatty, S.W. and Seastedt, T.R. 2004. Soil characteristics of Rocky Mountain National Park grasslands invaded by *Melilotus officinalis* and *M. alba*. Journal of Biogeography **31**(3):415-424.

Woodcock, B.A., Savage, J., Bullock, J.M., Nowakowski, M., Orr, R., Tallowin, J.R.B. and Pywell, R.F. 2014. Enhancing floral resources for pollinators in productive agricultural grasslands. Biological Conservation **171**:44-51.

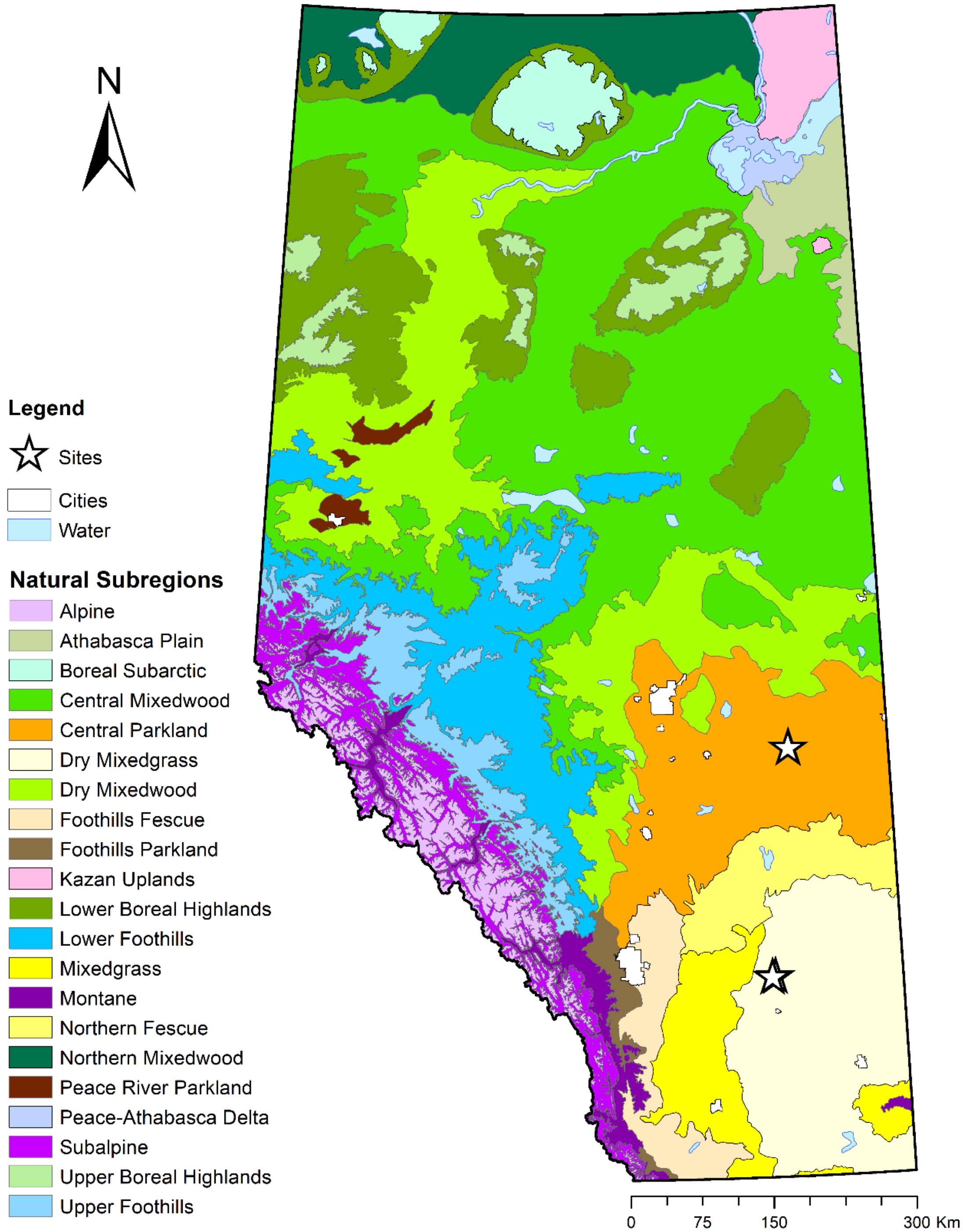


Figure 7.1. Map of study site locations within the Central Parkland and Dry Mixedgrass natural subregions within the province of Alberta, Canada. The Central Parkland locations were located within the Roy Berg Kinsella Research Ranch near Kinsella, Alberta, 140 km SE of Edmonton. Dry Mixedgrass sites were located at the Mattheis Research Ranch near Duchess, 150 km east of Calgary. Both Research Ranches are affiliated with the University of Alberta.

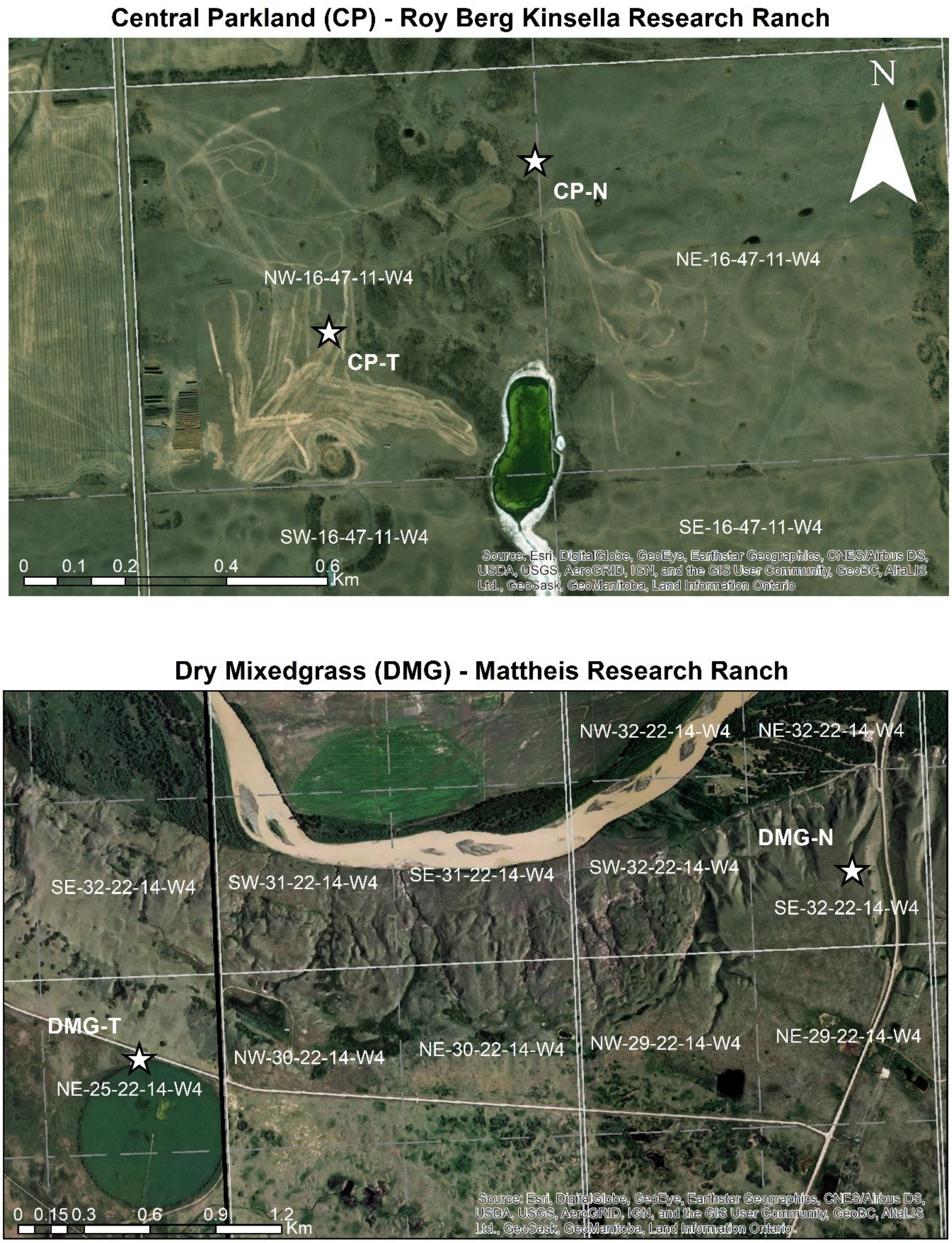


Figure 7.2. Map of the native (N) and tame (T) study site locations (white stars) within the Central Parkland (CP) and Dry Mixedgrass (DMG) at their respective research ranches.

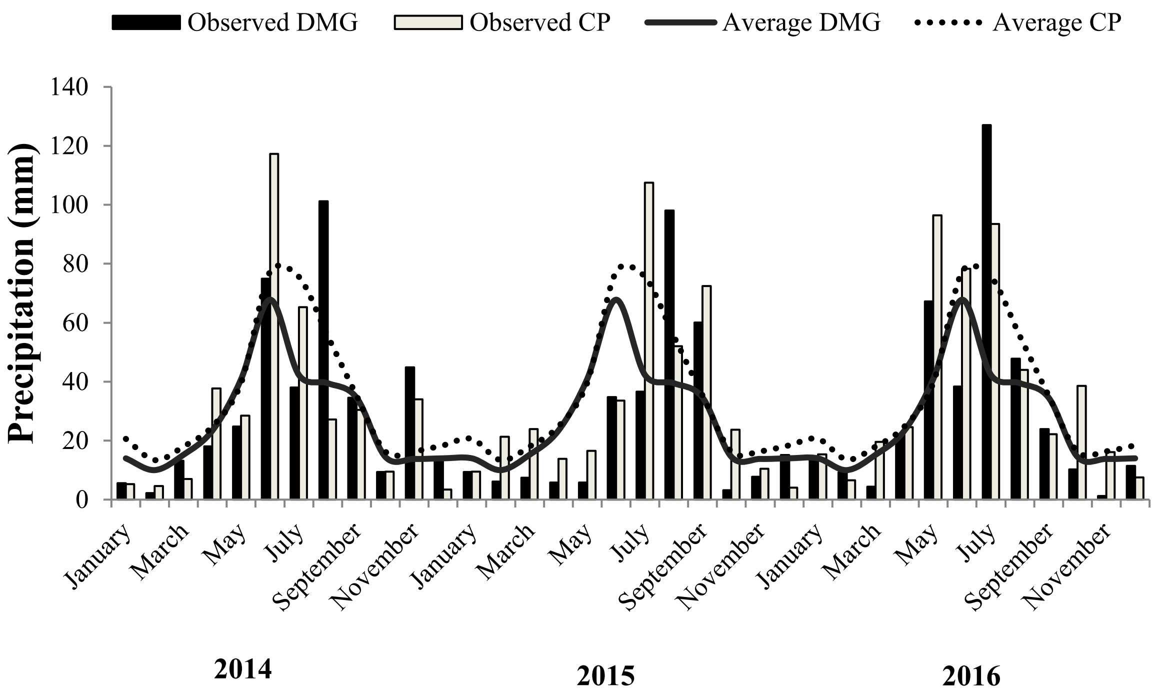


Figure 7.3. Observed and average monthly precipitation (mm) at the University of Alberta's Mattheis Research Ranch in the Dry Mixedgrass (DMG) prairie and Roy Berg Kinsella Research Ranch in the Central Parkland (CP) between 2014 and 2016 (Alberta Agriculture and Forestry, 2017).

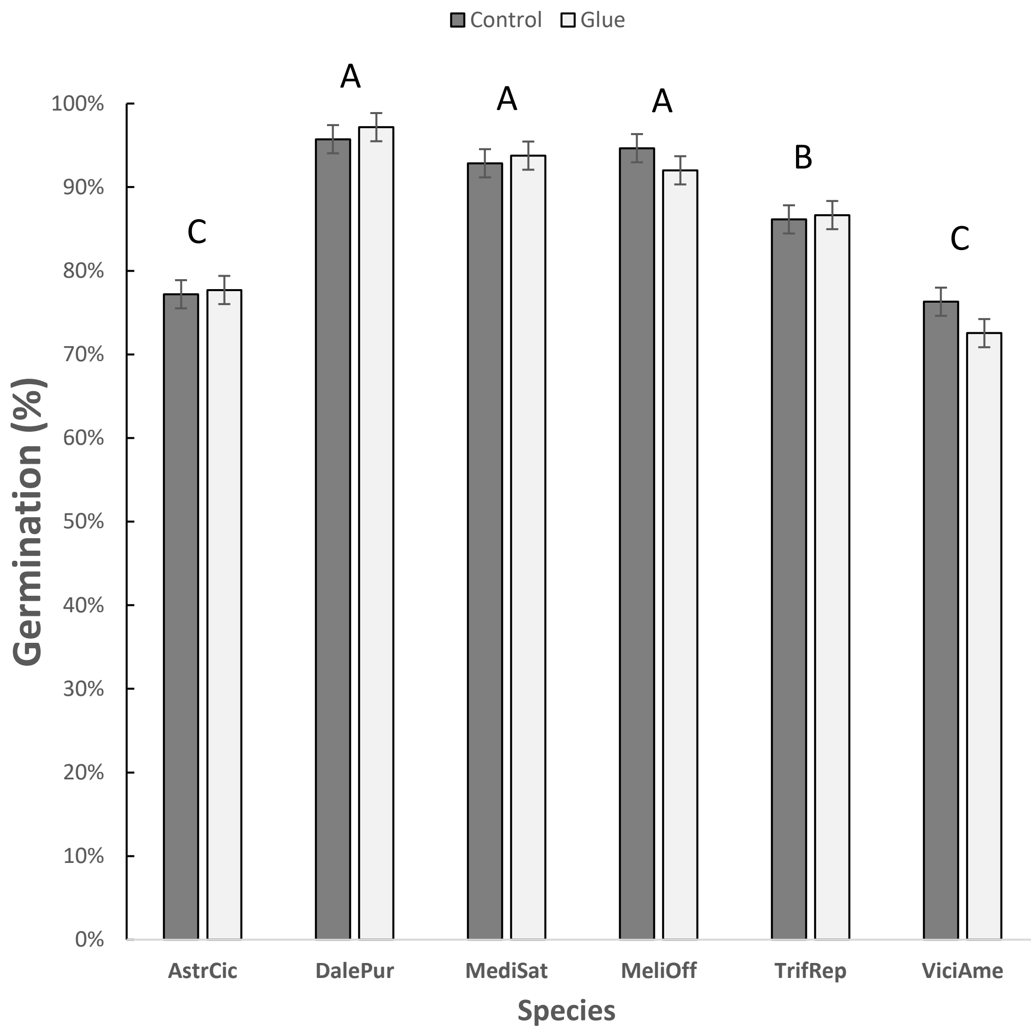


Figure 7.4. Summary of mean germination (% ± 1 SE) for each of 6 legume species when seeds were glued to toothpicks or left untreated to germinate on moist filter paper. Seeds were placed in the dark at ambient room temperature. Significant differences in overall germination among species were found ($P < 0.001$) and are distinguished by capital letters. Gluing seeds had no effect on germination and did not interact with species ($P > 0.05$).

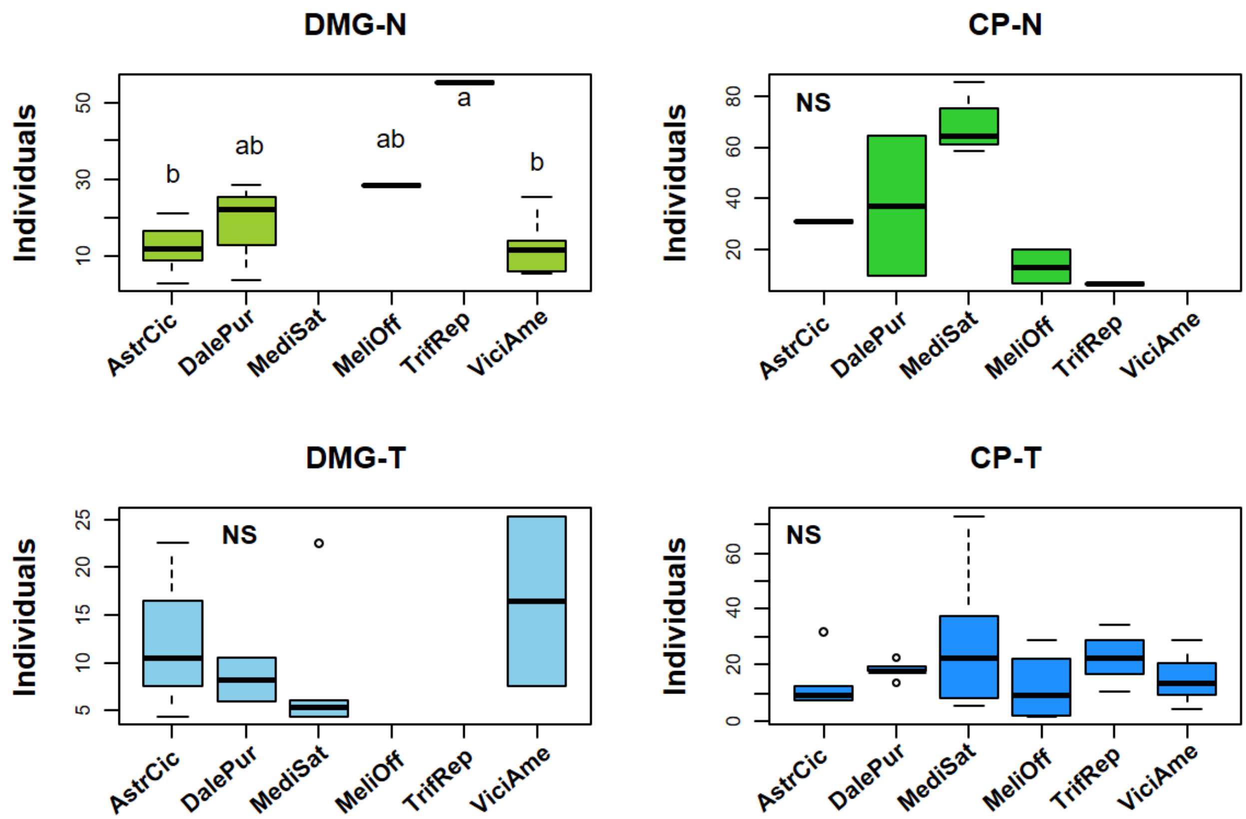


Figure 7.5. The number of individuals required to survive 1 time-step based on the survival of individual seedlings following the initial growing season for each of 6 legume species seeded into native (N) and tame (T) grasslands within the Dry Mixedgrass (DMG) and Central Parkland (CP). For species with adequate data, medians (±IQR) were compared; when individuals failed to survive from a species no median is displayed and the number of individuals required to survive 1 time-step could be interpreted as infinite.

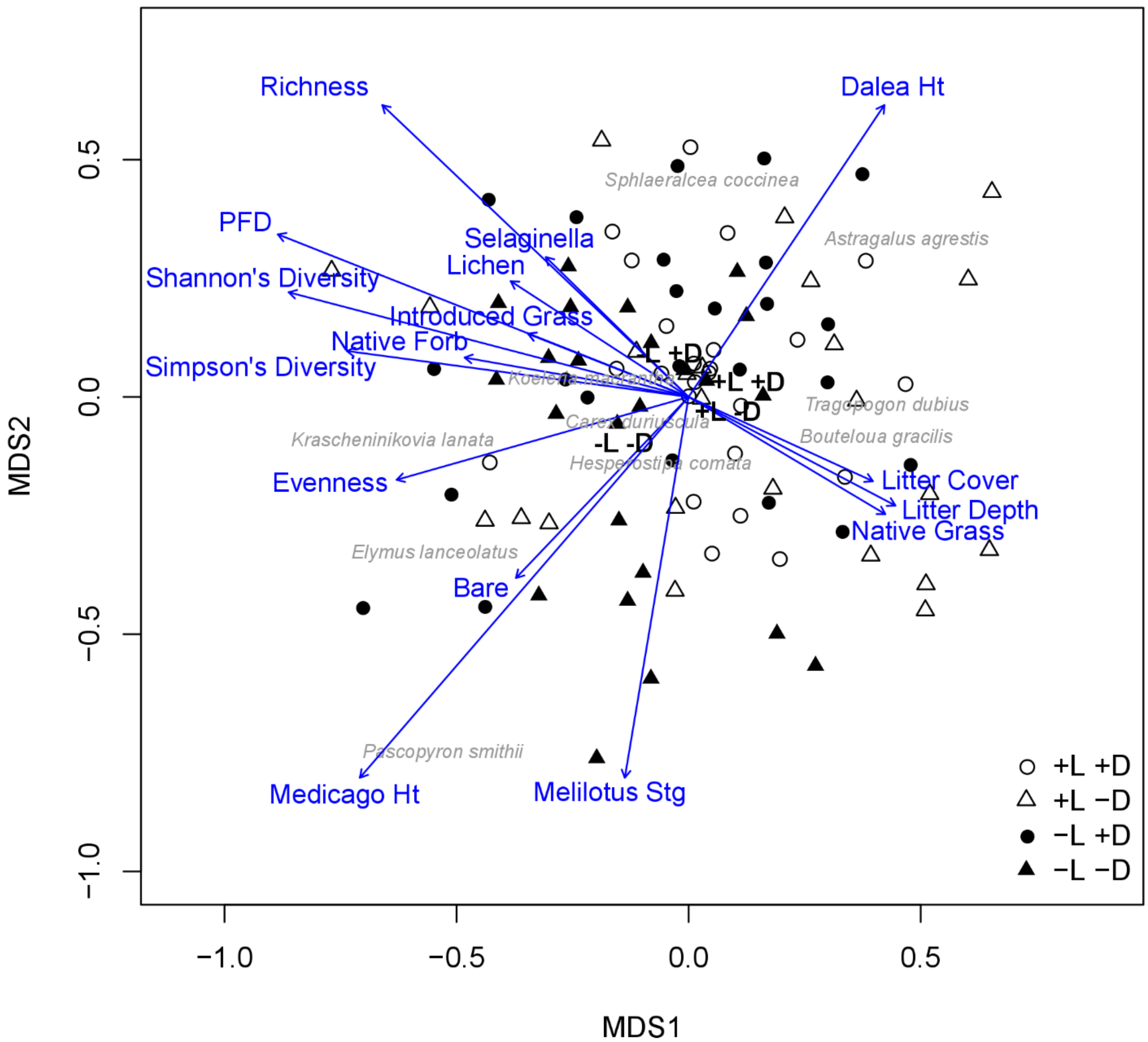


Figure 7.6. NMDS ordination of plant community composition at the native Dry Mixedgrass (DMG-N) location in 2014 (stress = 0.27, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were significantly different (P = 0.036). Plant species with significant correlations to the axes (P < 0.05) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots (P < 0.05). Biplots of legume seedling germination, mortality (over winter 2014), height (Ht), and stage (Stg.) were included if significant (P < 0.05).

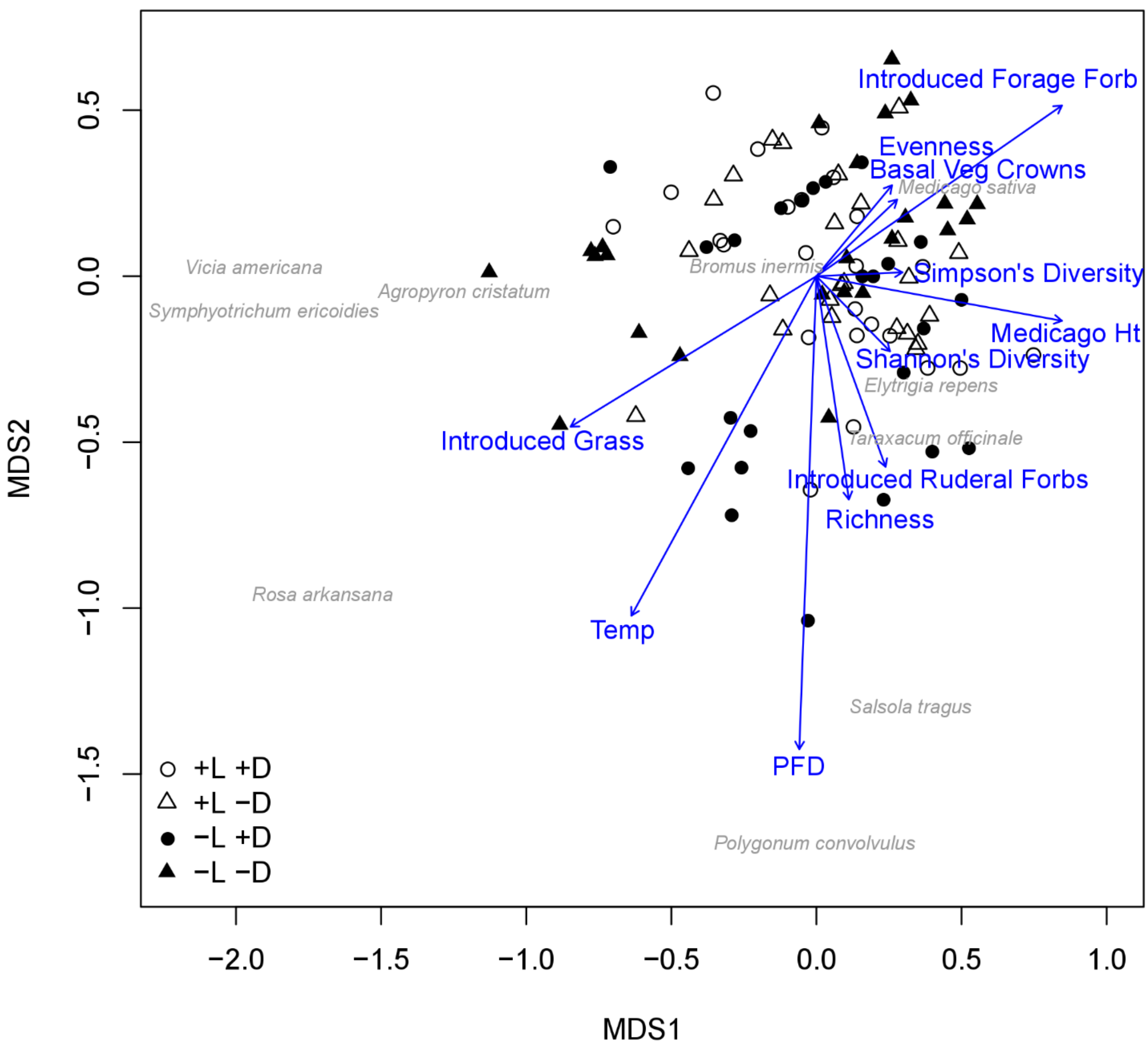


Figure 7.7. NMDS ordination of plant community composition at the tame Dry Mixedgrass (DMG-T) location in 2014 (stress = 0.21, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were not significantly different ($P = 0.098$) and not displayed. Plant species with significant correlations to the axes ($P < 0.05$) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots ($P < 0.05$). Biplots of legume seedling germination, mortality (over winter 2014), height (Ht), and stage (Stg.) were included if significant ($P < 0.05$).

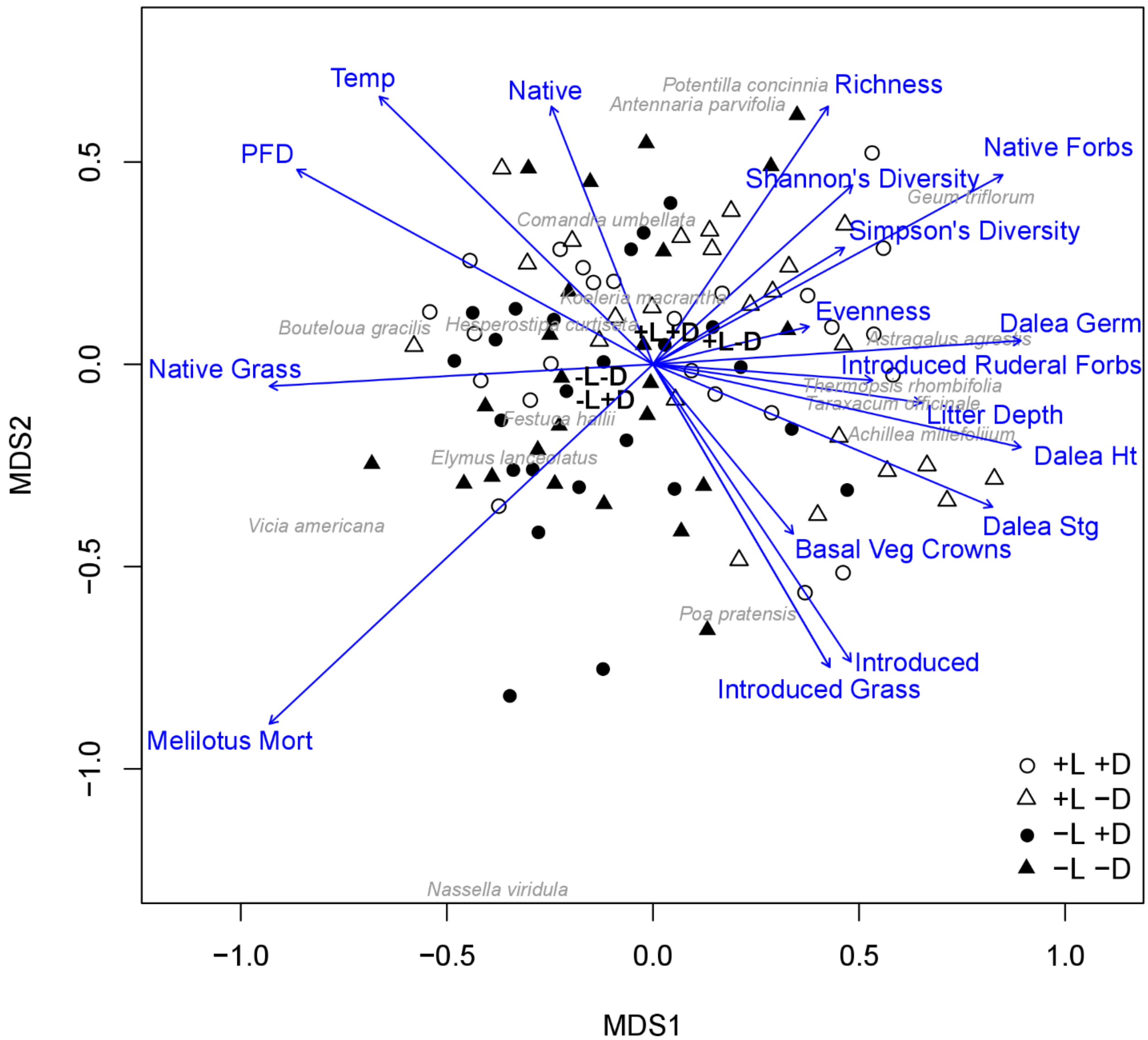


Figure 7.8. NMDS ordination of plant community composition at the native Central Parkland (CP-N) location in 2014 (stress = 0.26, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were significantly different ($P = 0.003$). Plant species with significant correlations to the axes ($P < 0.05$) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots ($P < 0.05$). Biplots of legume seedling germination (Germ.), mortality (over winter 2014) (Mort.), height (Ht), and stage (Stg.) were included if significant ($P < 0.05$).

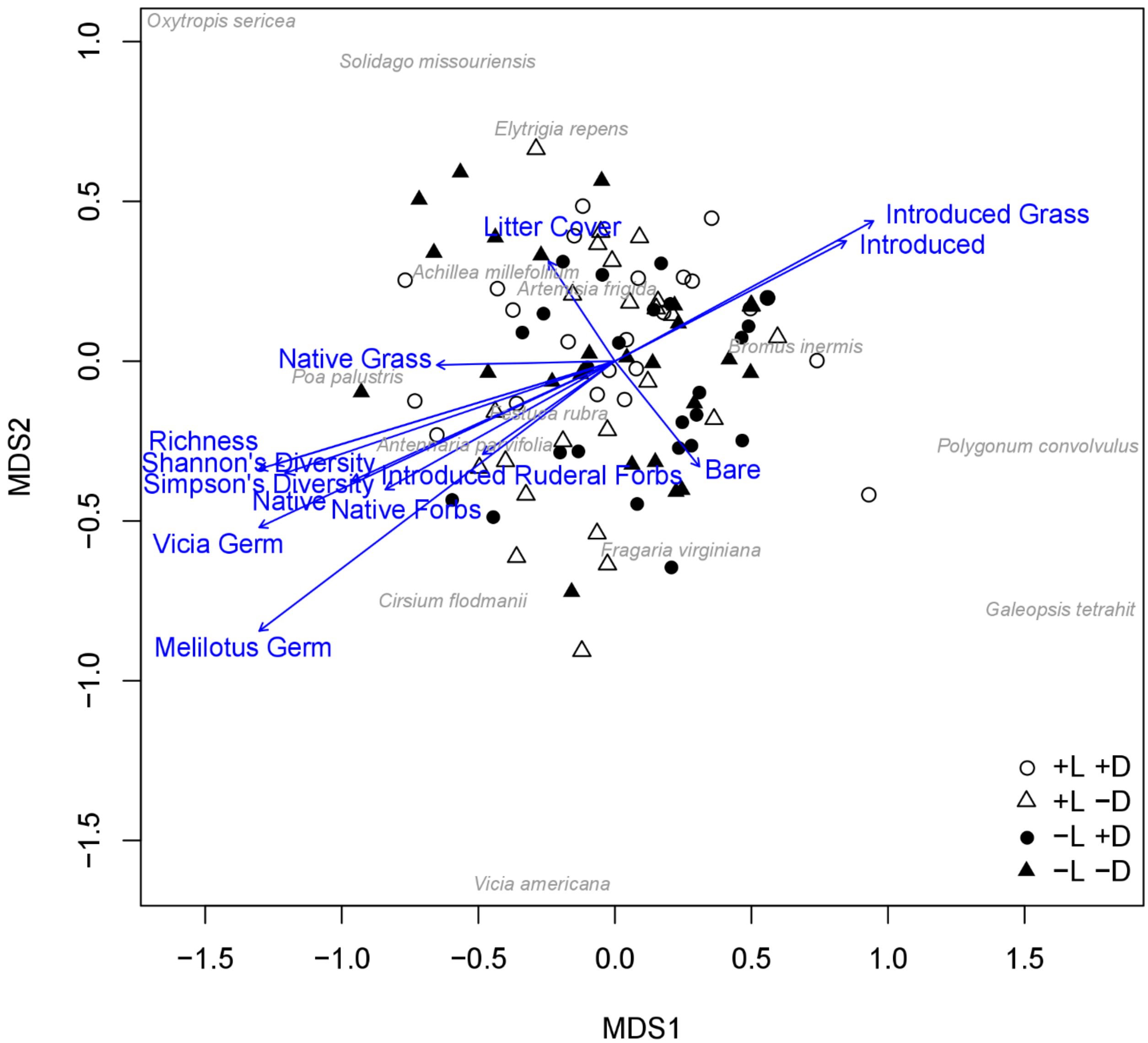


Figure 7.9. NMDS ordination of plant community composition at the tame Central Parkland (CP-T) location in 2014 (stress = 0.23, dimensions = 2, distance = Bray-Curtis). Treatments are symbolized by open symbols for plots with ambient litter (+L) and closed symbols for reduced litter (-L), circles represent defoliated plots (+D) and triangles represent non-defoliated plots (-D); centroids for treatments were not significantly different ($P = 0.331$) and not displayed. Plant species with significant correlations to the axes ($P < 0.05$) are plotted along with significant biplot vectors characterizing the plant community structure and microclimate of plots ($P < 0.05$). Biplots of legume seedling germination (Germ.), mortality (over winter 2014) (Mort.), height (Ht), and stage (Stg.) were included if significant ($P < 0.05$).

Table 7.1. Summary statistics for the germination of 6 legumes species when seeds were glued to toothpicks or left untreated to germinate on moist filter paper. Seeds were in the dark at ambient room temperature. Seeds that did not germinate were classified as 'hard' or 'degraded' and were grouped for analysis.

	Germination		Hard Seeds		Degraded Seeds	
Factor	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Legume Species	65.74	**<0.001**	60.81	**<0.001**	35.41	**<0.001**
Glue Treatment	0.299	0.587	0.69	0.410	0.046	0.832
Species * Glue Treatment	0.903	0.488	5.71	**<0.001**	4.32	**0.003**

Table 7.2. Summary of mean germination (% ± 1 SE) for each of 6 legume species when seeds were glued to toothpicks or left untreated to germinate on moist filter paper. Seeds were in the dark at ambient room temperature. Seeds that did not germinate were classified as 'hard' (dormant) or 'degraded' and were grouped for analysis.

Factor	Treatment	Germination (%)	Hard Seed (%)	Degraded Seed (%)
Legume Species	AstrCic	77.4 (±1.3) c	8.8 (±1.0) c	13.7 (±1.0) a
	DalePur	96.5 (±1.3) a	0.0 (±1.0) d	3.1 (±1.0) b
	MediSat	93.3 (±1.3) a	2.5 (±1.0) d	4.2 (±1.0) b
	MeliOff	93.3 (±1.3) a	3.9 (±1.0) d	2.8 (±1.0) b
	TrifRep	86.4 (±1.3) b	12.9 (±1.0) b	0.7 (±1.0) c
	ViciAme	74.4 (±1.3) c	19.8 (±1.0) a	5.8 (±1.0) b
Species * Glue Treatment (G)	AstrCic -G	77.2 (±1.7)	9.5 (±1.4) cd	13.4 (±1.2) a
	DalePur -G	95.7 (±1.7)	0.0 (±1.4) d	4.3 (±1.2) bc
	MediSat -G	92.9 (±1.7)	5.0 (±1.4) d	2.1 (±1.2) bc
	MeliOff -G	94.7 (±1.7)	0.0 (±1.4) d	5.3 (±1.2) b
	TrifRep -G	86.1 (±1.7)	13.9 (±1.4) bc	0.0 (±1.2) c
	ViciAme -G	76.3 (±1.7)	18.1 (±1.4) ab	5.6 (±1.2) b
	AstrCic +G	77.7 (±1.7)	8.2 (±1.4) cd	14.1 (±1.2) a
	DalePur +G	97.2 (±1.7)	0.0 (±1.4) d	2.0 (±1.2) bc
	MediSat +G	93.8 (±1.7)	0.0 (±1.4) d	6.2 (±1.2) b
	MeliOff +G	92.0 (±1.7)	7.8 (±1.4) cd	0.0 (±1.2) c
	TrifRep +G	86.7 (±1.7)	11.9 (±1.4) bc	1.4 (±1.2) bc
	ViciAme +G	72.5 (±1.7)	21.5 (±1.4) a	6.0 (±1.2) b

Glue Treatment: -G = not glued, +G glued

Table 7.3. Summary statistics for the effect of defoliation (D) and litter removal (L) on microsite variables measured such as litter depth (cm), live vegetation height (cm), bare soil exposure (%), soil surface temperature (℃), photon flux density (PFD), and soil moisture (%), within native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions.

		2014												2015							
		Litter Depth		Vegetation Height		Bare Soil		Soil Surface Temperature		PFD		Soil Moisture		Vegetation Height		Soil Surface Temperature		PFD		Soil Moisture	
Site	Factors	F Value	P Value	F Value	P Value	F Value	P Value	F Value	P Value	F Value	P Value	F Value	P Value	F Value	P Value	F Value	P Value	F Value	P Value	F Value	P Value
DMG-N	L	130.81	**<0.001**	29.33	**<0.001**	72.86	**<0.001**	12.71	**<0.001**	158.35	**<0.001**	1.19	0.277	1.90	0.175	3.74	0.057	5.39	**0.023**	2.99	0.088
	D	15.71	**<0.001**	23.46	**<0.001**	2.41	0.124	15.16	**<0.001**	142.52	**<0.001**	2.32	0.131	9.53	**0.003**	6.18	**0.015**	7.83	**0.007**	3.38	0.070
	L*D	8.802	**0.004**	0.17	0.688	5.72	**0.019**	3.74	0.054	9.11	**0.002**	0.19	0.667	13.22	**0.001**	0.83	0.367	0.12	0.734	2.21	0.142
DMG-T	L	242.23	**<0.001**	0.94	0.345	212.34	**<0.001**	0.40	0.528	77.61	**<0.001**	3.06	0.082	0.01	0.907	0.33	0.569	7.45	**0.009**	0.21	0.650
	D	18.33	**<0.001**	13.18	**0.002**	14.43	**<0.001**	68.95	**<0.001**	224.13	**<0.001**	14.48	**<0.001**	4.24	**0.045**	1.03	0.312	14.01	**<0.001**	0.83	0.364
	L*D	5.01	**0.028**	12.94	**0.002**	20.80	**<0.001**	2.01	0.157	11.24	**0.001**	0.40	0.526	0.42	0.521	5.11	**0.026**	4.58	**0.037**	5.32	**0.023**
CP-N	L	66.54	**<0.001**	3.90	0.062	29.38	**<0.001**	2.19	0.14	23.58	**<0.001**	0.43	0.514	4.21	**0.046**	0.58	0.449	5.00	**0.031**	4.33	**0.043**
	D	4.54	**0.036**	6.89	**0.016**	0.77	0.383	48.27	**<0.001**	104.70	**<0.001**	5.38	**0.022**	2.37	0.131	0.24	0.625	0.60	0.443	1.57	0.217
	L*D	5.65	**0.200**	1.09	0.309	0.10	0.749	0.14	0.705	11.74	**0.001**	3.99	**0.048**	2.92	0.094	4.33	**0.044**	6.81	**0.012**	0.01	0.905
CP-T	L	254.06	**<0.001**	2.64	0.12	115.08	**<0.001**	0.19	0.665	48.94	**<0.001**	0.96	0.329	14.21	**<0.001**	5.29	**0.026**	10.34	**0.002**	7.61	**0.009**
	D	56.31	**<0.001**	10.85	**0.004**	2.61	0.109	11.05	**0.001**	96.40	**<0.001**	2.56	0.113	3.02	0.089	0.01	0.917	0.33	0.568	0.64	0.429
	L*D	41.63	**<0.001**	0.03	0.857	0.43	0.514	13.77	**<0.001**	4.38	**0.037**	1.59	0.209	1.11	0.298	0.99	0.326	0.29	0.591	0.92	0.340

PFD = photon flux density

Table 7.4. Differences in the mean (± SE) microsite characteristics in relation to defoliation (D) and litter removal (L) on microsite variables measured such as litter depth (cm), vegetation height (cm), bare soil (%), soil surface temperature (℃), photon flux density (PFD), and soil moisture (%) in native (N) and tame (T) grasslands of the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions.

			2014						2015			
Site	**Trmt.**		**Litter Depth 2014 (cm)**	**Vegetation Height 2014 (cm)**	**Bare Soil 2014**	**Soil Surface Temp**	**PFD**	**Soil Moisture**	**Vegetation Height 2015 (cm)**	**Soil Surface Temp**	**PFD**	**Soil Moisture**
DMG-N	L	+L	2.3 (±0.3) a	60.1 (±2.2) a	1.3 (±1.0) b	28.6 (±2.6) b	577.3 (±72.4) b				239.5 (±68.7) b	
		-L	0.4 (±0.3) b	42.9 (±2.2) b	8.3 (±1.0) a	30.0 (±2.6) a	1008.6 (±72.4) a				327.4 (±68.7) a	
	D	-D	1.8 (±0.3) a	59.2 (±2.2) a		28.5 (±2.6) b	595.3 (±72.4) b		17.7 (±0.9) a	18.6 (±1.2) b	230.5 (±68.7) b	
		+D	0.9 (±0.3) b	43.8 (±2.2) b		30.1 (±2.6) a	990.6 (±72.4) a		13.7 (±0.9) b	19.7 (±1.2) a	336.4 (±68.7) a	
	L*D	+L -D	3.2 (±0.4) a		0.3 (±1.3) c		362.8 (±78.8) c		16.2 (±1.3) a			
		+L +D	1.4 (±0.4) b		2.4 (±1.3) b		791.9 (±78.8) b		16.9 (±1.3) a			
		-L -D	0.4 (±0.4) c		8.7 (±1.3) a		827.7 (±78.8) b		19.1 (±1.3) a			
		-L +D	0.3 (±0.4) c		7.8 (±1.3) a		1189.4 (±78.8) a		10.5 (±1.3) b			
DMG-T	L	+L	1.9 (±0.1) a		2.9 (±2.5) b		241.6 (±66.7) b				98.2 (±48.3) b	
		-L	0.4 (±0.1) b		35.9 (±2.5) a		490.4 (±66.8) a				148.1 (±48.3) a	
	D	-D	1.4 (±0.1) a	20.2 (±0.9) a	11.95 (±2.5) b	19.7 (±2.6) b	183.8 (±66.8) b	10.9 (±1.7) b	57.6 (±4.4) a		91.2 (±48.3) b	
		+D	0.9 (±0.1) b	15.7 (±0.9) b	26.77 (±2.5) a	23.2 (±2.6) a	548.2 (±66.8) a	12.6 (±1.7) a	47.5 (±4.4) b		155.1 (±48.3) a	
	L*D	+L -D	2.3 (±0.1) a	21.9 (±1.2) a	2.8 (±3.2) c		122.0 (±69.7) d			17.3 (±4.6) ab	89.6 (±50.3) b	5.4 (±2.4) b
		+L +D	1.4 (±0.1) b	12.9 (±1.2) b	3.0 (±3.2) c		361.2 (±69.7) b			16.8 (±4.6) ab	106.8 (±50.3) b	6.2 (±2.4) a
		-L -D	0.5 (±0.1) c	18.6 (±1.2) ab	21.1 (±3.2) b		245.7 (±69.9) c			16.1 (±4.6) b	92.8 (±50.3) b	5.9 (±2.4) ab
		-L +D	0.3 (±0.1) c	18.6 (±1.2) ab	50.6 (±3.2) a		735.1 (±69.8) a			17.6 (±4.6) a	203.4 (±50.3) a	5.8 (±2.4) ab
CP-N	L	+L	1.0 (±0.1) a		10.3 (±3.1) b		438.1 (±73.0) b		13.1 (±0.4) a		951.6 (±118.2) b	12.3 (±3.1) b
		-L	0.3 (±0.1) b		22.7 (±3.1) a		540.7 (±72.9) a		11.9 (±0.4) b		1124.4 (±118.2) a	13.5 (±3.1) a
	D	-D	0.8 (±0.1) a	17.0 (±0.7) a		23.1 (±2.8) b	358.8 (±73.0) b	18.1 (±3.1) a				
		+D	0.6 (±0.1) b	14.4 (±0.7) b		26.0 (±2.8) a	620.0 (±73.0) a	17.0 (±3.1) b				
	L*D	+L -D	1.2 (±0.2) a				273.2 (±76.0) c	17.4 (±3.2) ab		32.2 (±4.8) ab	820.8 (±130.3) b	
		+L +D	0.8 (±0.2) b				602.9 (±75.9) a	17.2 (±3.2) ab		33.7 (±4.8) ab	1082.3 (±130.3) ab	
		-L -D	0.3 (±0.2) c				444.4 (±75.8) b	18.9 (±3.2) a		33.9 (±4.8) a	1195.3 (±130.3) a	
		-L +D	0.3 (±0.2) c				637.1 (±76.1) a	16.8 (±3.2) b		31.3 (±4.8) b	1053.5 (±130.3) ab	
CP-T	L	+L	2.8 (±0.2) a		2.0 (±2.4) b		307.6 (±86.3) b		15.9 (±1.4) b	30.8 (±3.3) a	734.5 (±87.2) b	13.1 (±2.0) a
		-L	0.6 (±0.2) b		29.6 (±2.4) a		491.0 (±86.2) a		21.3 (±1.4) a	28.7 (±3.3) b	1048.1 (±87.2) a	11.7 (±2.0) b
	D	-D	2.4 (±0.2) a	33.2 (±2.0) a		24.0 (±2.5) b	272.4 (±86.3) b					
		+D	1.1 (±0.2) b	24.0 (±2.0) b		25.5 (±2.5) a	526.2 (±86.2) a					
	L*D	+L -D	4.1 (±0.2) a			23.1 (±2.5) b	164.0 (±89.2) c					
		+L +D	1.6 (±0.2) b			26.2 (±2.5) a	451.3 (±89.4) b					
		-L -D	0.6 (±0.2) c			25.0 (±2.5) a	380.8 (±89.4) b					
		-L +D	0.6 (±0.2) c			24.7 (±2.5) a	601.1 (±89.1) a					

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 7.5. Summary statistics for perMANOVA tests of plant community composition shifts under litter manipulation (L) and defoliation (D) treatments across all Dry Mixedgrass (DMG) and Central Parkland (CP) sites in 2014.

		DMG-N		**DMG-T**		**CP-N**		**CP-T**	
Factors	**Treatment**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Litter (L)		8.53	**0.001**	0.78	0.367	2.55	**0.010**	0.98	0.381
Defoliation (D)		5.36	**0.001**	1.98	0.131	7.20	**0.001**	2.94	**0.023**
L*D		2.51	**0.026**	0.27	0.711	0.91	0.467	2.06	0.081
Contrasts									
L*D	+L -D vs. +L +D	1.60	0.137						
	+L -D vs. -L -D	3.23	**0.014**						
	+L -D vs. -L +D	8.41	**0.001**						
	+L +D vs. -L -D	4.81	**0.002**						
	+L +D vs. -L +D	8.32	**0.001**						
	-L -D vs. -L +D	5.83	**0.001**						

Litter: ambient = +L, removed = -L

Defoliation: not defoliated = -D, defoliated = +D

Table 7.6. Summary statistics for plant community characteristics altered via defoliation (D) and litter removal (L) treatments at all Dry Mixedgrass (DMG) and Central Parkland (CP) sites during the summer of 2014.

		Native Grass Cover		**Native Forb Cover**		**Introduced Forage Cover**		**Richness**		**Shannon's Diversity**		**Pielou's Evenness**	
Site	**Factors**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
DMG-N	Litter (L)	7.97	**0.006**	0.11	0.739			4.88	**0.03**	7.92	**0.006**	3.84	0.053
	Defoliation (D)	15.73	**<0.001**	0.00	0.979			2.44	1.12	0.15	0.701	2.91	0.092
	L*D	0.58	0.449	0.91	0.343			2.43	0.123	3.50	0.065	1.87	0.175
DMG-T	L					0.49	0.487	6.69	**0.011**	12.44	**0.001**	5.36	**0.023**
	D					13.55	**<0.001**	5.44	**0.022**	13.32	**<0.001**	6.00	**0.016**
	L*D					3.35	0.071	2.96	0.089	1.96	0.165	0.12	0.734
CP-N	L	27.09	**<0.001**	28.13	**<0.001**			4.64	**0.034**	3.87	0.052	1.03	0.312
	D	1.20	0.277	2.58	0.112			0.04	0.842	0.25	0.615	0.24	0.628
	L*D	1.13	0.29	0.82	0.368			2.45	0.121	0.22	0.642	0.45	0.505
CP-T	L					0.93	0.338	0.21	0.646	0.87	0.354	1.38	0.244
	D					0.01	0.943	1.84	0.178	2.68	0.105	0.07	0.785
	L*D					0.04	0.841	0.90	0.346	0.02	0.876	1.15	0.287

Table 7.7. Summary mean (± SE) aboveground plant community characteristics altered via defoliation (D) and litter removal (L) treatments across all Dry Mixedgrass (DMG) and Central Parkland (CP) sites during summer 2014.

Site	Treatment	Contrasts	Native Grass %	Native Forb %	Introduced Forage %	Richness	Shannon's Diversity	Pielou's Evenness
DMG-N	Litter (L)	Ambient (+L)	77.4 (±1.8) a			7.4 (±0.2) b	1.32 (±0.04) b	
		Removed (-L)	70.0 (±1.8) b			8.1 (±0.2) a	1.48 (±0.04) a	
	Defoliation (D)	Not Defoliated (-D)	78.9 (±1.8) a					
		Defoliated (+D)	68.5 (±1.8) b					
DMG-T	L	+L				4.91 (±0.32) a	0.99 (±0.06) a	0.63 (±0.02) a
		-L				4.48 (±0.32) b	0.84 (±0.06) b	0.57 (±0.02) b
	D	-D			101.5 (±2.5) a	4.49 (±0.32) b	0.84 (±0.06) b	0.57 (±0.02) b
		+D			94.6 (±2.5) b	4.90 (±0.32) a	0.99 (±0.06) a	0.63 (±0.02) a
CP-N	L	+L	60.3 (±4.4) b	31.9 (±3.2) a		10.52 (±0.42) a		
		-L	73.6 (±4.4) a	19.1 (±3.2) b		9.76 (±0.42) b		

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 7.8. Summary statistics for germination (Germ.) and recruitment (Rec.) of legume seedlings over 3 successive years for each of 6 legume species planted in plots with defoliation (D) and litter removal (L) treatments at native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions.

		Germ. Year 1		**Germ. Year 2***		**Rec. Year 2***		**Germ. Year 3***		**Rec. Year 3***		**Germ. Overall**	
Site	**Factors**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
DMG-N	Litter (L)	3.21	0.132	0.24	0.624	0.04	0.858	1.94	0.168	0.37	0.549	2.81	0.154
	Defoliation (D)	0.003	0.955	2.55	0.115	1.42	0.285	0.06	0.811	2.47	0.214	0.05	0.837
	L*D	4.06	**0.049**	0.43	0.514	0.46	0.503	0.06	0.806	0.045	0.832	3.23	0.078
	Species (S)	9.65	**<0.001**	68.23	**<0.001**	43.77	**<0.001**	1.34	0.099	4.47	**0.001**	6.63	**<0.001**
	S*L	0.88	0.502	0.29	0.917	0.72	0.609	1.39	0.099	1.29	0.281	0.75	0.588
	S*D	0.45	0.811	0.67	0.650	0.86	0.514	0.06	0.806	1.74	0.137	0.53	0.749
	S*L*D	1.28	0.285	0.91	0.478	1.99	0.093	0.06	0.997	0.21	0.958	1.45	0.219
DMG-T	Litter (L)	6.72	**0.041**	1.58	0.213	0.22	0.652	0.71	0.402	0.34	0.602	7.38	**0.035**
	Defoliation (D)	1.97	0.165	0.10	0.766	5.89	**0.018**	0.71	0.402	2.32	0.132	1.96	0.167
	L*D	0.02	0.895	0.47	0.493	1.18	0.281	0.71	0.402	0.87	0.356	0.003	0.959
	Species (S)	6.61	**<0.001**	6.11	**<0.001**	7.35	**<0.001**	1.28	0.282	1.83	0.118	5.69	**<0.001**
	S*L	2.23	0.061	0.28	0.920	0.33	0.892	0.71	0.617	0.40	0.849	2.24	0.06
	S*D	3.49	**0.007**	0.39	0.853	2.34	0.051	0.71	0.617	1.37	0.245	3.32	**0.01**
	S*L*D	0.35	0.883	0.67	0.648	0.78	0.567	0.71	0.617	0.63	0.680	0.43	0.829
CP-N	Litter (L)	0.05	0.817	0.09	0.77	0.1	0.75	4.52	**0.037**	8.49	**0.005**	0.69	0.408
	Defoliation (D)	0.06	0.812	0.95	0.333	0.73	0.424	1.98	0.209	0.58	0.476	0.01	0.912
	L*D	0.01	0.927	1.34	0.251	0.11	0.736	3.25	0.076	1.13	0.292	0.003	0.956
	Species (S)	6.12	**<0.001**	48.95	**<0.001**	28.13	**<0.001**	22.07	**<0.001**	22.31	**<0.001**	5.9	**<0.001**
	S*L	1.29	0.278	0.53	0.75	0.21	0.958	2.94	**0.019**	3.63	**0.006**	1.18	0.331
	S*D	0.98	0.438	0.24	0.943	0.82	0.54	5.21	**<0.001**	1.13	0.292	0.95	0.456
	S*L*D	0.52	0.759	1.38	0.243	1.06	0.392	5.2142	**<0.001**	7.18	**<0.001**	0.47	0.795
CP-T	Litter (L)	0.29	0.611	0.09	0.766	0.04	0.851	0.72	0.400	0.36	0.576	0.20	0.671
	Defoliation (D)	17.81	**<0.001**	0.02	0.881	0.42	0.518	0.44	0.534	0.63	0.46	15.56	**<0.001**
	L*D	3.38	0.0705	1.64	0.204	1.28	0.263	4.12	**0.046**	0.71	0.403	5.02	**0.028**
	Species (S)	25.52	**<0.001**	4.51	**0.001**	1.52	0.195	2.87	**0.021**	3.24	**0.012**	20.64	**<0.001**
	S*L	0.57	0.72	0.50	0.772	0.13	0.985	0.59	0.706	0.3	0.909	0.46	0.806
	S*D	1.05	0.396	0.67	0.646	0.15	0.980	0.82	0.540	0.43	0.823	0.89	0.491
	S*L*D	1.12	0.358	1.77	0.130	1.24	0.300	0.76	0.579	0.83	0.536	0.79	0.561

*Non-transformable, zero-inflated; log(x+0.01) transformation used.

Table 7.9. Summary mean (± 1 SE) germination (Germ.) and recruitment (Rec.) as a percent (%) of legume seedlings placed into the initial seed bank over 3 successive years for each of 6 legume species subset in plots that had been treated with defoliation (D), litter removal (L), or combinations thereof, during the initial year at native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions.

Site	Trmt.	Contrasts	Germ. Year 1	Germ. Year 2	Rec. Year 2	Germ. Year 3	Rec. Year 3	Overall
DMG-N	L*D	+L -D	20.7 (±2.6) ab					
		+L +D	16.4 (±2.6) b					
		-L -D	20.4 (±2.6) ab					
		-L +D	24.8 (±2.6) a					
	S	AstrCic	13.2 (±2.8) b	10.2 (±0.6) a	12.0 (±1.0) a		1.1 (±0.2) a	23.8 (±3.0) ab
		DalePur	14.4 (±2.8) b	0.2 (±0.6) bc	1.7 (±1.0) c		0.1 (±0.2) b	14.5 (±3.0) b
		MediSat	27.3 (±2.8) a	0.2 (±0.6) bc	0.2 (±1.0) c		0.0 (±0.2) b	27.5 (±3.0) a
		MeliOff	30.8 (±2.8) a	0.2 (±0.6) bc	0.3 (±1.0) c		0.0 (±0.2) b	30.9 (±3.0) a
		TrifRep	25.2 (±2.8) a	0.0 (±0.6) c	0.3 (±1.0) c		0.0 (±0.2) b	25.2 (±3.0) ab
		ViciAme	13.1 (±2.8) b	1.1 (±0.6) b	4.1 (±1.0) b		0.0 (±0.2) b	14.2 (±3.0) b
DMG-T	L	+L	11.2 (±1.5) b					11.6 (±1.5) b
		-L	16.6 (±1.5) a					17.4 (±1.5) a
	D	-D			0.6 (±0.5) b			
		+D			2.1 (±0.5) a			
	S	AstrCic	8.6 (±2.4) b	2.0 (±0.3) a	3.0 (±0.8) ab			10.8 (±2.5) b
		DalePur	8.1 (±2.4) b	0.2 (±0.3) b	0.6 (±0.8) ab			8.8 (±2.5) b
		MediSat	22.7 (±2.4) a	0.3 (±0.3) b	3.9 (±0.8) a			23.1 (±2.5) a
		MeliOff	17.3 (±2.4) a	0.0 (±0.3) b	0.0 (±0.8) b			17.5 (±2.5) ab
		TrifRep	17.8 (±2.4) a	0.0 (±0.3) b	0.0 (±0.8) b			17.8 (±2.5) ab
		ViciAme	8.8 (±2.4) b	0.2 (±0.3) b	0.5 (±0.8) ab			8.9 (±2.5) b
	S*D	AstrCic -D	6.9 (±3.4) b					9.4 (±3.5) b
		DalePur -D	10.0 (±3.4) b					10.6 (±3.5) b
		MediSat -D	18.4 (±3.4) ab					18.8 (±3.5) ab
		MeliOff -D	19.7 (±3.4) ab					19.7 (±3.5) ab
		TrifRep -D	26.9 (±3.4) a					26.9 (±3.5) a
		ViciAme -D	9.7 (±3.4) b					10.0 (±3.5) b
		AstrCic +D	10.3 (±3.4) b					12.2 (±3.5) ab
		DalePur +D	6.3 (±3.4) b					6.9 (±3.5) b
		MediSat +D	26.9 (±3.4) a					27.5 (±3.5) a
		MeliOff +D	15.0 (±3.4) ab					15.3 (±3.5) ab
		TrifRep +D	8.8 (±3.4) b					8.8 (±3.5) b
		ViciAme +D	9.7 (±3.4) b					7.8 (±3.5) b
CP-N	L	+L				0.2 (±0.3) b	0.2 (±0.3) b	
		-L				1.1 (±0.3) a	1.2 (±0.3) a	
	S	AstrCic	17.8 (±4.6) ab	11.56 (±0.8) a	12.2 (±0.8) a	3.8 (±0.1) a	3.8 (±0.1) a	33.1 (±4.7) a
		DalePur	27.5 (±4.6) a	0.3 (±0.8) bc	0.9 (±0.8) b	0.0 (±0.1) b	0.0 (±0.1) b	27.8 (±4.7) a
		MediSat	31.9 (±4.6) a	0.0 (±0.8) c	0.5 (±0.8) b	0.0 (±0.1) b	0.0 (±0.1) b	31.9 (±4.7) a
		MeliOff	29.4 (±4.6) a	0.0 (±0.8) c	0.6 (±0.8) b	0.0 (±0.1) b	0.0 (±0.1) b	29.4 (±4.7) a
		TrifRep	19.7 (±4.6) ab	0.2 (±0.8) bc	0.2 (±0.8) b	0.0 (±0.1) b	0.0 (±0.1) b	19.8 (±4.7) a
		ViciAme	8.6 (±4.6) b	1.6 (±0.8) b	1.6 (±0.8) b	0.2 (±0.1) b	0.5 (±0.1) b	10.3 (±4.7) b
	S*L	AstrCic -L				6.3 (±0.7) a	6.3 (±0.7) a	
		DalePur -L				0.0 (±0.7) c	0.0 (±0.7) c	
		MediSat -L				0.0 (±0.7) c	0.0 (±0.7) c	
		MeliOff -L				0.0 (±0.7) c	0.0 (±0.7) c	
		TrifRep -L				0.0 (±0.7) c	0.0 (±0.7) c	
		ViciAme -L				0.3 (±0.7) bc	0.9 (±0.7) bc	
		AstrCic +L				1.3 (±0.7) b	1.3 (±0.7) b	
		DalePur +L				0.0 (±0.7) c	0.0 (±0.7) c	
		MediSat +L				0.0 (±0.7) c	0.0 (±0.7) c	
		MeliOff +L				0.0 (±0.7) c	0.0 (±0.7) c	
		TrifRep +L				0.0 (±0.7) c	0.0 (±0.7) c	

		ViciAme +L				0.0 (±0.7) c	0.0 (±0.7) c	
	S*D	AstrCic -D				1.3 (±0.8) b	1.3 (±0.8) b	
		DalePur -D				0.0 (±0.8) c	0.0 (±0.8) c	
		MediSat -D				0.0 (±0.8) c	0.0 (±0.8) c	
		MeliOff -D				0.0 (±0.8) c	0.0 (±0.8) c	
		TrifRep -D				0.0 (±0.8) c	0.0 (±0.8) c	
		ViciAme -D				0.3 (±0.8) bc	0.9 (±0.8) bc	
		AstrCic +D				6.3 (±0.8) a	6.3 (±0.8) a	
		DalePur +D				0.0 (±0.8) c	0.0 (±0.8) c	
		MediSat +D				0.0 (±0.8) c	0.0 (±0.8) c	
		MeliOff +D				0.0 (±0.8) c	0.0 (±0.8) c	
		TrifRep +D				0.0 (±0.8) c	0.0 (±0.8) c	
		ViciAme +D				0.0 (±0.8) c	0.0 (±0.8) c	
CP-T	D	-D	21.6 (±1.7) b					23.0 (±1.9) b
		+D	30.1 (±1.7) a					31.4 (±1.9) a
	LD	+L -D				0.7 (±0.3) ab		34.5 (±2.6) a
		+L +D				0.0 (±0.3) b		21.2 (±2.6) b
		-L -D				0.3 (±0.3) ab		28.3 (±2.6) ab
		-L +D				0.8 (±0.3) a		24.7 (±2.6) ab
	S	AstrCic	14.8 (±2.7) c	3.1 (±0.5) a		1.4 (±0.3) a	1.9 (±0.9) ab	19.4 (±2.8) c
		DalePur	29.8 (±2.7) b	0.5 (±0.5) ab		0.0 (±0.3) b	0.5 (±0.9) ab	30.3 (±2.8) b
		MediSat	42.8 (±2.7) a	0.2 (±0.5) b		0.5 (±0.3) ab	3.6 (±0.9) a	43.5 (±2.8) a
		MeliOff	18.3 (±2.7) c	0.5 (±0.5) ab		0.0 (±0.3) b	0.0 (±0.9) b	18.8 (±2.8) c
		TrifRep	36.7 (±2.7) ab	0.6 (±0.5) ab		0.5 (±0.3) ab	0.6 (±0.9) ab	37.8 (±2.8) ab
		ViciAme	12.7 (±2.7) c	0.3 (±0.5) ab		0.5 (±0.3) ab	1.3 (±0.9) ab	13.4 (±2.8) c

Litter: ambient (+L), removed (-L).
Defoliation: defoliated (+D), not defoliated (-D).
Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 7.10. Summary statistics for the assessment of mortality rate of 6 species (S) of legume seedlings between the first and second year for plots that were exposed to varying litter removal (L) and defoliation (D) treatments. Legumes were seeded into native (N) and tame (T) grassland in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions.

	DMG-N		DMG-T		CP-N		CP-T	
Factors	X^2	**P Value**	X^2	**P Value**	X^2	**P Value**	X^2	**P Value**
Litter (L)	4.92	**0.027**	0.35	0.556	0.00	1.000	1.25	0.263
Defoliation (D)	1.13	0.289	0.01	0.914	0.00	0.999	0.21	0.649
L*D	10.31	**0.001**	0.01	0.920	0.00	0.999	0.01	0.914
Species (S)	12.71	**0.026**	27.15	**<0.001**	0.00	1.000	79.92	**<0.001**
S*L	12.03	**0.034**	3.55	0.616	0.00	1.000	43.52	**<0.001**
S*D	0.02	1.000	0.01	1.000	0.00	1.000	21.17	**0.001**
S*L*D	13.32	**0.021**	0.01	1.000	0.00	1.000	5.89	0.317

Table 7.11. Differences in the mortality rate (%) of legume seedlings between the first and second year, in combination with all combinations of litter manipulation (L) and defoliation (D) treatments. Legumes were seeded into native (N) and tame (T) grassland in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions.

Treatment	Contrasts	DMG-N	DMG-T	CP-T
Litter	Ambient (+L)	94.2 (±1.7) a		
	Removed (-L)	93.8 (±1.7) b		
Litter*Defoliation	+L -D	92.8 (±2.4) ab		
	+L +D	95.6 (±2.4) a		
	-L -D	95.2 (±2.4) ab		
	-L +D	92.3 (±2.4) b		
Species (S)	AstrCic	86.8 (±2.9) ab	91.4 (±2.9) b	91.6 (±5.0) ab
	DalePur	95.1 (±3.0) ab	95.9 (±3.0) ab	95.2 (±4.9) ab
	MediSat	100.0 (±2.9) a	87.8 (±2.9) c	92.4 (±4.9) ab
	MeliOff	99.4 (±2.9) ab	99.9 (±3.0) a	88.8 (±5.0) b
	TrifRep	99.3 (±2.9) ab	100.0 (±2.9) a	97.1 (±4.9) a
	ViciAme	83.3 (±2.9) b	96.3 (±3.1) a	86.8 (±5.0) b
S*L	AstrCic -L	85.9 (±4.1) b		90.5 (±6.9) ab
	DalePur -L	91.8 (±4.1) b		93.7 (±6.9) a
	MediSat -L	100.0 (±4.1) a		90.2 (±6.9) ab
	MeliOff -L	100.0 (±4.1) a		81.0 (±6.9) b
	TrifRep -L	99.3 (±4.1) a		96.9 (±6.9) a
	ViciAme -L	85.6 (±4.1) b		91.1 (±7.4) ab
	AstrCic +L	87.7 (±4.1) b		92.6 (±7.4) a
	DalePur +L	98.3 (±4.4) a		96.7 (±6.9) a
	MediSat +L	100.0 (±4.1) a		94.6 (±6.9) a
	MeliOff +L	98.8 (±4.1) a		96.7 (±7.4) a
	TrifRep +L	99.3 (±4.1) a		97.4 (±6.9) a
	ViciAme +L	81.1 (±4.1) b		82.5 (±6.9) b
S*D	AstrCic -D			97.2 (±6.6) a
	DalePur -D			96.5 (±6.6) a
	MediSat -D			94.6 (±6.6) ab
	MeliOff -D			89.5 (±6.6) bc
	TrifRep -D			99.0 (±6.6) a
	ViciAme -D			96.3 (±6.6) a
	AstrCic +D			85.9 (±7.1) b
	DalePur +D			93.8 (±6.6) ab
	MediSat +D			90.2 (±6.6) b
	MeliOff +D			88.2 (±7.1) b
	TrifRep +D			95.3 (±6.6) a
	ViciAme +D			77.4 (±7.1) c

Defoliation: not defoliated = -D, defoliated = +D

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 7.12. Summary statistics for the height and growth stage of 6 legume species subset within plots that had been treated with defoliation (D) and litter removal (L) treatments during the initial year within native (N) and tame (T) grasslands of the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions.

		Height Year 1		Stage Year 1		Height Year 2		Stage Year 2	
Site	**Factors**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
DMG-N	Litter (L)	0.07	0.789	0.01	0.931	0.32	0.596	1.74	0.264
	Defoliation (D)	18.14	**<0.001**	2.73	0.164	0.12	0.743	2.90	0.152
	L*D	0.197	0.659	1.37	0.248	0.13	0.731	0.24	0.648
	Species (S)	72.26	**<0.001**	25.81	**<0.001**	0.43	0.667	0.58	0.658
	S*L	2.84	**0.022**	1.49	0.216	0.10	0.758	2.37	0.183
	S*D	1.5	0.2	1.92	0.113	0.02	0.886	1.28	0.306
	S*L*D	1.39	0.238	1.52	0.206	-	-	-	-
DMG-T	L	0.6	0.805	1.44	0.269	3.77	0.066	2.87	0.105
	D	2.27	0.147	2.45	0.14	5.61	0.147	2.32	0.143
	L*D	0.18	0.909	0.04	0.837	1.03	0.376	-	-
	S	1.85	0.182	3.03	0.055	1.12	0.369	0.32	0.73
	S*L	0.63	0.68	0.78	0.587	-	-	0.20	0.820
	S*D	0.18	0.909	0.39	0.76	-	-	0.30	0.590
	S*L*D	-	-	-	-	-	-	-	-
CP-N	L	0.02	0.877	0.12	0.73	1.49	0.290		
	D	0.20	0.656	0.01	0.928	2.43	0.194		
	L*D	1.42	0.241	0.002	0.962	0.75	0.436		
	S	15.27	**<0.001**	8.61	**0.002**	6.65	**0.047**		
	S*L	2.23	0.087	1.31	0.328	0.79	0.425		
	S*D	0.83	0.545	0.76	0.596	0.55	0.501		
	S*L*D	0.61	0.691	0.18	0.967	-	-		
CP-T	L	3.31	0.074	0.26	0.612	0.04	0.835	0.83	0.369
	D	0.37	0.574	0.03	0.874	3.11	0.088	2.10	0.157
	L*D	0.21	0.652	0.32	0.576	0.05	0.828	2.15	0.152
	S	55.93	**<0.001**	19.76	**<0.001**	6.18	**0.001**	19.24	**<0.001**
	S*L	5.65	**<0.001**	1.16	0.34	0.19	0.941	0.30	0.878
	S*D	0.62	0.684	1.21	0.312	1.73	0.168	1.85	0.14
	S*L*D	1.33	0.272	0.84	0.528	0.02	0.895	0.38	0.540

Table 7.13. Mean (± 1 SE) height (cm) and growth stage of 6 legume species subset in plots that had been treated with defoliation (D) and litter removal (L) during the initial year of establishment at native (N) and tame (T) grasslands in the Dry Mixedgrass (DMG) and Central Parkland (CP) natural subregions. See Appendix E (Table E.2) for definition of growth stages.

Site	Treatment	Contrasts	Height Year 1	Stage Year 1	Height Year 2	Stage Year 2
DMG-N	Defoliation	Not Defoliated (-D)	2.90 (±0.16) a			
		Defoliated (+D)	1.89 (±0.17) b			
	Species (S)	AstrCic	0.94 (±0.30) cd	2.2 (±0.2) c		
		DalePur	2.31 (±0.35) bc	3.0 (±0.2) b		
		MediSat	1.51 (±0.30) c	2.2 (±0.2) c		
		MeliOff	3.48 (±0.19) b	3.1 (±0.1) b		
		TrifRep	0.36 (±0.27) d	2.0 (±0.1) c		
		ViciAme	5.74 (±0.24) a	3.7 (±0.1) a		
	S * Litter (L)	AstrCic –L	0.72 (±0.40) d			
		DalePur –L	2.37 (±0.29) c			
		MediSat –L	2.08 (±0.43) cd			
		MeliOff –L	2.87 (±0.22) bc			
		TrifRep –L	0.34 (±0.30) d			
		ViciAme –L	5.49 (±0.35) a			
		AstrCic +L	1.17 (±0.45) d			
		DalePur +L	2.25 (±0.63) cd			
		MediSat +L	0.95 (±0.43) d			
		MeliOff +L	4.09 (±0.31) b			
		TrifRep +L	0.39 (±0.45) d			
		ViciAme +L	6.00 (±0.34) a			
CP-N	S	AstrCic	2.31 (±0.37) bc	2.9 (±0.2) bc	-	
		DalePur	2.74 (±0.22) b	3.0 (±0.1) b	-	
		MediSat	1.31 (±0.34) c	2.7 (±0.1) bc	-	
		MeliOff	3.29 (±0.20) b	3.2 (±0.1) ab	-	
		TrifRep	1.26 (±0.34) c	2.4 (±0.1) c	-	
		ViciAme	6.27 (±0.57) a	3.7 (±0.2) a	13.1 (±1.4) a	
CP-T	S	AstrCic	2.49 (±0.93) b	3.0 (±0.2) bc	5.2 (±4.2) b	3.3 (±0.6) b
		DalePur	3.07 (±0.55) b	3.1 (±0.1) b	-	-
		MediSat	2.78 (±0.57) b	3.0 (±0.1) bc	-	-
		MeliOff	7.17 (±0.65) a	3.5 (±0.1) a	24.5 (±3.7) a	8.8 (±0.5) a
		TrifRep	0.74 (±0.58) c	2.7 (±0.1) c	-	
		ViciAme	6.56 (±0.63) a	3.8 (±0.1) a	-	
	S * L	AstrCic –L	1.89 (±1.32) d			
		DalePur –L	2.48 (±0.72) cd			
		MediSat –L	2.83 (±0.78) cd			
		MeliOff –L	9.84 (±0.92) a			
		TrifRep –L	0.50 (±0.83) d			
		ViciAme –L	5.93 (±0.89) bc			
		AstrCic +L	3.09 (±1.26) cd			
		DalePur +L	3.66 (±0.74) c			
		MediSat +L	2.74 (±0.75) cd			
		MeliOff +L	4.51 (±0.86) c			
		TrifRep +L	0.98 (±0.72) d			
		ViciAme +L	7.19 (±0.83) b			

Lower caser letters distinguish Bonferroni corrected mean comparisons within a treatment, or combination thereof.

Table 7.14. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Dry Mixedgrass native site and their relationship to plant community characteristics.

Species	Factor	β-Estimate	SE	DF	T Value	P Value
Astragalus	Intercept	0.565	0.631	1.7	0.9	0.481
	Native Grass (%)	0.004	0.003	2.3	1.1	0.345
	Native Forb (%)	0.013	0.005	3.0	2.8	0.067
	Introduced Ruderal Forbs (%)	0.015	0.012	1.0	1.3	0.427
	Species Richness	-0.075	0.080	3.6	-0.9	0.405
	Shannon's Diversity	0.874	0.940	4.0	0.9	0.405
	Pielou's Evenness	-2.346	1.944	3.9	-1.2	0.245
	Bare Soil (%)	0.024	0.009	3.9	2.8	0.052
	Litter Cover (%)	0.000	0.003	3.8	0.1	0.910
	Litter Depth (cm)	-0.042	0.033	3.0	-1.3	0.293
	Lichen (%)	0.014	0.024	3.9	0.6	0.578
	Selaginella densa (%)	0.001	0.003	3.8	0.3	0.813
Dalea	Intercept	2.140	1.787	3.7	1.2	0.302
	Native Grass (%)	-0.009	0.012	2.4	-0.7	0.542
	Native Forb (%)	0.002	0.020	3.3	0.1	0.917
	Introduced Ruderal Forbs (%)	-0.014	0.014	3.9	-1.0	0.386
	Species Richness	-0.116	0.139	2.7	-0.8	0.470
	Shannon's Diversity	1.289	1.552	2.0	0.8	0.492
	Pielou's Evenness	-4.167	2.483	1.8	-1.7	0.252
	Bare Soil (%)	0.010	0.016	2.6	0.6	0.581
	Litter Cover (%)	0.010	0.011	2.0	0.8	0.489
	Litter Depth (cm)	0.029	0.032	3.2	0.9	0.424
	Lichen (%)	0.202	0.117	4.0	1.7	0.158
	Selaginella densa (%)	-0.002	0.014	2.7	-0.1	0.903
Medicago	Intercept	-1.795	1.128	3.2	-1.6	0.205
	Native Grass (%)	0.009	0.004	3.2	2.4	0.089
	Native Forb (%)	-0.008	0.007	3.2	-1.0	0.371
	Introduced Ruderal Forbs (%)	0.024	0.014	3.2	1.7	0.193
	Introduced Forages (%)	0.654	0.070	3.2	9.3	**0.002**
	Species Richness	0.019	0.086	3.2	0.2	0.836
	Shannon's Diversity	-0.979	0.930	3.2	-1.1	0.366
	Pielou's Evenness	2.296	2.182	3.2	1.1	0.367
	Bare Soil (%)	0.003	0.009	3.2	0.4	0.716
	Litter Cover (%)	0.014	0.010	3.2	1.4	0.256
	Litter Depth (cm)	-0.027	0.031	3.2	-0.9	0.440
	Lichen (%)	-0.038	0.071	3.2	-0.5	0.627
	Selaginella densa (%)	0.024	0.017	3.2	1.4	0.249
Melilotus	Intercept	-4.315	0.941	5.6	-4.6	**0.004**
	Native Grass (%)	0.007	0.002	5.6	3.6	**0.014**
	Native Forb (%)	-0.002	0.003	5.6	-0.6	0.548
	Introduced Ruderal Forbs (%)	0.043	0.019	5.6	2.2	0.072
	Species Richness	0.648	0.124	5.6	5.2	**0.002**
	Shannon's Diversity	-7.540	1.416	5.6	-5.3	**0.002**
	Pielou's Evenness	15.558	2.993	5.6	5.2	**0.002**
	Bare Soil (%)	-0.007	0.005	5.6	-1.3	0.239
	Litter Cover (%)	-0.016	0.002	5.6	-6.7	**0.001**
	Litter Depth (cm)	-0.067	0.021	5.6	-3.2	**0.021**
	Lichen (%)	-0.012	0.013	5.6	-1.0	0.380
	Selaginella densa (%)	-0.011	0.003	5.6	-3.2	**0.020**
Trifolium	Intercept	0.973	2.156	4.3	0.5	0.673
	Native Grass (%)	0.008	0.002	4.3	4.6	**0.008**
	Native Forb (%)	0.006	0.001	4.3	4.8	**0.007**
	Introduced Ruderal Forbs (%)	0.072	0.028	4.3	2.5	0.059
	Species Richness	-0.275	0.223	4.3	-1.2	0.281
	Shannon's Diversity	1.989	.2.63	4.3	0.8	0.489
	Pielou's Evenness	-3.897	5.554	4.3	-0.7	0.519
	Bare Soil (%)	-0.021	0.002	4.3	-13.9	**<0.001**
	Litter Cover (%)	0.007	0.003	4.3	2.5	0.062
	Litter Depth (cm)	-0.104	0.099	4.3	-1.1	0.346
	Lichen (%)	0.044	0.014	4.3	3.2	**0.031**
	Selaginella densa (%)	0.015	0.005	4.3	2.9	**0.042**
Vicia	Intercept	3.212	1.136	4.1	2.8	**0.047**
	Native Grass (%)	-0.002	0.004	4.1	-0.5	0.673
	Native Forb (%)	0.002	0.005	4.1	0.3	0.749
	Introduced Ruderal Forbs (%)	-0.011	0.017	4.1	-0.6	0.566
	Species Richness	-0.316	0.099	4.1	-3.2	**0.033**
	Shannon's Diversity	3.317	1.051	4.1	3.2	**0.034**
	Pielou's Evenness	-7.452	2.206	4.1	-3.4	**0.027**
	Bare Soil (%)	0.005	0.003	4.1	1.8	0.134
	Litter Cover (%)	0.000	0.002	4.1	-0.2	0.856
	Litter Depth (cm)	-0.001	0.027	4.1	0.0	0.972
	Lichen (%)	0.014	0.027	4.1	0.5	0.626
	Selaginella densa (%)	-0.001	0.003	4.1	-0.3	0.767

Table 7.15. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Dry Mixedgrass tame site and their relationship to plant community characteristics.

Species	Factor	β-Estimate	SE	DF	T Value	P Value
Astragalus	Intercept	1.192	1.009	5.8	1.2	0.284
	Native Forb (%)	-0.007	0.024	4.9	-0.3	0.788
	Introduced Ruderal Forbs (%)	-0.017	0.020	4.2	-0.8	0.444
	Introduced Forages (%)	-0.004	0.002	5.8	-2.6	**0.044**
	Species Richness	-0.043	0.089	4.5	-0.5	0.654
	Shannon's Diversity	0.347	0.661	4.2	0.5	0.626
	Pielou's Evenness	-0.579	0.952	4.5	-0.6	0.572
	Bare Soil (%)	-0.005	0.011	5.1	-0.5	0.660
	Litter Cover (%)	-0.005	0.011	5.0	-0.5	0.660
	Litter Depth (cm)	0.016	0.027	5.9	0.6	0.599
Dalea	Intercept	1.866	0.773	7.0	2.4	**0.047**
	Introduced Ruderal Forbs (%)	-0.033	0.021	7.0	-1.6	0.156
	Introduced Forages (%)	-0.002	0.003	7.0	-0.6	0.569
	Species Richness	0.038	0.094	7.0	0.4	0.699
	Shannon's Diversity	-0.073	0.659	7.0	-1.0	0.915
	Pielou's Evenness	-0.547	1.016	7.0	-0.5	0.607
	Bare Soil (%)	-0.019	0.007	7.0	-2.2	0.603
	Litter Cover (%)	-0.014	0.007	7.0	-2.1	0.073
	Litter Depth (cm)	-0.051	0.029	7.0	-1.8	0.123
Medicago	Intercept	-0.514	0.403	13.1	-1.3	0.225
	Native Forb (%)	0.080	0.049	13.1	1.6	0.126
	Introduced Ruderal Forbs (%)	0.028	0.027	13.1	1.0	0.315
	Introduced Forages (%)	0.008	0.002	13.1	4.2	**0.001**
	Species Richness	-0.031	0.016	13.1	-1.9	0.076
	Shannon's Diversity	0.001	0.022	13.1	0.0	0.973
	Pielou's Evenness	0.972	0.005	13.1	177.7	**<0.001**
	Bare Soil (%)	-0.001	0.007	13.1	-0.1	0.852
	Litter Cover (%)	-0.003	0.008	13.1	-0.4	0.661
	Litter Depth (cm)	-0.160	0.003	13.1	-43.0	**<0.001**
Melilotus	Intercept	-2.552	2.198	5.7	-1.2	0.292
	Native Forb (%)	-0.021	0.011	5.8	-1.9	0.106
	Introduced Ruderal Forbs (%)	-0.037	0.011	5.3	-3.3	**0.020**
	Introduced Forages (%)	-0.001	0.003	4.6	-0.4	0.703
	Species Richness	0.317	0.152	3.6	2.1	0.113
	Shannon's Diversity	-1.311	0.755	4.5	-1.7	0.150
	Pielou's Evenness	1.787	0.894	3.9	2.0	0.119
	Bare Soil (%)	0.021	0.026	5.1	0.8	0.469
	Litter Cover (%)	0.017	0.029	5.1	0.6	0.593
	Litter Depth (cm)	0.015	0.074	4.8	0.2	0.842
Trifolium	Intercept	2.827	1.714	6.0	1.6	0.150
	Native Forb (%)	-0.065	0.034	6.0	-1.9	0.106
	Introduced Ruderal Forbs (%)	-0.050	0.032	6.0	-1.8	0.121
	Introduced Forages (%)	-0.006	0.004	6.0	-1.4	0.212
	Species Richness	-0.270	0.253	6.0	-1.1	0.326
	Shannon's Diversity	2.784	1.859	6.0	1.5	0.185
	Pielou's Evenness	-5.019	2.734	6.0	-1.8	0.116
	Bare Soil (%)	-0.003	0.009	6.0	-0.4	0.730
	Litter Cover (%)	-0.003	0.009	6.0	-0.4	0.705
	Litter Depth (cm)	0.034	0.051	6.0	0.7	0.529
Vicia	Intercept	1.072	1.089	6.0	1.0	0.363
	Native Forb (%)	0.026	0.034	6.0	0.8	0.474
	Introduced Ruderal Forbs (%)	0.013	0.024	6.0	0.5	0.609
	Introduced Forages (%)	-0.001	0.004	6.0	-0.3	0.799
	Species Richness	-0.141	0.106	6.0	-1.3	0.232
	Shannon's Diversity	0.878	0.678	6.0	1.3	0.243
	Pielou's Evenness	-0.907	0.878	6.0	-1.0	0.341
	Bare Soil (%)	-0.005	0.009	6.0	-0.6	0.582
	Litter Cover (%)	-0.004	0.009	6.0	-0.5	0.618
	Litter Depth (cm)	-0.010	0.016	6.0	-0.7	0.537

Table 7.16. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Central Parkland's native fescue prairie site and their relationship to plant community characteristics.

Species	Factor	β-Estimate	SE	DF	T Value	P Value
Astragalus	Intercept	-0.626	0.946	2.1	-7.0	0.575
	Native Grass (%)	-0.019	0.008	2.9	-2.4	0.099
	Native Forb (%)	-0.010	0.006	3.5	-1.6	0.197
	Introduced Ruderal Forbs (%)	-0.043	0.019	1.5	-2.2	0.200
	Introduced Forages (%)	-0.009	0.009	3.9	-1.0	0.380
	Species Richness	0.226	0.085	2.8	2.7	0.081
	Shannon's Diversity	-2.525	1.180	2.6	-2.1	0.134
	Pielou's Evenness	6.230	2.790	2.8	2.2	0.118
	Bare Soil (%)	0.005	0.005	3.9	-0.9	0.411
	Litter Cover (%)	-0.006	0.006	4.0	0.9	0.400
	Litter Depth (cm)	0.027	0.051	3.8	0.5	0.628
	Lichen (%)	0.005	0.006	3.8	0.5	0.628
Dalea	Intercept	-1.124	4.268	4.0	-0.3	0.805
	Native Grass (%)	-0.029	0.015	4.0	-1.9	0.126
	Native Forb (%)	-0.032	0.022	4.0	-1.4	0.214
	Introduced Ruderal Forbs (%)	0.087	0.069	4.0	1.2	0.282
	Introduced Forages (%)	-0.028	0.024	4.0	-1.2	0.298
	Species Richness	0.412	0.310	4.0	1.3	0.253
	Shannon's Diversity	-4.648	3.721	4.0	-1.2	0.280
	Pielou's Evenness	10.959	9.747	4.0	1.1	0.324
	Bare Soil (%)	-0.004	0.010	4.0	-0.5	0.661
	Litter Cover (%)	-0.004	0.008	4.0	-0.6	0.596
	Litter Depth (cm)	0.213	0.345	4.0	0.6	0.571
	Lichen (%)	-0.019	0.021	4.0	-1.0	0.394
Medicago	Intercept	-8.611	3.332	4.0	-2.6	0.061
	Native Grass (%)	-0.022	0.013	4.0	-1.7	0.167
	Native Forb (%)	-0.025	0.015	4.0	-1.7	0.173
	Introduced Ruderal Forbs (%)	0.065	0.051	4.0	1.3	0.271
	Introduced Forages (%)	-0.020	0.016	4.0	-1.3	0.274
	Species Richness	1.011	0.341	4.0	3.0	**0.042**
	Shannon's Diversity	-0.123	4.173	4.0	-3.0	**0.042**
	Pielou's Evenness	0.291	9.682	4.0	3.0	**0.039**
	Bare Soil (%)	0.004	0.009	4.0	0.4	0.689
	Litter Cover (%)	0.000	0.009	4.0	0.0	0.998
	Litter Depth (cm)	0.194	0.132	4.0	1.5	0.217
	Lichen (%)	0.034	0.033	4.0	1.0	0.357
Melilotus	Intercept	12.720	1.408	4.5	9.0	**<0.001**
	Native Grass (%)	-0.010	0.005	4.5	-2.4	0.069
	Native Forb (%)	-0.020	0.005	4.5	-4.4	**0.009**
	Introduced Ruderal Forbs (%)	0.001	0.026	4.5	0.0	0.977
	Introduced Forages (%)	0.000	0.000	4.5	4.1	0.104
	Species Richness	-1.294	0.142	4.5	-9.1	**<0.001**
	Shannon's Diversity	14.610	1.712	4.5	8.5	**0.001**
	Pielou's Evenness	-30.970	3.883	4.5	-8.0	**0.001**
	Bare Soil (%)	-0.004	0.001	4.5	-3.8	**0.015**
	Litter Cover (%)	-0.006	0.002	4.5	-3.8	**0.015**
	Litter Depth (cm)	0.060	0.021	4.5	2.9	**0.038**
	Lichen (%)	-0.025	0.007	4.5	-3.7	**0.016**
Trifolium	Intercept	1.516	4.300	4.1	0.4	0.738
	Native Grass (%)	0.017	0.022	4.1	0.8	0.479
	Native Forb (%)	0.015	0.021	4.1	0.7	0.503
	Introduced Ruderal Forbs (%)	0.096	0.053	4.1	1.8	0.144
	Introduced Forages (%)	0.012	0.029	4.1	1.8	0.144
	Species Richness	-0.276	0.236	4.1	-1.2	0.308
	Shannon's Diversity	3.551	2.964	4.1	1.2	0.296
	Pielou's Evenness	-8.252	6.918	4.1	-1.2	0.298
	Bare Soil (%)	-0.006	0.008	4.1	-0.7	0.508
	Litter Cover (%)	-0.001	0.005	4.1	-0.2	0.829
	Litter Depth (cm)	-0.018	0.066	4.1	-0.3	0.791
	Lichen (%)	-0.001	0.014	4.1	-0.1	0.959
Vicia	Intercept	-3.288	4.642	4.2	-0.7	0.516
	Native Grass (%)	-0.010	0.010	4.2	-1.1	0.338
	Native Forb (%)	-0.018	0.006	4.2	-2.9	**0.039**
	Introduced Ruderal Forbs (%)	0.149	0.056	4.2	2.7	0.052
	Introduced Forages (%)	-0.010	0.011	4.2	-0.9	0.415
	Species Richness	0.366	0.371	4.2	1.0	0.378
	Shannon's Diversity	-3.942	4.014	4.2	-1.0	0.379
	Pielou's Evenness	10.120	9.460	4.2	1.1	0.342
	Bare Soil (%)	0.001	0.006	4.2	0.1	0.938
	Litter Cover (%)	-0.001	0.005	4.2	-0.2	0.886
	Litter Depth (cm)	0.009	0.054	4.2	0.2	0.872
	Lichen (%)	0.048	0.024	4.2	2.0	0.113

Table 7.17. Coefficients for mixed effects models evaluating the first-year germination of legume species seeded at the Central Parkland's tame site and their relationship to plant community characteristics.

Species	Factor	β-Estimate	SE	DF	T Value	P Value
Astragalus	Intercept	3.534	1.262	5.0	2.8	**0.038**
	Native Grass (%)	0.027	0.018	5.0	1.5	0.195
	Native Forb (%)	-0.007	0.006	5.0	-1.2	0.298
	Introduced Ruderal Forbs (%)	-0.081	0.040	5.0	-2.0	0.099
	Introduced Forages (%)	-0.011	0.006	5.0	-1.8	0.132
	Species Richness	-0.050	0.092	5.0	-0.5	0.609
	Shannon's Diversity	-0.069	0.682	5.0	-0.1	0.923
	Pielou's Evenness	-0.107	0.938	5.0	-0.1	0.914
	Bare Soil (%)	-0.024	0.007	5.0	-3.2	**0.023**
	Litter Cover (%)	-0.024	0.007	5.0	-3.4	**0.019**
	Litter Depth (cm)	0.026	0.019	5.0	1.3	0.239
Dalea	Intercept	-2.977	0.847	0.3	-3.5	0.477
	Native Grass (%)	-0.004	0.015	0.0	-0.3	0.955
	Native Forb (%)	0.002	0.003	0.1	0.5	0.865
	Introduced Ruderal Forbs (%)	0.037	0.011	0.0	3.3	0.905
	Introduced Forages (%)	0.006	0.003	0.3	2.0	0.593
	Species Richness	-0.115	0.059	0.1	-2.0	0.812
	Shannon's Diversity	0.869	0.433	0.1	2.0	0.836
	Pielou's Evenness	-0.263	0.497	0.1	-0.5	0.887
	Bare Soil (%)	0.030	0.008	0.1	3.8	0.774
	Litter Cover (%)	0.031	0.008	0.1	3.8	0.736
	Litter Depth (cm)	0.045	0.018	0.2	2.4	0.589
Medicago	Intercept	0.111	1.894	5.0	0.1	0.956
	Native Grass (%)	0.001	0.012	5.0	0.0	0.966
	Native Forb (%)	0.001	0.009	5.0	0.1	0.947
	Introduced Ruderal Forbs (%)	0.119	0.195	5.0	0.6	0.567
	Introduced Forages (%)	0.001	0.007	5.0	0.2	0.865
	Species Richness	0.066	0.231	5.0	0.3	0.787
	Shannon's Diversity	-0.517	1.907	5.0	-0.3	0.797
	Pielou's Evenness	1.208	2.324	5.0	0.5	0.626
	Bare Soil (%)	-0.004	0.017	5.0	-0.2	0.839
	Litter Cover (%)	-0.004	0.016	5.0	-0.3	0.806
	Litter Depth (cm)	0.011	0.044	5.0	0.2	0.822
Melilotus	Intercept	-0.908	0.571	5.9	-1.6	0.164
	Native Grass (%)	-0.062	0.021	5.9	-3.0	**0.025**
	Native Forb (%)	0.007	0.006	5.9	1.2	0.295
	Introduced Ruderal Forbs (%)	-0.003	0.007	5.9	-0.4	0.696
	Introduced Forages (%)	0.007	0.002	5.9	3.0	**0.024**
	Species Richness	0.106	0.051	5.9	2.1	0.086
	Shannon's Diversity	0.075	0.658	5.9	0.1	0.913
	Pielou's Evenness	-0.169	0.919	5.9	-0.2	0.860
	Bare Soil (%)	0.000	0.006	5.9	-0.1	0.966
	Litter Cover (%)	0.000	0.006	5.9	0.0	0.971
	Litter Depth (cm)	0.016	0.009	5.9	1.8	0.124
Trifolium	Intercept	-1.734	0.523	5.1	-3.3	**0.021**
	Native Grass (%)	-0.012	0.004	5.1	-3.3	**0.022**
	Native Forb (%)	-0.003	0.004	5.1	-0.5	0.646
	Introduced Ruderal Forbs (%)	-0.098	0.041	5.1	-2.4	0.059
	Introduced Forages (%)	0.009	0.002	5.1	3.6	**0.015**
	Species Richness	0.016	0.055	5.1	0.3	0.778
	Shannon's Diversity	0.620	0.558	5.1	1.1	0.317
	Pielou's Evenness	-0.307	0.815	5.1	-0.4	0.722
	Bare Soil (%)	0.016	0.004	5.1	4.3	**0.007**
	Litter Cover (%)	0.011	0.004	5.1	2.5	0.052
	Litter Depth (cm)	0.036	0.012	5.1	3.0	**0.029**
Vicia	Intercept	0.321	0.724	4.4	0.4	0.678
	Native Grass (%)	0.040	0.009	3.2	1.2	0.317
	Native Forb (%)	0.006	0.009	3.8	0.7	0.503
	Introduced Ruderal Forbs (%)	0.022	0.040	5.0	0.5	0.612
	Introduced Forages (%)	0.004	0.007	3.8	0.6	0.575
	Species Richness	0.018	0.031	4.8	0.6	0.583
	Shannon's Diversity	-0.024	0.202	5.0	-0.1	0.909
	Pielou's Evenness	0.011	0.014	4.1	0.8	0.240
	Bare Soil (%)	-0.009	0.006	4.1	-1.4	0.250
	Litter Cover (%)	-0.009	0.007	4.6	-1.3	0.248
	Litter Depth (cm)	0.047	0.026	2.2	1.8	0.203

Chapter 8

Synthesis of Seed Bank Research

8.1 Key Results

Seed banks are an important component of ecosystems, storing propagules in the top soil for revegetating disturbances and allowing for the sporadic recruitment of individuals in suitable microsites. Disturbance history and management of grasslands influences shifts in plant community composition between disturbance-tolerant species (i.e. introduced weeds to ruderal native forbs and graminoids) and desirable communities dominated by forages or late seral perennial grasses and forbs (i.e. seeded tame pastures to native grassland), which in turn, can influence the transient and persistent seed bank composition. Persistent seed banks formed over a long history of disturbance regimes (i.e. grazing history) or under acute disturbances (i.e. cultivation, herbicide application, or industrial disturbance) hold a record of disturbance legacy (Renne and Tracy 2007).

In some cases, we found legacy effects in the seed bank that were not expressed in the aboveground plant community, with numerous examples coming from Chapters 4 and 5. Timing of grazing, herbicide use, and manure spreading had significant effects on seed bank community composition that were not observed in the corresponding plant communities. The mechanism driving this divergence is likely linked to how these management actions effect seed production (i.e. grazing all year limits reproduction, or herbicide use reduces forbs) and seed inputs (i.e. manure can be a vector for seed introduction and influence soil properties), and indirect influences on seed bank formation and seed dormancy (i.e. soil compaction, seed entrapment by litter, etc.). One interesting example with a less understood mechanism was the divergent responses of vegetation and seed bank to indicators of fire history. Plant communities responded to recent indicators of fire identified in producer interviews (i.e. natural ignition/accidental, or prescribed), while seed banks differed based on historical indicators of fire (charcoal layer in the top 15 cm of mineral soil).

Seed banks can also evolve over time. Cultivation is a threat to native grasslands and significantly alters the composition of both seed banks and plant communities. We found that seeded areas provided greater time to form a seed bank since the last time the field was cultivated developed more desirable seed banks in the Parkland. These pastures contained less weedy and ruderal species, accumulated greater densities of graminoids, and even showed evidence of native perennial forbs returning. In our survey of pipelines in the Mixedgrass Prairie, disturbance age had less conspicuous effects on seed bank composition as legacy effects of disturbance were strong but did reflect trends in management (i.e. use of *Agropyron cristatum* on older and wider disturbances).

Seed bank composition was responsive to plant communities, edaphic factors, and elements of ground cover. In both the Parkland and Mixedgrass study sites, soil salinity and texture had strong associations with seed bank characteristics, where salinity was often associated with higher richness and texture was often associated with distinct plant communities that likely had differing seed inputs.

Similarity in richness (Sørenson's index) between plant communities and seed banks was examined in Chapters 5 and 6. We found mean similarity for Parkland seedbanks was 34.0 %, while that of Dry Mixedgrass prairie was 25.2%, indicating high dissimilarity in richness for grassland seed banks in Western Canadian grasslands. This appeared to be due to a few factors, including that seed banks in both studies had higher densities and representation of ruderal species. In the Parkland, where the grassland was dominated by forage grasses and legumes, we saw high densities of ruderal forbs, many of which were typically associated with annually seeded fields (i.e. *Chenopodium* spp., *Thlapsi arvense,* etc.) that had limited cover or representation in pastures with healthy, productive communities. Both native and introduced ruderals that accumulated high densities with limited cover aboveground are suspected to be species that form persistent seed banks (Kinucan and Smeins 1992). In both the Parkland and Dry Mixedgrass, we observed high densities of seeds from species adapted to mesic to hygric ecosites (i.e. *Juncus* spp., *Typha latifolia*, and wetland forbs from the Parkland like *Gnaphalium uliginosum*). There was also release of native ruderal graminoids and forbs which are similarly less competitive in established grassland communities. Perennial grasses were less likely to emerge, especially native grasses in the Dry Mixedgrass prairie, likely resulting from an ephemeral, transient seed bank (Kinucan and Smeins 1992).

Grasses like Kentucky bluegrass (*Poa praternsis*) formed a large seed bank, specifically in the Parkland and along pipelines in Dry Mixedgrass prairie, indicating that this species is likely to recover post disturbance or quickly occupy open niches. Also in the Parkland, smooth brome (*Bromus inermis*) is a productive, rhizomatous grass that provides forage for livestock that also invades native communities. We found limited germination from this species, meaning this species may not form a persistent seed bank. Native bromes were more likely to germinate despite limited observation of these species. In Dry Mixedgrass prairie, relatively desirable native grasses that formed a seed bank were blue grama (*Bouteloua gracilis*) and Junegrass (*Koeleria macrantha*) associated with non-disturbed soils and species rich, abundant biological crust cover. Emergence of later seral grasses like *Hesperostipa* spp., *Pascopyrum smithii*, and *Festuca hallii* were rarely observed, thus the recruitment of early to mid-seral grasses and sedges have the potential to revegetate a site with palatable, non-ruderal vegetation are desirable.

8.2 Implications

The Parkland study demonstrates that a wide variety of management actions and historical disturbances influence seed bank composition, with recent and intense disturbances creating a seed bank rich in propagules that are less desirable for recruitment. Cultivation of Parkland prairies has significantly altered these communities from their natural states and eliminated most native plant diversity, especially grasses. Similar to cultivation, evidence of historical fire was associated with strong legacy effects on seed bank composition, notably including the overall reduction of native and introduced ruderal forbs. Management actions that were associated with high management intensity or directly introduced propagules (e.g. manure, hay) influenced seed bank composition. Grazing systems did not influence seed banks (composition and density); this outcome was attributed to the occurrence of relatively uniformly high stocking rates regardless of grazing management. At a landscape level herbicide did not impede germinable legume seed bank populations.

Along pipelines, we found strong legacy effects on vegetation, with the persistence of agronomics in the plant community, some with invasive properties like *Poa pratensis, Agropyron cristatum,* and *Melilotus* spp. Seed banks were affected less distinctly, this is possibly due to legacy effects on seed bank extending beyond the area intensively studied. Ecosite significantly influenced Dry Mixed-grassland seed banks and interacted with aspects of pipeline disturbance. On loam ecosites, native graminoids seed densities were significantly reduced along pipelines, coarse-textured soils contained greater densities of native forbs, and salinity was associated with halophytes and ruderals. We recommend minimizing pipeline diameters as wider disturbances were associated with greater negative effects on plant community composition, seed banks, and biological crusts. Biological crusts exhibited very poor recovery along pipeline trenches and did not significantly recover with age, thus we urge minimizing disturbance to this fragile community layer and greater attention and research should be invested in aiding its recover. Seed banks are in-part influenced by the interaction between seed rain and soil surface, and biological crusts play an important role.

8.3 Future Research

There are a few main topics related to seed banks, grassland recovery, and disturbance I would like to see examined further from this research. First, with this current data set, additional questions related to seed ecology and behaviour could be examined. Plant reproductive strategies, fruits/seed dispersal mechanisms, and other diaspore traits (e.g. size, weight, seed coat thickness, etc.) likely influenced germinable seed bank observations in the green house (i.e. indurate seeds may have not germinated in the period of observation), seed bank formation, richness, similarity, and much more. Examination of reproductive strategies and seed traits could yield additional insights into seed bank ecology and the disturbance ecology of pastures and native rangelands in Canada. Seed ecology and dispersal mechanisms likely effected dynamics between plant communities and seed banks along pipeline disturbance.

Later on in my studies, I was exposed to studies examining bud banks (Ott et al. 2016), *in-situ* recruitment of seedlings from the existing seed bank in the field (Ren and Bai 2017), and treatments applied to seed bank soils (Ren and Bai 2016). Much like seed banks, buds provide opportunities for plant recruitment and are particularly important for the recruitment of perennial rhizomatous grasses (Klimes 2007). Bud banks have been shown to respond to disturbances like defoliation and changes in soil moisture and temperature, which influences the invasibility of introduced species like *Bromus inermis* and persistence of native grasses like *Pascopyrum smithii* (Ott et al. 2016). Further, observation of the *in-situ* seedling recruitment in our Parkland pasture survey, the pipeline survey, or in response to treatments imposed for the legume demography study would have greatly improved the data set available for analysis and provided insight into natural recruitment under the current disturbance regimes. Overall, the community assembly mechanisms that would affacet seedling recruitment and survival were not examined in this study. Studies exploring manipulative treatments on soil samples containing a seed bank are rare. Considering many seed bank studies are typically limited in the number of management factors or environments explored, this may be an interesting way to ask applied questions and impose unique conditions on emergent seedlings. Further, availability of soil resources often limit the recruitment of legumes into plant communities (Turnbull et al. 2005), it would have been interesting to pursue this question with seed bank samples. Experimental seed bank studies could have examined the influence of litter from a potentially allelopathic plant like *Melilotus* spp. (for example, see Wu et al. 2010) or examined its potential to interfere with recruitment. The influence of residual herbicides on germinable seed bank recruitment could have also been examined experimentally.

Disturbances influence seed banks in a few main ways, primarily through shifts in plant community composition and influencing the reproductive potential of plants in the species pool or by altering the environment (soil properties, soil cover, etc.) in which the seed enters the seed bank.

Examining the influence of disturbance on mechanisms regulating seed bank formation could further improve our understanding of managing seed banks for species of concern, both positive and negative. Notably, this research also showed a link between seed banks and biological soil crusts, the relationship of which is understudied and warrants further study to better understand its complexity.

Additionally, in retrospect I would have preferred to develop a more detailed producer survey. There were apparent limitations in the story the data could tell based on the questions asked and the survey design. In a highly populated and intensively managed landscape around an urban centre it would have been beneficial if more sociological information were collected in addition to management philosophy. These factors likely had an influence on pasture management (i.e. decisions to use continuous vs. rotational grazing) which could have indirectly influenced rangeland health scores.

8.4 Conclusion

These surveys supplemented an apparent knowledge gap in seed bank composition and diversity hidden in the soil of managed pastures and native grassland. Diverse disturbances beyond grazing were found to influence the seed densities of important functional plant groups like legumes, weedy species, and canopy-dominant perennial grasses. Disturbance legacies were an important factor shaping seed banks of pastures examined from the Parkland-Boreal region, while certain management inputs (manure, bale grazing, etc.) had diverse influences on both the seedbank and existing vegetation. In xeric Dry Mixedgrass prairies, disturbance legacies associated with pipelines were associated with the introduction and persistence of undesirable species in the seed bank. Additionally, pipelines were associated with long-term reductions in biological soil crust cover, which in turn was linked to shifts in seed bank composition.

8.5 Literature Cited

Klimes, J. 2007. Bud banks and their role in vegetative regeneration–a literature review and proposal for simple classification and assessment. Perspectives in Plant Ecology, Evolution and Systematics **8**(3):115-129.

Ott, J.P., Butler, J.L., Rong, Y., and Xu, L. 2016. Greater bud outgrowth of *Bromus inermis* than *Pascopyrum smithii* under multiple environmental conditions. Journal of Plant Ecology **10**(3):518-527.

Ren, L. and Bai, Y. 2016a. Smoke and ash effects on seedling emergence from germinable soil seed bank in Fescue Prairie. Rangeland Ecology and Management **69**(6):499-507.

Ren, L. and Bai, Y. 2017. Burning modifies composition of emergent seedlings in fescue prairie. Rangeland Ecology and Management **70**(2):230-237.

Renne, I.J. and Tracy, B.F. 2007. Disturbance persistence in managed grasslands: shifts in aboveground community structure and the weed seed bank. Plant Ecology **190**(1):71-80.

Turnbull, L.A., Rahm, S., Baudois, O., Eichenberger-Glinz, S., Wacker, L. and Schmid, B. 2005. Experimental invasion by legumes reveals non-random assembly rules in grassland communities. Journal of Ecology **93**(6):1062-1070.

Wu, C.X., Guo, X.X., Li, Z.H. and Shen, Y.X. 2010. Feasibility of using the allelopathic potential of yellow sweet clover for weed control. Allelopathy Journal **25**(1):173-183.

R Packages Used

ade4

Dray, S., and Dufour, A.B. 2007. The ade4 package: implementing the duality diagram for ecologists. Journal of Statistical Software **22**(4):1-20.

agricolae

De Mendiburu, F. 2017. agricolae: Statistical Procedures for Agricultural Research. R package version 1.2-6.
https://CRAN.R-project.org/package=agricolae

colorRamps

Keitt, T. 2012. colorRamps: Builds color tables. R package version 2.3.
http://CRAN.R-project.org/package=colorRamps

ecodist

Goslee, S.C. and Urban, D.L. 2007. The ecodist package for dissimilarity-based analysis of ecological data. Journal of Statistical Software **22**(7):1-19.

FactoMineR

Le, S., Josse, J., and Husson, F. 2008. FactoMineR: An R package for multivariate analysis. Journal of Statistical Software **25**(1):1-18. 10.18637/jss.v025.i01

indicspecies

De Caceres, M., Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. Ecology, URL
http://sites.google.com/site/miqueldecaceres/

lme4

Bates, D., Maechler, M., Bolker, B., and Walker, S. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software **67**(1):1-48. doi:10.18637/jss.v067.i01.

lmerTest

Kuznetsova, A., Brockhoff, P.B., and Christensen, R.H.B. 2016. lmerTest: Tests in linear mixed effects models. R package version 2.0-33.
https://CRAN.R-project.org/package=lmerTest

lsmeans

Lenth, R.V. 2016. Least-Squares Means: The R Package lsmeans. Journal of Statistical Software **69**(1): 1-33. doi:10.18637/jss.v069.i01 *nlme*

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., and Tb. 2017. nlme: Linear and nonlinear mixed effects models. R package version 3.1-131
https://CRAN.R-project.org/package=nlme

nortest

Gross, J. and Ligges, U. 2015. nortest: Tests for normality. R package version 1.0-4.
https://CRAN.R-project.org/package=nortest

vegan

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, K., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson. G.L., Solymos, P., M., Stevens, M.H., Szoecs, E., and Wagner, H. 2017. vegan: Community ecology package. R package version 2.4-4.
https://CRAN.R-project.org/package=veg

Bibliography

AbaData^tm^ Oil and Gas Map Software. 2016. Accessed December 2016. http://www.abacusdatagraphics.com/abadata.asp?gclid=CjwKEAiArIDFBRCe_9DJi6Or0UcSJAAK1nFvOjqjBaXTWN8BJLqkQfFOijSTxvv8qZDLR9U-vUjfgxoC-Czw_wcB

Abu, Y., Romo, J. T., Bai, Y. and Coulman, B. 2016. Priming seeds in aqueous smoke solutions to improve seed germination and biomass production of perennial forage species. Canadian Journal of Plant Science **96**(4):551-563.

[ACIMS] Alberta Conservation Information Management System. 2015. 2015 ACIMS Plant Species Ranking. Accessed October 2, 2017.

Acharya, S.N., Kastelic, J.P., Beauchemin, K.A., and Messenger, D.F. 2006. A review of research progress on cicer milkvetch (*Astragalus cicer* L.). Canadian Journal of Plant Science **86**(1):49-62.

Acharya, S.N., Kokko, E.G. and Fraser, J. 1993. Storage duration and freeze-thaw effects on germination and emergence of cicer milkvetch (*Astragalus cicer*) seeds. Journal of Seed Technology **17**(2):9-21.

Adams, B. W., Ehlert, G., Stone, C., Lawrence, D., Alexander, M., Willoughby, M., Hincz, C., Moisey, D., Burkinshaw, A., Carlson, J. and France, K. 2005. Rangeland health assessment for grassland, forest and tame pasture. Public Lands and Forests Division, Alberta Sustainable Resource Development.

[AER] Alberta Energy Regulator. 2015. Statistical Series 59: Alberta Drilling Activity Monthly Statistics. Date Modified: 2015-04-28. https://osi.alberta.ca/osicontent/Pages/OfficialStatistic.aspx?ipid=948

[AER] Alberta Energy Regulator. 2016. One Stop: Reclamation Certificate Mapping Tool. Date Accessed: 2016-10. https://extmapviewer.aer.ca/Onestop/RecCert/public/index.html

Agnew, W., Uresk, D. W., and Hansen, R. M. 1986. Flora and fauna associated with prairie dog colonies and adjacent ungrazed mixed-grass prairie in western South Dakota. Journal of Range Management **39**(2):135-139.

Agriculture Canada. 2014. Human Activity and the Environment: Agriculture in Canada. Statistics Canada Catalogue no.16-201-X.

Alberta Agriculture and Forestry. 2016. AgroClimatic Information Service. Accessed May 10, 2016 for -University of Alberta's South Campus weather station. http://agriculture.alberta.ca/acis/alberta-weather-data-viewer.jsp

Albrecht, H. and Auerswald, K. 2003. Arable weed seedbanks and their relation to soil properties. Aspects of Applied Biology **69**:11-20.

Allden, W. G. and McDWhittaker, I. A., 1970. The determinants of herbage intake by grazing sheep: the interrelationship of factors influencing herbage intake and availability. Crop and Pasture Science **21**(5):755-766.

Allred, B.W., Smith, W.K., Twidwell, D., Haggerty, J.H., Running, S.W., Naugle, D.E., and Fuhlendorf, S.D. 2015. Ecosystem services lost to oil and gas in North America: Net primary production reduced in crop and rangeland. Science **348**(6233):401-402.

Ambrose, L.G. and Wilson, S.D. 2003. Emergence of the Introduced Grass *Agropyron cristatum* and the Native Grass *Bouteloua gracilis* in a Mixed-grass Prairie Restoration. Restoration Ecology **11**(1):110-115.

Ambrosio, L., Iglesias, L., Marin, C., and Del Monte, J.P. 2004. Evaluation of sampling methods and assessment of the sample size to estimate the weed seedbank in soil, taking into account spatial variability. Weed Research **44**:224-236.

Anderson, M. J. 2005. Permutational multivariate analysis of variance. Department of Statistics, University of Auckland **26**:32-46.

Anderson, H. G. and Bailey, A. W. 1980. Effects of annual burning on grassland in the aspen parkland of east-central Alberta. Canadian Journal of Botany **58**(8):985-996.

Anderson, D.C., Harper, K.T. and Holmgren, R.C. 1982. Factors influencing development of cryptogamic soil crusts in Utah deserts. Rangeland Ecology and Management **35**(2):180-185.

Anderson, J.D., Ingram, L.J. and Stahl, P.D. 2008. Influence of reclamation management practices on microbial biomass carbon and soil organic carbon accumulation in semiarid mined lands of Wyoming. Applied Soil Ecology **40**(2):387-397.

Aniszewski, T. 2010. Canopy behavior of three milkvetch (*Astragalus*) species in acclimation to new habitat. Acta Biologica Cracoviensia Series Botanica **52**(1):45-54.

Archibold, O.W., Ripley, E.A., and Delanoy, L. 2003. Effects of season burning on the microenvironment of fescue prairie in central Saskatchewan. Canadian Field-Naturalist **117**(2):257-266.

Aydin, I., and Uzun, F. 2005. Nitrogen and Phosphorus fertilization of rangelands affects yield, forage quality and the botanical composition. European Journal of Agronomy **23**(1):8-14.

Badger, K.S. and Ungar, I.A. 1994. Seed bank dynamics in an inland salt marsh, with special emphasis on the halophyte *Hordeum jubatum* L. International Journal of Plant Sciences **155**(1):66-72.

Bagavathiannan, M.V., Gulden, R.H., Begg, G.S. and Van Acker, R.C. 2010. The demography of feral alfalfa (*Medicago sativa* L.) populations occurring in roadside habitats in Southern Manitoba, Canada: implications for novel trait confinement. Environmental Science and Pollution Research **17**(8):1448-1459.

Bagavathiannan, M.V., Gulden, R.H. and Van Acker, R.C. 2011. The ability of alfalfa (*Medicago sativa*) to establish in a seminatural habitat under different seed dispersal times and disturbance. Weed Science **59**(3):314-320.

Bagavathiannan, M.V. and Van Acker, R.C. 2009. The biology and ecology of feral alfalfa (*Medicago sativa* L.) and its implications for novel trait confinement in North America. Critical Reviews in Plant Sciences **28**(1-2):69-87.

Bailey, A. W. and Anderson, M. L. 1978. Prescribed burning of a *Festuca-Stipa* grassland. Journal of Range Management **31**(6):446-449.

Bailey, A. W., Irving, B. D. and Fitzgerald, R. D. 1990. Regeneration of woody species following burning and grazing in aspen parkland. Journal of range management **43**(3):212-215.

Bailey, A.W., McCartney, D., and Schellenberg, M.P. 2010. Management of prairie rangeland. Agriculture and Agri-Food Canada.

Bailey, A.W., and Wroe, R.W. 1974. Aspen invasion in a portion of the Alberta Parklands. Journal of Range Management **27**(4):263-266.

Bakker, J.P., Poschlod, P., Strykstra, R.J., Bekker, R.M., Thompson, K. 1996. Seed banks and seed dispersal: important topics in restoration ecology. Acta Botanica Neerlandica **45**(4):461-490.

Ball, D.A. 1992. Weed seedbank response to tillage, herbicides, and crop rotation sequence. Weed Science **40**(4):654-659.

Baltanás, A. 1992. On the use of some methods for the estimation of species richness. Oikos **65**(3):484-492.

Baron, V.S., Dick, A.C., Mapfumo, E., Malhi, S.S., Naeth, M.A., and Chanasyk, D.S. 2001. Grazing impacts on soil nitrogen and phosphorus under parkland pastures. Journal of Range Management **54**(6):704-710.

Barnes, P. W., Throop, H. L., Hewins, D. B., Abbene, M. L. and Archer, S. R. 2012. Soil coverage reduces photodegradation and promotes the development of soil-microbial films on dryland leaf litter. Ecosystems **15**(2): 311-321.

Baron, V.S., Dick, A.C., Mapfumo, E., Malhi, S.S., Naeth, M.A., and Chanasyk, D.S. 2001. Grazing impacts on soil nitrogen and phosphorus under parkland pastures. Journal of Range Management **54**(6):704-710.

Baron, V.S., Mapfumo, E., Dick, A.C., Naeth, M.A., Okine, E.K., and Chanasyk, D.S. 2002. Grazing intensity impacts on pasture carbon and nitrogen flow. Journal of Range Management **55**(6):535-541.

Barret, J. P. and Silander Jr., J.A. 1992. Seedling recruitment limitation in white clover (*Trifolium repens*; Leguminosae). American Journal of Botany **79**(6):643-649.

Baskin. J.M., Baskin, C.C., and Li, X. 2000. Taxonomy, anatomy and evolution of physical dormancy in seeds. Plant Species Biology **15**:139-152.

Beare, M. H., Parmelee, R. W., Hendrix, P. F., Cheng, W., Coleman, D. C. and Crossley, D. A. 1992. Microbial and faunal interactions and effects on litter nitrogen and decomposition in agroecosystems. Ecological Monographs **62**(4):569-591.

Bekker, R.M., Verweij, G.L., Smith, R.E.N., Reine, R., Bakker, J.P., and Schneider, S. 1997. Soil seed bank in European grasslands: does land use affect regeneration perspectives? Journal of Applied Ecology **34**:1293-1310.

Bell, F. and Nutman, P.S. 1971. Experiments on nitrogen fixation by nodulated lucerne. Plant and Soil **35**(1):231-264.

Belnap, J. 1995. Surface disturbances: their role in accelerating desertification. In Desertification in Developed Countries. Springer, Netherlands. pp. 39-57.

Belnap, J. 2003. The world at your feet: desert biological soil crusts. Frontiers in Ecology and the Environment **1**(4):181-189.

Belnap, J., Büdel, B. and Lange, O.L. 2001a. Biological soil crusts: characteristics and distribution. In Biological soil crusts: structure, function, and management. Springer Berlin Heidelberg. pp 3-30.

Belnap, J. and Eldridge, D. 2001. Disturbance and recovery of biological soil crusts. In Biological soil crusts: structure, function, and management. Springer Berlin Heidelberg. pp. 363-383.

Belnap, J., Phillips, S.L. and Troxler, T. 2006. Soil lichen and moss cover and species richness can be highly dynamic: the effects of invasion by the annual exotic grass *Bromus tectorum*, precipitation, and temperature on biological soil crusts in SE Utah. Applied Soil Ecology **32**(1):63-76.

Belnap, J., Prasse, R. and Harper, K.T. 2001b. Influence of biological soil crusts on soil environments and vascular plants. Biological soil crusts: structure, function, and management. Springer Berlin Heidelberg. pp. 281-300.

Benoit, D. L., N. C. Kenkel, and Cavers, P.B. 1989. Factors influencing the precision of soil seed bank estimates. Canadian Journal of Botany **67**:2833-2840.

Benvenuti, S. 2007. Natural weed seed burial: effect of soil texture, rain and seed characteristics. Seed Science Research **17**(3):211-219.

Bertiller, M.B. and Ares, J.O. 2011. Does sheep selectivity along grazing paths negatively affect biological crusts and soil seed banks in arid shrublands? A case study in the Patagonian Monte, Argentina. Journal of Environmental Management **92**(8):2091-2096.

Bigwood, D. W. and Inouye, D.W. 1988. Spatial pattern analysis of seed banks: an improved method and optimized sampling. Ecology **69**(2):497-507.

[BLM] Bureau of Land Management. 2001. Biological soil crusts: Ecology and Management, Technical Reference 1730-2. Denver, CO, USA.

Blonski, L.J., Bork, E.W., Blenis, P.V. 2004. Herbage yield and crude protein concentration of rangeland and pasture following hog manure application in southeastern Alberta. Canadian Journal of Plant Science **84**(3):773-783.

Boeken, B. and Shachak, M., 1998. The dynamics of abundance and incidence of annual plant species during colonization in a desert. Ecography **21**(1):63-73.

Bonsal, B. and Regier, M. 2007. Historical comparison of the 2001/2002 drought in the Canadian Prairies. Climate Research **33**(3):229-242.

Booth, B.D., and Swanton, C.J. 2002. 50th anniversary—invited article: Assembly theory applied to weed communities. Weed Science **50**:2-13.

Borcard, D., Gillet, F., and Legendre, P. 2011. Numerical ecology with R. Springer Science and Business Media, New York, USA.

Bork, E. 1993. Interaction of burning and herbivory in aspen communities in Elk Island National Park, Alberta. M.Sc. Thesis, University of Alberta, Edmonton.

Bork, E.W. 2015. Baseline quantification of carbon stored in Alberta rangelands. Final report to the Alberta Livestock and Meat Agency Ltd, Research and Development Program.

Bork, E. W., Adams, B. W., and Willms, W. D. 2002. Resilience of foothills rough fescue, Festuca campestris, rangeland to wildfire. Canadian Field-Naturalist **116**(1):51-59.

Bork, E.W., and Blonski, L.J. 2012. Short-term native grassland compositional responses following liquid hog manure application. Canadian Journal of Plant Science **92**(1):55-65.

Bork, E. W., Grekul, C. W., and De Bruijn, S. L. 2007. Extended pasture forage sward responses to Canada thistle (*Cirsium arvense*) control using herbicides and fertilization. Crop Protection **26**(10):1546-1555.

Bork, E.W., Hewins, D.B., Tannas, S. and Willms, W.D. 2017. *Festuca campestris* density and defoliation regulate abundance of the rhizomatous grass *Poa pratensis* in a fallow field. Restoration Ecology **26**(1):82-90.

Bork, E. W., Lambert, B. D., Banerjee, S. and Blonski, L. J. 2013. Soil mineral nitrogen responses following liquid hog manure application to semiarid forage lands. Canadian Journal of Soil Science **93**(3):369-378.

Bouyoucos, G.J. 1927. The hydrometer method as a new method for the mechanical analysis of soils. Soil Science **23**(5):343-354.

Bowes, G. G. 1981. Improving aspen poplar and prickly rose-covered rangeland with herbicides and fertilizer. Canadian Journal of Plant Science **61**(2):401-405.

Bowes, G. G. 1982. Changes in the yield of forage following the use of herbicides to control aspen poplar. Journal of Range Management **35**(2):246-248.

Bowes, G.G., and Spurr, D.T. 1996. Control of aspen, balsam poplar, prickly rose and western snowberry with metsulfuron-methyl and 2, 4-D. Canadian Journal of Plant Science **76**(4):885-889.

Bowker, M.A. 2007. Biological soil crust rehabilitation in theory and practice: an underexploited opportunity. Restoration Ecology **15**(1):13-23.

Briske, D.D. 1996. Strategies of plant survival in grazed systems: a functional interpretation. The Ecology and Management of Grazing Systems. pp 37-67.

Briske, D.D., Derner, J.D., Brown, J.R., Fuhlendorf, S.D., Teague, W.R., Havstad, K.M., Gillen, R.L., Ash., A.J., and Willms, W.D. 2008. Rotational grazing on rangelands: Reconciliation of perception and experimental evidence. Rangeland Ecology and Management **61**:3-17.

Briske, D.D. Fuhlendorf, S.D., and Smeins, F.E. 2005. State-and-transition models, thresholds, and rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management **58**:1-10.

Brummer, E.C. and Moore, K.J. 2000. Persistence of perennial cool-season grass and legume cultivars under continuous grazing by beef cattle. Agronomy Journal **92**(3):466-471.

Buhler, D.D., Hartzler, R.G., and Forcella, F. 1997. Implications of weed seedbank dynamics to weed management. Weed Science **45**:329-336.

Bullock, J.M., Hill, B.C., Dale, M.P. and Silvertown, J. 1994. An experimental study of the effects of sheep grazing on vegetation change in a species-poor grassland and the role of seedling recruitment into gaps. Journal of Applied Ecology **31**(3):493-507.

Burgess, C. P., Chapman, R., Singleton, P. L. and Thom, E. R. 2000. Shallow mechanical loosening of a soil under dairy cattle grazing: Effects on soil and pasture. New Zealand Journal of Agricultural Research **43**(2):279-290.

Burity, H.A., Ta, T.C., Faris, M.A., and Coulman, B.E. 1989. Estimation of nitrogen fixation and transfer from alfalfa to associated grasses in mixed swards under field conditions. Plant and Soil **114**(2):249-255.

Burns, J.H. 2014. To what degree are invaders drivers or passengers of phylogenetic community structure? Journal of Vegetation Science **25**(6):1311-1312.

Bylo, L. N., Koper, N., and Molloy, K. A. 2014. Grazing intensity influences ground squirrel and American badger habitat use in mixed-grass prairies. Rangeland Ecology and Management **67**(3):247-254.

Carlson, J.M., and Crist, T.O. 1999. Plant responses to pocket-gopher disturbances and topography. Journal of Range Management **52**(6):637-645.

Carr, P.M., Poland, W.W., and Tisor, L.J. 2005. Forage legume regeneration from the soil seed bank in western North Dakota. Agronomy Journal **97**(2):505-513.

Carren, C.J., Wilson, A.M., Cuany, R.L. and Thor, G.L. 1987. Caryopsis weight and planting depth of blue grama I. Morphology, emergence, and seedling growth. Journal of Range Management **40**(3):207-211.

Caswell, H. 1989. Matrix population models: construction, analysis, and interpretation. Sinauer Associates Inc., Sunderland, Massachusetts.

Chorney, B. and Josephson, R. 2000. A survey of pasture management on the Canadian prairies with emphasis on rotational grazing and managed riparian areas. M. Sc. Thesis, University of Manitoba, Department of Agricultural Economics and Farm Management, Winnipeg, Manitoba.

Christian, J.M. and Wilson, S.D. 1999. Long-term ecosystem impacts of an introduced grass in the northern Great Plains. Ecology **80**:2397-2407.

Cialdella, N., Dobremez, L., and Madelrieux, S. 2009. Livestock farming systems in urban mountain regions: Differentiated paths to remain in time. Outlook on Agriculture **38**(2):127-135.

Clements, D.R., Krannitz, P.G., and Gillespie, S.M. 2007. Seed bank responses to grazing history by invasive and native plant species in a semi-desert shrub-steppe environment. Northwest Science **81**(1):37-49.

Coffin, D.P. and Laurenroth, W.K. 1989. Spatial and temporal variation in the seed bank of a semiarid grassland. American Journal of Botany **76**(1):53-58.

Cole, D.N. 1990. Trampling disturbance and recovery of cryptogamic soil crusts in Grand Canyon National Park. The Great Basin Naturalist **50**(4):321-325.

Conant, R.T., Paustian, K. and Elliott, E.T. 2001. Grassland management and conversion into grassland: effects on soil carbon. Ecological Applications **11**(2):343-355.

Connell, J.H. 1978. Diversity in tropical rain forests and coral reefs. Science **199**(4335):1302-1310.

Connell, J.H. and Slayter, R.O. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. The American Naturalist **111**:1119-1144.

Coupland, R.T., and Brayshaw, T.C.1953. The fescue grassland in Saskatchewan. Ecology **34**(2):386-405.

Cournane, F.C., McDowell, R.W., Littlejohn, R.P, Houlbrooke, D.J, and Condron, L.M. 2011. Is mechanical soil aeration a strategy to alleviate soil compaction and decrease phosphorus and suspended sediment losses from irrigated and rain-fed cattle-grazed pastures? Soil Use and Management **27**(3):376-384.

Cox, R. D. and Allen, E. B. 2008. Composition of Soil Seed Banks in Southern California Coastal Sage Scrrub and Adjacent Exotic Grassland. Plant Ecology **198**(1):37-46.

Cramer, V. A., Hobbs, R. J. and Standish, R. J. 2008. What's new about old fields? Land abandonment and ecosystem assembly. Trends in Ecology and Evolution **23**(2):104-112.

Culley, J.L.B., Dow, B.K., Presant, E.W. and MacLean, A.J. 1982. Recovery of productivity of Ontario soils disturbed by an oil pipeline installation. Canadian Journal of Soil Science **62**(2):267-279.

Damschen, E.I., Brudvig, L.A., Haddad, N.M., Levey, D.J., Orrock, J.L., and Tewksbury, J.J. 2008. The movement ecology and dynamics of plant communities in fragmented landscapes. Proceedings of the National Academy of Sciences of the United States of America **105**:19078-19083.

D'Antonio, C. and Meyerson, L.A. 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restoration Ecology **10**(4):703-713.

Dastgheib, F. 1989. Relative importance of crop seed, manure and irrigation water as sources of weed infestation. Weed Research **29**(2):113-116.

Davis, M.A. and Pelsor, M. 2001. Experimental support for a resource-based mechanistic model of invasibility. Ecology Letters **4**(5):421-428.

De Bruijn, S.L., and Bork, E.W. 2006. Biological control of Canada thistle in temperate pastures using high density rotational cattle grazing. Biological Control **36**(3):305-315.

De Bruijn, S.L., Bork, E.W. and Grekul, C.W. 2010. Neighbor defoliation regulates Canada thistle (*Cirsium arvense*) in pasture by mediating interspecific competition. Crop Protection **29**(12):1489-1495.

De Keyser, E.S., Dennhardt, L.A., and Hendrickson, J. 2015. Kentucky bluegrass (*Poa pratensis*) invasion in the Northern Great Plains: a story of rapid dominance in an endangered ecosystem. Weed Science **64**(3):409-420.

Deák, B., Valkó, O., Kelemen, A., Török, P., Miglécz, T., Ölvedi, T., Lengyel, S., and Tóthmérész, B. 2011. Litter and graminoid biomass accumulation suppresses weedy forbs in grassland restoration. Plant Biosystems - An International Journal Dealing with all Aspects of Plant Biology **145**(3):730-737.

Delach, A. and Kimmerer, R.W. 2002. The effect of *Polytrichum piliferum* on seed germination and establishment on iron mine tailings in New York. The Bryologist **105**(2):249-255.

Desserud, P., Gates, C.C., Adams, B. and Revel, R.D. 2010. Restoration of foothills rough fescue grassland following pipeline disturbance in southwestern Alberta. Journal of Environmental Management **91**(12):2763-2770.

Desserud, P.A. and Hugenholtz, C. 2015. Do three invasive species: *Amaranthus blitoides*, *Descurainia sophia* and *Bassia scoparia*, respond to soil properties? Ecological Restoration **33**(2):127-130.

Desserud, P.A. and Hugenholtz, C.H. 2017. Restoring industrial disturbances with native hay in Mixedgrass Prairie in Alberta. Ecological Restoration **35**(3):228-236.

Desserud, P.A. and Naeth, M.A. 2011. Promising results restoring grassland disturbances with native hay (Alberta). Ecological Restoration **29**(3):215-219.

Desserud, P.A, and Naeth, M.A. 2013. Natural recovery of rough fescue (*Festuca hallii* (Vasey) Piper) grassland after disturbance by pipeline construction in central Alberta, Canada. Natural Areas Journal **33**(1):91-98.

Desserud, P.A. and Naeth, M.A. 2014. Predicting grassland recovery with a state and transition model in a natural area, Central Alberta, Canada. Natural Areas Journal **34**(4):429-442.

Deutsch, E. S., Bork, E. W. and Willms, W. D. 2010a. Separation of grassland litter and ecosite influences on seasonal soil moisture and plant growth dynamics. Plant Ecology **209**(1):135-145.

Deutsch, E. S., Bork, E. W. and Willms, W. D., 2010b. Soil moisture and plant growth responses to litter and defoliation impacts in Parkland grasslands. Agriculture, Ecosystems & Environment **135**(1):1-9.

DiVittorio, C.T., Corbin, J.D. and D'Antonio, C.M. 2007. Spatial and temporal patterns of seed dispersal: an important determinant of grassland invasion. Ecological Applications **17**(2):311-316.

Dixon, J.M. 2000. *Koeleria macrantha* (Ledeb.) Schultes (*K. alpigena* Domin, *K. cristata* (L.) Pers. pro parte, *K. gracilis* Pers., *K. albescens* auct. non DC.). Journal of Ecology **88**(4):709-726.

Dodds, W.K., Gudder, D.A. and Mollenhauer, D. 1995. The ecology of *Nostoc*. Journal of Phycology **31**(1):2-18.

Donkor, N.T., Gedir, J.V., Hudson, R.J., Bork, E.W., Chanasyk, D.S., and Naeth, M.A. 2002. Impacts of grazing systems on soil compaction and pasture production in Alberta. Canadian Journal of Plant Science **82**(1):1-8.

Dormaar, J.F., Adams, B.W., and Willms, W.D. 1997. Impacts of rotational grazing on mixed prairie soils and vegetation. Journal of Range Management **50**(6):647-651.

Dornier, A., Pons, V., and Cheptou, O. 2011. Colonization and extinction dynamics of an annual plant population in an urban environment. Oikos **120**:1240-1246.

Dutt, T. E., Harvey, R. G. and Fawcett, R. S. 1982. Feed quality of hay containing perennial broadleaf weeds. Agronomy Journal **74**(4):673-676.

Dyksterhuis, E.J. 1949. Condition and management of rangeland based on quantitative ecology. Journal of Range Management **2**(3):104-115.

Edwards, A. R., and Younger, A. 2006. The dispersal of traditionally managed hay meadow plants via farmyard manure application. Seed Science Research **16**(02):137-147.

Elsinger, M.E. 2009. Reclamation status of plains rough fescue grasslands at Rumsey Block after well site and pipeline disturbance. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Emery, S.M. and Gross, K.L. 2007. Dominant species identity, not community evenness, regulates invasion in experimental grassland plant communities. Ecology **88**(4):954-964.

Entz, M.H., Bullied, W.J., and Katepa-Mupondwa, F. 1995. Rotational benefits of forage crops in Canadian prairie cropping systems. Journal of Production Agriculture **8**(4):521-529.

Eriksson, A. and Eriksson, O. 1997. Seedling recruitment in semi-natural pastures: the effects of disturbance, seed size, phenology and seed bank. Nordic Journal of Botany **17**(5):469-480.

Eschtruth, A.K. and Battles, J.J. 2009. Assessing the Relative Importance of Disturbance, Herbivory, Diversity, and Propagule Pressure in Exotic Plant Invasion. Ecological Monographs **79**:265-280.

Evans, R.D. and Belnap, J. 1999. Long-term consequences of disturbance on nitrogen dynamics in an arid ecosystem. Ecology **80**(1):150-160.

Evans, R.D., Rimer, R., Sperry, L. and Belnap, J. 2001. Exotic plant invasion alters nitrogen dynamics in an arid grassland. Ecological Applications **11**(5):1301-1310.

Facelli, J. M., and Pickett, S.T.A. 1991. Plant litter: its dynamics and effects on plant community structure. Botanical Review **57**:2-32.

Fargione, J., Brown, C.S., Tilman, D. 2003. Community assembly and invasion: An experimental test of neutral verses niche processes. Proceeding of the National Academy of Sciences of the United States of America **100**(15):8916-8920.

Ferrandis, P., Herranz, J., Martínez, J. and Martínez-Sánchez, J. 2001. Response to fire of a predominantly transient seed bank in a Mediterranean weedy pasture (eastern-central Spain). Ecoscience **8**(2):211-219.

Fick, G.W, and S.C. Mueller. 1989. Alfalfa quality, maturity, and mean stage of development. Cornell Coop. Extension Information Bulletin 217. Cornell University, Ithaca, NY.

Fitch, L., and Adams, B.W. 1998. Can cows and fish co-exist? Canadian Journal of Plant Science **78**(2):191-198.

Fitch, L., Adams, B.W., O'Shaughnessy, K. 2003. Riparian areas and grazing management. Cows and Fish Program.

Fitzsimmons, M. J., Pennock, D. J. and Thorpe, J. 2004. Effects of deforestation on ecosystem carbon densities in central Saskatchewan, Canada. Forest Ecology and management **188**(1):349-361.

Freedman, B. 2010. Environmental Science: A Canadian Prospective. Pearson Canada Inc., Toronto: pg 60-61.

Froud-Williams, R. J., Chancellor, R. J. and Drennan, D. S. H. 1983. Influence of cultivation regime upon buried weed seeds in arable cropping systems. Journal of Applied Ecology **20**(1):199-208.

Fuhlendorf, S. D., Engle, D. M., Elmore, R. D., Limb, R. F., and Bidwell, T. G. 2012. Conservation of pattern and process: Developing an alternative paradigm of rangeland management. Rangeland Ecology and Management **65**(6):579-589.

Fulkerson, W.J. and Michell, P.J. 1987. The effect of height and frequency of mowing on the yield and composition of perennial ryegrass-white clover swards in the autumn to spring period. Grass and Forage Science **42**(2):169-174.

Gabruck, D. T., Bork, E. W., Hall, L. M., King, J. R. and Hare, D. D. 2013. Interspecific relationships between white clover, Kentucky bluegrass, and Canada thistle during establishment. Agronomy Journal **105**(6):1467-1474.

Galinato, M.I. and Van der Valk, A.G. 1986. Seed germination traits of annuals and emergents recruited during drawdowns in the Delta Marsh, Manitoba, Canada. Aquatic Botany **26**:89-102.

Gardener, C. J., McIvor, J. G., and Jansen, A. 1993. Passage of legume and grass seeds through the digestive tract of cattle and their survival in faeces. Journal of Applied Ecology **30**(1):63-74.

Gauthier, D.A. and Wiken, E. B. 2003. Monitoring the conservation of grassland habitats, prairie ecozone, Canada. Environmental Monitoring and Assessment **88**(1):343-364.

Gelbard, J.L. and Belnap, J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology **17**(2):420-432.

Ghaderi-Far, F., Gherekhloo, J., and Alimagham, M. 2010. Influence of environmental factors on seed germination and seedling emergence of yellow sweet clover. Planta Daninha **28**(3):463-469.

Gifford, M. and Otfinowski, R. 2013. Landscape disturbances impact affect the distribution of exotic grasses in northern fescue prairies. Invasive Plant Science and Management **6**(4):577-584.

Gioria, M., Jarosik, V., and Pysek, P. 2014. Impact of invasions by alien plants on soil seed bank communities: emerging patterns. Evolution and Systematics **16**:132-142.

Glass, G.V., Peckham, P.D. and Sanders, J.R. 1972. Consequences of failure to meet assumptions underlying the fixed effects analyses of variance and covariance. Review of Educational Research **42**(3): 237-288.

Gleason, R.A., Euliss, N.H., Hubbard, D.E., and Duffy, W.G. 2003. Effects of sediment load on emergence of aquatic invertebrate and plants from wetland soil egg and seed banks. Wetlands **23**(1):26-34.

Gonzalez, S. and Ghermandi, L. 2008. Postfire seed bank dynamics in semiarid grasslands. Plant Ecology **199**:175-185.

Gordon, I.J. 1989. Vegetation community selection by ungulates on the Isle of Rhum. II. Vegetation community selection. Journal of Applied Ecology **26**(1):53-64.

Goslee, S., Sanderson, M., and Gonet, J. 2009. No persistent changes in pasture vegetation or seed bank composition after fallowing. Agronomy Journal **101**(5):1168-1174.

Gossner, M. M., Lewinsohn, T. M., Kahl, T., Grassein, F., Boch, S., Prati, …Allan, E. 2016. Land-use intensification causes multitrophic homogenization of grassland communities. Nature **540**(7632)266-269.

Gotelli, N.J, and Graves, G.R. 1996. Null Models in Ecology. Smithsonian Institution Press.

Government of Alberta. 2001. Native plant revegetation guidelines for Alberta.

Government of Alberta. 2010. Rangeland health assessment for grassland, forest & tame pasture. Alberta Government.

Government of Alberta. 2013. Range plant communities and range health assessment guidelines for the Central Parkland subregion of Alberta. Alberta Government, Red Deer, AB.

Grace, J.B. and Harrison, J.S. 1986. The biology of Canadian weeds.: 73. *Typha latifolia* L., *Typha angustifolia* L. and *Typha* x *glauca* Godr. Canadian Journal of Plant Science **66**(2):361-379.

Graham, P.H. 2005. Practices and issues in the inoculation of prairie legumes used in revegetation and restoration. Ecological Restoration **23**(3):187-195.

Gramineae Services Ltd. 2013. Recovery strategies for industrial development in native prairie: the Dry Mixedgrass natural subregion of Alberta. Prepared by Gramineae Services Ltd. for Alberta Environment and Sustainable Resource Development.

Grekul, C.W. and Bork, E.W. 2004. Herbage yield losses in perennial pasture due to Canada thistle (*Cirsium arvense*). Weed Technology **18**(3):784-794.

Grekul, C. W. and Bork, E. W. 2007. Fertilization augments Canada thistle (*Cirsium arvense* L. Scop.) control in temperate pastures with herbicides. Crop Protection **26**(4):668-676.

Grekul, C.W., Cole, D.E. and Bork, E.W. 2005. Canada thistle (*Cirsium arvense*) and pasture forage responses to wiping with various herbicides. Weed Technology **19**(2):298-306.

Grime, J.P. 1979. Plant strategies and vegetation processes. John Wiley & Sons, New York, USA.

.

Groya, F.L. and Sheaffer, C.C. 1981. Establishment of sod-seeded alfalfa at various levels of soil moisture and grass competition. Agronomy Journal **73**(3):560-565.

Gunn, C.R. 1965. The *Vicia americana* complex (Leguminosae). Retrospective Theses and Dissertations, Iowa State University. Paper 4086.

Gunn, C.R. 1970. Genus *Vicia* with notes about tribe Vicieae (Fabaceae) in Mexico and Central America. Technical Bulletin 1601, United States Department of Agriculture.

Guo, Y., Zhao, H., Zuo, X., Drake, S. and Zhao, X. 2008. Biological soil crust development and its topsoil properties in the process of dune stabilization, Inner Mongolia, China. Environmental Geology **54**(3):653-662.

Gustafson, D.J., Gibson, D.J. and Nickrent, D.L. 2002. Genetic diversity and competitive abilities of *Dalea purpurea* (Fabaceae) from remnant and restored grasslands. International Journal of Plant Sciences **163**(6):979-990.

Gustafson, D.J., Gibson, D.J. and Nickrent, D.L. 2005. Using local seeds in prairie restoration—data support the paradigm. Native Plants Journal **6**(1):25-28.

Hale, C.N., Lowther, W.L. and Lloyd, J.M. 1979. Effect of inoculant formulation on survival of *Rhizobium trifolii* and the establishment of oversown white clover (*Trifolium repens*). New Zealand Journal of Experimental Agriculture **7**(3):311-314.

Halfacree, K. 2007. Back-to-the-land in the twenty-first century: Making connections with rurality. Tijdschrift Voor Economische en Sociale Geografie **98**(1):3-8.

Halvorson, G.A. and Bauer, A. 1984. Yield and botanical composition of a grass-legume mixture on reclaimed land as affected by N and P fertilizer. Agronomy Journal **76**(3):355-358.

Hamdoun, A.M. 1972. Regenerative capacity of root fragments of *Cirsium arvense* (L.) Scop. Weed Research **12**(2):128-136.

Hamer, K.C. and Hill, J.K. 2000. Scale-dependent effects of habitat disturbance on species richness in tropical forests. Conservation Biology **14**(5):1435-1440.

Hammermeister, A.M., Naeth, M.A., Schoenau, J.J. and Biederbeck, V.O. 2003. Soil and plant response to wellsite rehabilitation on native prairie in southeastern Alberta, Canada. Canadian Journal of Soil Science **83**(5):507-519.

Hansen, M. J. and Clevenger, A.P. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation **125**(2):249-259.

Hao, X. and Chang, C. 2003. Does long-term heavy cattle manure application increase salinity of a clay loam soil in semi-arid southern Alberta? Agriculture, Ecosystems & Environment **94**(1):89-103.

Harker, K. N., Baron, V. S., Chanasyk, D.S., Naeth, M.A., and Stevenson, F.C. 2000. Grazing intensity effects on weed populations in annual and perennial pasture systems. Weed Science **48**(2):231-238.

Harper, J.L. 1977. Population biology of plants. Academic Press, London.

Hartnett, D.C., Samenus, R.J., Fischer, L.E. and Hetrick, B.A.D. 1994. Plant demographic responses to mycorrhizal symbiosis in tallgrass prairie. Oecologia **99**(1-2):21-26.

Hastings, A., Hom, C.L., Ellner, S., Turchin, P., and Godfray, H.C.J. 1993. Chaos in ecology: is mother nature a strange attractor? Annual Review of Ecology and Systematics **24**:1-33.

Havstad, K.M., Peters, D.P., Skaggs, R., Brown, J., Bestelmeyer, B., Fredrickson, E., Herrick, J. and Wright, J. 2007. Ecological services to and from rangelands of the United States. Ecological Economics **64**(2):261-268.

Hawkes, C.V. 2004. Effects of biological soil crusts on seed germination of four endangered herbs in a xeric Florida shrubland during drought. Plant Ecology **170**(1):121-134.

Helsen, K., Hermy M., and Honnay, O. 2015. Changes in species and functional trait composition of the seed bank during semi-natural grassland assembly: seed bank disassembly or ecological palimpsest? Journal of Vegetation Science **26**:58-67.

Helm, A., Kalamees, R., and Zobel, M. 2014. Vegetation patterns and their underlying processes: where are we now? Journal of Vegetation Science **25**(5):1113-1116.

Henderson, D.C., Ellert, B.H., and Naeth, M.A. 2004. Grazing and soil carbon along a gradient of Alberta rangelands. Journal of Range Management **57**(4):402-410.

Henderson, D.C. and Naeth, M.A. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. Biological Invasions **7**:639-650.

Hewins, D.B., Lyseng, M.P., Schoderbek, D.F., Alexander, M., Willms, W.D., Carlyle, C.N., Chang, S.X. and Bork, E.W. 2018. Grazing and climate effects on soil organic carbon concentration and particle-size association in northern grasslands. Scientific Reports **8**(1):1336.

Hickman, L.K. 2010. Reclamation Outcomes on Energy Disturbances in Silver Sagebrush Communities. M. Sc. Thesis, University of Calgary, Department of Environmental Science. Calgary, Alberta.

Hickman, L.K., Desserud, P.A., Adams, B.W. and Gates, C.C. 2013. Effects of disturbance on silver sagebrush communities in Dry Mixed-Grass prairie. Ecological Restoration **31**(3):274-282.

Hilger, H. and Lamb, E.G. 2017. Quantifying optimal rates of litter retention to maximize annual net primary productivity on mixed-grass prairie. Rangeland Ecology and Management **70**(2):219-224.

Hoffman, M. L., Owen, M. D. and Buhler, D. D. 1998. Effects of crop and weed management on density and vertical distribution of weed seeds in soil. Agronomy Journal **90**(6):793-799.

Hogenbirk, J. C. and Wein, R. W. 1992. Temperature effects on seedling emergence from boreal wetland soils: implications for climate change. Aquatic Botany **42**(4):361-373.

Honnay, O., Jacquemyn, H., Bossuyt, B., and Hermy, M. 2005. Forest fragmentation effects on patch occupancy and population viability of herbaceous plant species. New Phytologist **166**:723-736.

Hopfensperger, K.N. 2007. A Review of Similarity between seed sank and standing vegetation across ecosystems. Oikos **116**(9):1438-1448.

Howe, H.F. and Brown, J.S. 2000. Early effects of rodent granivory on experimental forb communities. Ecological Applications **10**(3):917-924.

Howe, H.F. and Brown, J.S. 2001. The ghost of granivory past. Ecology Letters **4**(4):371-378.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological monographs **54**(2):187-211.

Hutchings, M.J. and Booth, K.D. 1996. Studies on the feasibility of re-creating chalk grassland vegetation on ex-arable land. I. The potential roles of the seed bank and the seed rain. Journal of Applied Ecology **33**(5):1171-1181.

Iverson, L.R. and Wali, M.K. 1982. Buried, viable seeds and their relation to revegetation after surface mining. Journal of Range Management **35**(5):648-652.

Jacquemyn, H., Mechelen, C.V., Brys, R., and Honnay, O. 2011. Management Effects on the Vegetation and Soil Seed Bank of Calcareous Grasslands: An 11-Year Experiment. Biological Conversation **144**(1):416-422.

Jefferson, P.G., Iwaasa, A.D., Schellenberg, M.P., and McLeod, J.G. 2005. Re-evaluation of native plant species for seeding and grazing by livestock on the semiarid prairie of western Canada. Prairie Forum.

Jensen, K. and Gutekunst, K. 2003. Effects of litter on establishment of grassland plant species: the role of seed size and successional status. Basic and Applied Ecology **4**(6):579-587.

Johansen, J.R. 1993. Cryptogamic crusts of semiarid and arid lands of North America. Journal of Phycology **29**:140-147.

Johnston, D. 2011. Movement of weed seeds in reclaimed areas. Restoration Ecology **19**(4):446-449.

Johnston, A., Smoliak, S. and P.W. Stringer. 1969. Viable seed populations in Alberta prairie topsoils. Canadian Journal of Plant Science **49**(1):75-82.

Jong, E.D., and Button, R.G. 1973. Effects of pipeline installation on soil properties and productivity. Canadian Journal of Soil Science **53**(1):37-47.

Josephson, R.M. 1993. Economics of agricultural encroachment on wildlife habitat in southwest Manitoba. Canadian Journal of Agricultural Economics **41**(4):429-435.

Jungnitsch, P. F., Schoenau, J. J., Lardner, H. A. and Jefferson, P. G. 2011. Winter feeding beef cattle on the western Canadian prairies: Impacts on soil nitrogen and phosphorus cycling and forage growth. Agriculture, Ecosystems & Environment **141**(1):143-152.

Kachergis, E., Derner, J., Roche, L., Tate, K., Lubell, M., Mealor, R., and Magagna, J. 2013. Characterizing Wyoming ranching operations: Natural resource goals, management practices and information sources. Natural Resources **4**:45-54.

Kalamees, R. and Zobel, M. 2002. The role of the seed bank in gap regeneration in a calcareous grassland community. Ecology **83**(4):1017-1025.

Keddy, P.A. 1990. Competitive hierarchies and centrifugal organization in plant communities. Perspectives on Plant Competition. Edited by Grace, J.B. and D. Tilman. Academic Press Inc., Sandiego, California, USA.

Keddy, P.A and MacLellan, P. 1990. Centrifugal organization in forests. Oikos **59**(1):75-84.

Katepa-Mupondwa, F., Singh, A., Smith Jr, S.R. and McCaughey, W.P. 2002. Grazing tolerance of alfalfa (*Medicago* spp.) under continuous and rotational stocking systems in pure stands and in mixture with meadow bromegrass (*Bromus riparius* Rehm. syn. *B. biebersteinii* Roem & Schult). Canadian Journal of Plant Science **82**(2):337-347.

Kinucan, R.J. and Smeins, F.E. 1992. Soil seed bank of a semiarid Texas grassland under three long-term (36-years) grazing regimes. The American Midland Naturalist **128**(1):11-21.

Kjorlien, M.E. 1977. A review of historical information of fire history and vegetation description of Elk Island and the Beaver Hills. Parks Canada, Elk Island National Park, Fort Saskatchewan, Alberta.

Kleine, S. and Müller, C. 2011. Intraspecific plant chemical diversity and its relation to herbivory. Oecologia **166**(1):175-186.

Klemow, K.M., and Raynal, D.J. 1981. Population ecology of *Melilotus alba* in a limestone quarry. The Journal of Ecology **69**(1):33-44.

Klimes, J. 2007. Bud banks and their role in vegetative regeneration-a literature review and proposal for simple classification and assessment. Perspective in Plant Ecology, Evolution and Systematics **8**(3):115-129.

Klock, M.M., Barrett, L.G., Thrall, P.H. and Harms, K.E. 2015. Host promiscuity in symbiont associations can influence exotic legume establishment and colonization of novel ranges. Diversity and Distributions **21**(10):1193-1203.

Knott, D.M., Wenner, E.L. and Wendt, P.H. 1997. Effects of pipeline construction on the vegetation and macrofauna of two South Carolina, USA salt marshes. Wetlands **17**(1):65-81.

Kunelius, H.T. and Campbell, A.J. 1984. Performance of sod-seeded temperate legumes in grass dominant swards. Canadian Journal of Plant Science **64**(3):643-650.

Kupsch, T., France, K., Loonen, H., Burkinshaw, A., Willoughby, M., and McNeil, R. L. 2013. Guide to range plant community types and carrying capacity for the Central Parkland subregion of Alberta. Alberta Sustainable Resource Development, Government of Alberta.

Kunelius, H.T. and Campbell, A.J. 1984. Performance of sod-seeded temperate legumes in grass dominant swards. Canadian Journal of Plant Science **64**(3):643-650.

Kurstjens, D. A. G. and Kropff, M. J. 2001. The impact of uprooting and soil-covering on the effectiveness of weed harrowing. Weed Research **41**(3):211-228.

Laird, A.S. 2014. Residual effect of herbicides used in pastures on clover establishment and productivity. M. Sc. Thesis, Louisiana State University and Agricultural and Mechanical College, Department of Plant, Environmental and Soil Science, Baton Rouge, Louisiana, USA.

Lane, D.R., Coffin, D.P. and Lauenroth, W.K. 1998. Effects of soil texture and precipitation on above-ground net primary productivity and vegetation structure across the Central Grassland region of the United States. Journal of Vegetation Science **9**(2):239-250.

Langhans, T.M., Storm, C. and Schwabe, A. 2009. Biological soil crusts and their microenvironment: impact on emergence, survival and establishment of seedlings. Flora-Morphology, Distribution, Functional Ecology of Plants **204**(2):157-168.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2000. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on dry matter yield and forage quality. Canadian Journal of Plant Science **80**: 781-791.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2001. The effect of rejuvenation of Aspen Parkland ecoregion grass-legume pastures on botanical composition. Canadian Journal of Plant Science **81**(4):673-683.

Lardner, H.A., Wright, S.B.M., Cohen, R.D.H., Curry, P., and MacFarlane, L. 2002. Rejuvenation affects nutritive value of long-established tame forages. Canadian Journal of Animal Science **82**(4):621-626.

Larney, F. J. and Blackshaw, R. E. 2003. Weed seed viability in composted beef cattle feedlot manure. Journal of Environmental Quality **32**(3):1105-1113.

Larson, J.L. and Siemann, E. 1998. Legumes may be symbiont-limited during old-field succession. The American midland naturalist **140**(1):90-95.

Lauenroth, W.K. and Adler, P.B. 2008. Demography of perennial grassland plants: survival, life expectancy and life span. Journal of Ecology **96**(5):1023-1032.

Laughlin, D.C. 2003. Lack of native propagules in a Pennsylvania, USA, limestone prairie seed bank: futile hopes for a role in ecological restoration. Natural Areas Journal **23**(2):158-164.

Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangelands: a viewpoint. Journal of Range Management **44**(5):427-433.

Lavkulich, L.M., and Wiens, J.H. 1970. Division S-3—Soil microbiology and biochemistry: comparison of organic matter destruction by hydrogen peroxide and sodium hypochlorite and its effects on selected mineral constituents. Soil Science Society of America Proceedings **34**:755-758.

Ledgard, S.F. and Steele, K.W. 1992. Biological nitrogen fixation in mixed legume/grass pastures. Plant and Soil **141**:137-153.

Legendre, P. and Legendre, L. 1998. Numerical Ecology. 2nd ed. Elsevier, Amsterdam, Netherlands.

Levassor, C., M. Ortega, and Peco, B. 1990. Seed bank dynamics of Mediterranean pastures subjected to mechanical disturbance. Journal of Vegetation Science **1**(3):339-344.

Li, X. Jia, X., Long, L., and Zerbe, S. 2005. Effects of biological soil crusts on seed bank, germination and establishment of two annual plant species in the Tengger Desert (N China). Plant and Soil **277**(1):375-385.

Li, X.Y., Liu, L.Y., and Wang, J.H. 2004. Wind tunnel simulation of aeolian sandy soil erodibility under human disturbance. Geomorphology **59**(1):3-11.

Lockwood, J.L., Cassey, P. and Blackburn, T. 2005. The role of propagule pressure in explaining species invasions. Trends in Ecology & Evolution **20**(5):223-228.

López-Mariño, A., Luis-Calabuig, E., Fillat, F., and Bermudez, F.F. 2000. Floristic composition of established vegetation and the soil seed bank in pasture communities under different traditional management regimes. Agriculture, Ecosystems & Environment **78**(3):273-282.

Luo, W. and Zhao, W. 2015. Burial depth and diameter of the rhizome fragments affect the regenerative capacity of a clonal shrub. Ecological Complexity **23**:34-40.

Low, C.H. 2016. Impacts of a six year old pipeline right of way on *Halimolobos virgata* (Nutt.) OE Schulz (slender mouse ear cress), native Dry Mixedgrass prairie uplands, and wetlands. Ph. D. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Loydi, A., Eckstein, R.L., Otte, A. and Donath, T.W. 2013. Effects of litter on seedling establishment in natural and semi-natural grasslands: a meta-analysis. Journal of Ecology **101**(2):454-464.

Lyseng, M.P., Bork, E.W., Hewins, D.B., Alexander, M.J., Carlyle, C.N., Chang, S.X., and Willms, W.D. 2018. Long-term grazing impacts on vegetation diversity, composition and exotic species presence across an aridity gradient on northern temperate grasslands. Plant Ecology **219**(6):649-663.

Ma, M., Zhou, X., and Du, G. 2010. Role of soil seed bank along a disturbance gradient in an alpine meadow on the Tibet Plateau. Flora-Morphology, Distribution, Functional Ecology of Plants **205**(2):128-134.

MacDougall, A.S. and Turkington, R. 2005. Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology **86**(1):42-55.

Macdonald, I.D. 2005. Status of the slender mouse-ear-cress (*Halimolobos virgata*) in Alberta. Fish and Wildlife Division, Alberta Sustainable Resource Development. Alberta Wildlife Status Report No. 55.

Malhi, S.S., Heier, K., Nielsen, K., Davies, W.E., and Gill, K.S. 2000. Efficacy of pasture rejuvenation through mechanical aeration and N fertilization **80**:813-815.

Malo, J.E., and Suárez, F. 1995. Establishment of pasture species on cattle dung: the role of endozoochorous seeds. Journal of Vegetation Science **6**(2):169-174.

Mapfumo, E., Chanasyk, D. S., Baron, V. S., and Naeth, M. A. 2000. Impacts on selected soil parameters under short-term forage sequences. Journal of Range Management **53**(5):466-470.

Mapfumo, E., Chanasyk, D. S., Naeth, M. A. and Baron, V. S. 1999. Soil compaction under grazing of annual and perennial forages. Canadian Journal of Soil Science **79**(1):191-199.

Mapfumo, E., Naeth, M. A., Baron, V. S., Dick, A. C. and Chanasyk, D. S. 2002. Grazing impacts on litter and roots: perennial versus annual grasses. Journal of Range Management **55**(1):16-22.

Marlette, G.M., and Anderson, J.E. 1986. Seed banks and propagule dispersal in crested-wheatgrass stands. Journal of Applied Ecology **23**(1):161-175.

Martin, R.E., Miller, R.L, and Cushwa, C.T. 1975. Germination response of legume seeds subjected to moistand dry heat. Ecology **56**:1441-1445.

Martínez-Garza, C., Osorio-Beristain, M., Valenzuela-Galván, D. and Nicolás-Medina, A. 2011. Intra and inter-annual variation in seed rain in a secondary dry tropical forest excluded from chronic disturbance. Forest Ecology and Management **262**(12):2207-2218.

Masters, J.A. 2014. Invasive plants as drivers and passengers of community change in a disturbed urban forest. Ph.D. Thesis, University of Louisville.

Matthews, J.W. and Spyreas, G. 2010. Convergence and divergence in plant community trajectories as a framework for monitoring wetland restoration progress. Journal of Applied Ecology **47**(5):1128-1136.

Mattingly, W.B, Hewlate, R. and Reynolds, H.L. 2007. Species evenness and invasion resistance of experimental grassland communities. Oikos **116**(7):1164-1170.

Mayor, J. P and Dessaint, F. 1998. Influence of weed management strategies on soil seedbank diversity. Weed Research **38**:95-105.

Maxwell, B.D., Wilson, M.V., and Radosevich, S.R. 1988. Population modeling approach for evaluating leafy spurge (*Euphorbia esula*) development and control. Weed Technology **2**(2):132-138.

McClay, A. 2012. Revising Alberta's Provincial Weeds List: Experiences and Lessons Learned. Weeds Across Borders Conference, At Cancún, Quintana Roo, Mexico.

McGeehan, S.L. and Naylor, D.V. 1988. Automated instrumental analysis of carbon and nitrogen in plant and soil samples. Communication in Soil Science and Plant Analysis **19**(4):493-505.

McGill, B.J., Enquist, B.J., Weiher, E., and Westoby, M. 2006. Rebuilding community ecology from functional traits. Trends in Ecology and Evolution **21**(4):178-185.

McGraw, R.L., Shockley, F.W., Thompson, J.F, and Roberts, C.A. 2004. Evaluation of native legume species for forage yield, quality, and seed production. Native Plants Journal Fall.

McLean, A. and Wikeem, S. 1985. Rough fescue response to season and intensity of defoliation. Journal of Range Management **38**(2):100-103.

McLeod, E.M., Banerjee, S., Bork, E.W., Hall, L.M., and Hare, D.D. 2015. Structural equation modeling reveals complex relationships in mixed forage swards. Crop Protection **78**:106-113.

McNeill, J. 1977. The biology of Canadian weeds: 25. *Silene alba* (Miller) E. H. L. Krause. Canadian Journal of Plant Science **57**(4):1103-1114.

Menalled, F. D., Kohler, K. A., Buhler, D. D. and Liebman, M. 2005. Effects of composted swine manure on weed seedbank. Agriculture, Ecosystems & Environment **111**(1):63-69.

Meyer, S.E., Quinney, D., Nelson, D.L. and Weaver, J. 2007. Impact of the pathogen *Pyrenophora semeniperda* on *Bromus tectorum* seedbank dynamics in North American cold deserts. Weed Research **47**(1):54-62.

Miklas, P.N., Townsend, C.E. and Ladd, S.L. 1987. Seed coat anatomy and the scarification of cicer milkvetch seed. Crop Science **27**(4):766-772.

Mikutta, R., Kleber, M., Kaiser, K., and Jahn, R. 2005. Review: organic matter removal from soil using hydrogen peroxide, sodium hypochlorite, and disodium peroxodisulfate. Soil Science Society of America Journal **69**:120-135.

Milchunas, D.G., Sala, O.E. and Lauenroth, W.K. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. The American Naturalist **132**(1):87-106.

Miller, A.J. 2013. Recovery of legumes in northern temperate pastures following the application of broadleaf herbicides. M. Sc. Thesis, University of Alberta, Agricultural, Food and Nutritional Science. Edmonton, Alberta.

Miller, A.J., Bork, E.W., Hall, L.M., and Summers, B. 2015. Long-term forage dynamics in pastures sprayed with residual broadleaf herbicide: A test of legume recovery. Canadian Journal of Plant Science **95**(1):43-53.

Miller, J., Chanasyk, D., Curtis, T., Entz, T., and Willms, W. 2010. Influence of streambank fencing with a cattle crossing on riparian health and water quality of the Lower Little Bow River in Southern Alberta, Canada. Agricultural Water Management **97**(2):247-258.

Mills, A. and Sina Adl, M. 2006. The effects of land use intensification on soil biodiversity in the pasture. Canadian Journal of Plant Science **86**(Special Issue):1339-1343.

Minta, S.C., and Marsh, R.E. 1988. Badgers (*Taxidea taxus*) as occasional pests in agriculture. Vertebrate Pest Conference Proceedings Collection VPC13:42.

Mischkolz, J.M., Schellenberg, M.P., and Lamb, E.G. 2013. Early productivity and crude protein content of establishing forage swards composed of combinations of native grass and legume species in mixed-grassland ecoregions. Canadian Journal of Plant Science **93**(3):445-454.

Moeslund, J.E., Brunbjerg, A.K., Clausen, K.K., Dalby, L., Fløjgaard, C., Juel, A. and Lenoir, J. 2017. Using dark diversity and plant characteristics to guide conservation and restoration. Journal of Applied Ecology.

Moisey, D., Young, J., Lawrence, D., Stone, C., and Willoughby, M. 2012. Guide to range plant community types and carrying capacity for the Dry and Central Mixedwood Subregions in Alberta. Alberta Sustainable Resource Development, Government of Alberta.

Molano-Flores, B. 2012. Diaspore morphometrics and self-burial in *Hesperostipa spartea* from loam and sandy soils. The Journal of the Torrey Botanical Society **139**(1):56-62.

Molles, M.C., and Cahill, J.F. 2008. Ecology: concepts and applications: Canadian edition. McGraw-Hill, Dubuque, IA.

Moore, R.J. 1975. The biology of Canadian Weeds. 13. *Cirsium arvense* (L.) Scop. Canadian Journal of Plant Science **55**:1033-1048.

Mortenson, M.C., Schuman, G.E., Ingram, L.J., Nayigihugu, V. and Hess, B.W. 2005. Forage production and quality of a mixed-grass rangeland interseeded with *Medicago sativa* ssp. *falcata*. Rangeland Ecology & Management **58**(5):505-513.

Moss, E.H. 2010. Flora of Alberta, 2nd ed. University of Toronto Press, Toronto.

Naeth, M.A., Bailey, A.W., and McGill, W.B. 1987. Persistence of changes in selected soil chemical and physical properties after pipeline installation in solonetzic native rangeland. Canadian Journal of Soil Science **67**(4):747-763.

Naeth, M.A. Bailey, A.W., Pluth, D.J., Chanasyk, D.S., and Hardin, R.T. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland. Journal of Range Management **44**(1):7-12.

Naeth, M.A., Chanasyk, D.S. and McGill, W.B. 1993. Soil temperature regime in mixed prairie rangeland after pipeline construction and operation. Canadian agricultural engineering **35**(2):88-95.

Nannt, M.R. 2014. Impacts of distance to pipeline disturbance on mixed grass prairie and *Halimolobos virgata* (Nutt.) OE Schulz (slender mouse ear cress). M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Nasen, L.C., Noble, B.F. and Johnstone, J.F. 2011. Environmental effects of oil and gas lease sites in a grassland ecosystem. Journal of Environmental Management **92**(1):195-204.

Nemergut, D.R., Anderson, S.P., Cleveland, C.C., Martin, A.P., Miller, A.E., Seimon, A. and Schmidt, S.K. 2007. Microbial community succession in an unvegetated, recently deglaciated soil. Microbial Ecology **53**(1):110-122.

Neto, M.S., Jones, R.M. and Ratcliff, D. 1987. Recovery of pasture seed ingested by ruminants. 1. Seed of six tropical pasture species fed to cattle, sheep and goats. Australian Journal of Experimental Agriculture **27**(2):239-246.

Neville, M., Alexander, M., Adams, B., DeMaere, C., Lawrence, D., and McGillvray, S. 2016. Principles for minimizing surface disturbances in native grassland: principles, guidelines, and tools for all industrial activity in native grassland in the prairie and parkland landscapes of Alberta. Alberta Environment and Parks, Edmonton, Alberta.

Noyd, R.K., Pfleger, F.L., Norland, M.R. and Sadowsky, M.J. 1995. Native prairie grasses and microbial community responses to reclamation of taconite iron ore tailing. Canadian Journal of Botany **73**(10):1645-1654.

Ojima, D. S., Schimel, D. S., Parton, W. J. and Owensby, C. E. 1994. Long-and short-term effects of fire on nitrogen cycling in tallgrass prairie. Biogeochemistry **24**(2):67-84.

Olson, E.R. and Doherty, J.M. 2012. The legacy of pipeline installation on the soil and vegetation of southeast Wisconsin wetlands. Ecological Engineering **39**:53-62.

O'Meilia, M.E., Knopf, F.L, and J.C. Lewis. 1982. Consequences of competition between prairie dogs and beef cattle. Journal of Range Management **35**(5):580-585.

Ostermann, D.K. 2001. Revegetation assessment of a twelve-year-old pipeline on native rangeland in southern Alberta. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Otfinowski, R., and Kenkel, N.C. 2008. Clonal integration facilitates the proliferation of smooth brome clones invading northern fescue prairies. Plant Ecology **199**(2):235-242.

Otfinowski, R., Kenkel, N.C., and Van Acker, R.C. 2008. Reconciling seed dispersal and seed bank observations to predict smooth brome (*Bromus inermis*) invasions of northern prairie. Invasive Plant Science and Management **1**:279-286.

Ott, J.P., Butler, J.L., Rong, Y., and Xu, L. 2016. Greater bud outgrowth of *Bromus inermis* than *Pascopyrum smithii* under multiple environmental conditions. Journal of Plant Ecology **10**(3):518-527.

Palit, R., Bai, Y., Romo, J., Coulman, B. and Warren, R. 2016. Seed production in *Festuca hallii* is regulated by adaptation to long-term temperature and precipitation patterns. Rangeland Ecology and Management **70**(2):238-243.

Pärtel, M., Szava-Kovats, R. and Zobel, M. 2011. Dark diversity: shedding light on absent species. Trends in ecology & evolution **26**(3):124-128.

Paul, E.A., Myers, R.J.K. and Rice, W.A. 1971. Nitrogen fixation in grassland and associated cultivated ecosystems. Plant and Soil **35**(1):495-507.

Payne, J., Livesey, C., and Murphy, A. 2015. Cattle Poisoning: Principles of Toxicological Investigations. Bovine Medicine, 3rd ed. John Wiley & Sons ltd. Sussex, UK: pg 211-224.

Peroni, P. A. and Armstrong, R. T. 2001. Density, dispersion and population genetics of a *Silene latifolia* seed bank from southwestern Virginia. Journal of the Torrey Botanical Society **128**(4):400-406.

Peterson, P.R., Scheafer, C.C., and Hall, M.H. 1992. Drought effects on perennial forage legume yield and quality. Agronomy Journal **84**(5):774-779.

Petherbridge, W.L. 2000. Sod salvage and minimal disturbance pipeline reclamation techniques: implications for native prairie restoration. M. Sc. Thesis, University of Alberta, Department of Renewable Resources. Edmonton, Alberta.

Pleasant, J.M.T., and Schlather, K.J. 1994. Incidence of weed seed in cow (*Bos* sp.) manure and its importance as a weed source for cropland. Weed Technology **8**(2):304-310.

Poiani, K.A. and Johnson, W.C. 1988. Evaluation of the emergence method in estimating seed bank composition of prairie wetlands. Aquatic Botany **32**:91-97.

Ponomarenko, E. V., and Anderson, D. W. 2001. Importance of charred organic matter in Black Chernozem soils of Saskatchewan. Canadian Journal of Soil Science **81**:285-297.

Popp, M., Chorney, B. and Keisling, T. 2004. A Case study on rotational grazing and riparian zone management: Implications for producers and a conservation agency. Journal of Natural Resources and Life Sciences Education **33**:28-34.

Portnoy, S. and Willson, M.F. 1993. Seed dispersal curves: behavior of the tail of the distribution. Evolutionary Ecology **7**(1):25-44.

Proulx, G. 2010. Factors contributing to the outbreak of Richardson's ground squirrel populations in the Canadian prairies. Proceedings of the 24th Vertebrate Pest Conference, Sacramento, California (pp. 213-217).

Province of Alberta. 2010. Weed Control Act. Her Majesty the Queen in the Right of Alberta, Edmonton.

Pyke, D.A. 1990. Comparative demography of co-occurring introduced and native tussock grasses: persistence and potential expansion. Oecologia **82**(4):537-543.

Pyle, L., Hall, L.M. and Bork, E.W. 2017. Linking management practices with range health in northern temperate pastures. Canadian Journal of Plant Science.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Rashford, B.S., Walker, J.A., and Bastian, C.T. 2011. Economics of grassland conversion to cropland in the prairie pothole region. Conservation Biology **25**(2):276-284.

Reaume, T. 2009. 620 Wild Plants of North America. University of Toronto Press, Toronto: pg 294 and 306.

Reed, A.W., Kaufman, G.A. and Kaufman, D.W. 2006. Effect of plant litter on seed predation in three prairie types. The American Midland Naturalist **155**(2):278-285.

Rees, M., 1996. Evolutionary ecology of seed dormancy and seed size. Philosophical Transactions: Biological Sciences **351**(1345):1299-1308.

Reeve, I.J., Coleman, M.J., and Sindel, B.M. 2015. Factors influencing rural landholder support for a mandated weed control policy. Land Use Policy **46**:314-323.

Ren, L. and Bai, Y. 2016a. Smoke and ash effects on seedling emergence from germinable soil seed bank in Fescue Prairie. Rangeland Ecology and Management **69**(6):499-507.

Ren, L. and Bai, Y. 2016b. Smoke originating from different plants has various effects on germination and seedling growth of species in Fescue Prairie. Botany **94**(12):1141-1150.

Ren, L. and Bai, Y. 2017. Burning modifies composition of emergent seedlings in fescue prairie. Rangeland Ecology & Management **70**(2):230-237.

Renne, I.J. and Tracy, B.F. 2007. Disturbance persistence in managed grasslands: shifts in aboveground community structure and the weed seed bank. Plant Ecology **190**(1):71-80.

Robinson, D.H. 1947. Leguminous forage plants, 2nd ed.

Rogers, M.E., Colmer, T.D., Frost, K., Henry, D., Cornwall, D., Hulm, E., Deretic, J., Hughes, S.R., and Craig, D. 2008. Diversity in the genus *Melilotus* for tolerance to salinity and waterlogging. Plant and Soil **304**(1-2):89-101.

Rogler, G.A. and Lorenz, R.J. 1983. Crested Wheatgrass: Early History in the United States. Journal of Range Management **36**(1):91-93.

Romo, J.T., Grilz, P.L., Bubar, C. J. and Young, J. A. 1991. Influences of temperature and water stress on germination of plains rough fescue. Journal of Range Management **44**(1):75-81.

Romo, J.T. and Gross, D.V. 2011. Preburn history and seasonal burning effects on the soil seed bank in the Fescue Prairie. The American Midland Naturalist **165**(1):74-90.

Rook, A.J., Dumont, B., Isselstein, J., Osoro, K., WallisDeVries, M.F., Parente, G., and Mills, J. 2004. Matching type of livestock to desired biodiversity outcomes in pastures – a review. Biological Conservation **119**(2):137-150.

Rowan, R.C. 1994. Are small-acreage livestock producers real ranchers? Rangelands **16**(4):161-166.

Rowe, J.S. 1987. Status of the aspen parkland in the Prairie Provinces. Endangered Species in the Prairie Provinces, Prairie Conservation and Endangered Species Conference, pp 27-33.

Rowland, J. 2008. Ecosystem impacts of historical shallow gas wells within the CFB Suffield National Wildlife Area. Report submitted February 2, 2008.

Rozema J., Tosserams M., Nelissen H. J. M, Vanheerwaarden L., Broekman R. A., and Flierman, N. 1997. Stratospheric ozone reduction and ecosystem processes: enhanced UV-B radiation affects chemical quality and decomposition of leaves of the dune grassland species *Calamagrostis epigeios*. Plant Ecology **128**(17):284–94.

Ruprecht, E. 2006. Successfully recovered grassland: a promising example from Romanian old-fields. Restoration Ecology **14**(3):473-480.

Russi, L., Cocks, P.S. and Roberts, E.H. 1992. Hard-seededness and seed bank dynamics of six pasture legumes. Seed Science Research **2**(4):231-241.

Samson, F. and Knopf, F. 1994. Prairie Conservation in North America. BioScience **44**(6):418-421.

Samson, F.B., Knopf, F.L. and Ostlie, W.R. 2004. Great Plains ecosystems: past, present, and future. Wildlife Society Bulletin **32**(1):6-15.

Sanderson, M.A., S. C. Goslee, K. D. Klement, and Soder, K.J. 2007. Soil seed bank composition in pastures of diverse mixtures of temperate forages. Agronomy Journal **99**:1514-1520.

Sanderson, M.A., Skinner, R.H., Barker, D.J., Edwards, G.R., Tracy, B.F., and Wedin, D.A. 2004. Plant species diversity and management of temperate forage and grazing land ecosystems. Crop Science **44**:1132-1144.

Sanderson, M.A., Stout, R., Goslee, S., Gonet, J., Smith, R.G. 2014. Soil seed bank community structure of pastures and hayfields on an organic farm. Canadian Journal of Plant Science 10.4141/CJPS2013-288.

Savory, A. and Parsons, D.S. 1980. The Savory grazing method. Rangelands **2**: 234–237.

Sayre, N.E. 2004. Viewpoint: The need for qualitative research to understand ranch management. Journal of Range Management **57**:668-674.

Schellberg, J., Möseler, B.M., Kühbauch, W., and Rademacher, I.F. 2001. Long-term effects of fertilizer on soil nutrient concentration, yield, forage quality and floristic composition of a hay meadow in the Eifel Mountains, Germany. Grass and Forage Science **54**(3):195-207.

Scheffler, EJ. 1976. Aspen forest vegetation in a portion of the east-central Alberta parklands. M.Sc. thesis, University of Alberta, Edmonton, Alberta.

Sharafatmandrad, M., Mesdaghi, M., Bahremand, A., and Barani, H. 2010. The role of litter in rainfall interception and maintenance of superficial soil water content in arid rangeland in Khabr National Park in south-eastern Iran. Arid Land Research and Management **24**:213-222.

Schröder, R. and Prasse, R. 2013. Do cultivated varieties of native plants have the ability to outperform their wild relatives? PLoS One **8**(8):e71066.

Schwinning, S. and Parsons, A. J. 1996. Analysis of the coexistence mechanisms for grasses and legumes in grazing systems. Journal of Ecology **84**(6):799-813.

Schwinning, S. and Weiner, J. 1998. Mechanisms determining the degree of size asymmetry in competition among plants. Oecologia **113**(4):447-455.

Semiadil, G., Barry, T.N., Muir, P.D. and Hodgson, J. 1995. Dietary preferences of sambar (*Cervus unicolor*) and red deer (*Cervus elaphus*) offered browse, forage legume and grass species. The Journal of Agricultural Science **125**(1):99-107.

Shi, P., Xiao, J., Wang, Y.F. and Chen, L.D. 2014. The effects of pipeline construction disturbance on soil properties and restoration cycle. Environmental Monitoring and Assessment **186**(3):1825-1835.

Šimek, M., Brůček, P., Hynšt, J., Uhlířová, E. and Petersen, S. O. 2006. Effects of excretal returns and soil compaction on nitrous oxide emissions from a cattle overwintering area. Agriculture, Ecosystems & Environment **112**(2):186-191.

Simmers, S.M., and S.M. Galatowitsch. 2010. Factors affecting revegetation of oil field access roads in semiarid grassland. Restoration Ecology **18**(s1):27-39.

Sinkins, P.A., and Otfinowski, R. 2012. Invasion or retreat? The fate of exotic invaders on the northern prairies, 40 years after cattle grazing. Plant Ecology **213**(8):1251-1262.

Sleugh, B., Moore, K.J., George, J.R. and Brummer, E.C. 2000. Binary legume–grass mixtures improve forage yield, quality, and seasonal distribution. Agronomy Journal **92**(1):24-29.

Smith, S.R., J. H. Bouton and Hoveland, C.S. 1988. Alfalfa persistence and regrowth potential under continuous grazing. Agronomy Journal **81**:960-965.

Smolenski, S.J., Kinghorn, A.D. and Balandrin, M.F. 1981. Toxic constituents of legume forage plants. Economic Botany **35**(3):321-355.

Smoliak, S. 1974. Range vegetation and sheep production at three stocking rates on *Stipa-Bouteloua* prairie. Journal of Range Management **27**(1):23-26.

Soil Classification Working Group. 1998. The Canadian system of soil classification, Third Edition. Research Branch, Agriculture and Agri-Food Canada. NRC Research Press, Ottawa.

Soon, Y.K., Arshad, M.A., Rice, W.A. and Mills, P. 2000. Recovery of chemical and physical properties of boreal plain soils impacted by pipeline burial. Canadian Journal of Soil Science **80**(3):489-497.

Spehn, E.M., Scherer-Lorenzen, M., Schmid, B., Hector, A., Caldeira, M.C., Dimitrakopoulos, P.G., Finn, J.A., Jumpponen, A., O'donnovan, G., Pereira, J.S. and Schulze, E.D. 2002. The role of legumes as a component of biodiversity in a cross-European study of grassland biomass nitrogen. Oikos **98**(2):205-218.

Spribille, T., Tuovinen, V., Resl, P., Vanderpool, D., Wolinski, H., Aime, M.C., Schneider, K., Stabentheiner, E., Toome-Heller, M., Thor, G. and Mayrhofer, H. 2016. Basidiomycete yeasts in the cortex of ascomycete macrolichens. Science **353**(6298):488-492.

Sprinkle, J.W. 2010. Bud bank density regulates invasion by exotic plants. M.Sc. Thesis, Oklahoma State University.

Staden, J.V., Brown, N.A., Jäger, A. K., and Johnson, T.A. 2000. Smoke as a germination cue. Plant Species Biology **15**(2):167-178.

Statistics Canada. 2011. Farm and farm operator data: 2011 census of agriculture. Statistics Canada Catalogue no. 95-640-X.

Stephenson, G. R. and Veigel, A. 1987. Recovery of compacted soil on pastures used for winter cattle feeding. Journal of Range Management **40**(1):46-48.

Stewart, W.L. 2006. The effects of remnant seed source size on plant performance in a prairie restoration. M. Sc. Thesis, Eastern Illinois University, Department of Biological Sciences, Charleston, Illinois.

Stoa, T.E. 1933. Persistence of viability of sweet clover seed in a cultivated soil. Journal of the American Society of Agronomy **25**:177-81.

Suding, K.N. 2011. Toward an era of restoration in ecology: successes, failures, and opportunities ahead. Annual Review of Ecology, Evolution, and Systematics **42**:465-487.

Symstad, A.J. and Jonas, J.L. 2011. Incorporating biodiversity into rangeland health: Plant species - richness and diversity in the Great Plains grasslands. Rangeland Ecology & Management **64**(6):555-572.

Tallowin, J.R.B., Rook, A.J., and Rutter, S.M. 2005. Impact of grazing management on biodiversity of grasslands. Animal Science **81**(2):193-198.

Tannas, K. 2004. Common plants of the western rangelands: Volume 3: Forbs. Alberta Agriculture, Food and Rural Development, Her Majesty the Queen in the Right of Alberta, Edmonton.

Tannas, S. 2011. Mechanisms regulating *Poa pratensis* L. and *Festuca campestris* Rybd. Within the foothills fescue grasslands of southern Alberta. Ph.D. Thesis, University of Alberta, Department of Agriculture, Food and Nutritional Science. Edmonton, Alberta.

Tannas, S., Hewins, D.B., and Bork, E.W. 2015. Isolating the role of soil resources, defoliation, and interspecific competition on early establishment of late successional bunchgrass *Festuca campestris*. Restoration Ecology **23**(4):366-374.

Teague, R. Provenza, F., Kreuter, U., Steffens, T., and Barnes, M. 2013. Multi-paddock grazing on rangelands: Why the perceptual dichotomy between research results and rancher experience? Journals of Environmental Management **128**:699-717.

Ter Heerdt, G.N.J, Verweij, G.L., Bekker, R.M., and Bakker, J.P. 1996. An improved method for seed-bank analysis: seedling emergence after removing soil by sieving. Functional Ecology **10**:144-151.

Thomas, A. G. 1985. Weed survey system used in Saskatchewan for cereal and oilseed crops. Weed Science **33**(1):34-43.

Thompson, K. and Grime, J.P. 1979. Seasonal variation in the seed banks of herbaceous species in ten contrasting habitats. Journal of Ecology **67**(3):893-921.

Thompson, K., Band, S.R., and Hodgson, J.G. 1993. Seed size and shape predict persistence in soil. Functional Ecology **7**(2):236-241.

Thompson, K., Bakker, J.P., and Bekker, R.M. 1997. The soil seed banks of north west Europe: Methodology, density and longevity. Cambridge University Press, Cambridge, UK.

Tilman, D. 1985. The resource-ratio hypothesis of plant succession. The American Naturalist **125**(6):827-852.

Tilman, D. 2004. Niche trade-offs, neutrality, and community structure: A stochastic theory of resource competition, invasion, and community assembly. Proceeding of the National Academy of Sciences of the United States of America **101**(30):10854-10861.

Tlusty, B., Grossman, J.M. and Graham, P.H. 2004. Selection of rhizobia for prairie legumes used in restoration and reconstruction programs in Minnesota. Canadian Journal of Microbiology **50**(11):977-983.

Townsend, C.E. 1972. Influence of seed size and depth of planting on seedling emergence of two milkvetch species. Agronomy Journal **64**(5):627-630.

Toynbee, K. 1987. Prolific flowering year for plains rough fescue at the Kernen Prairie. Blue Jay **45**:142-143.

Tracy, B.F., and Sanderson, M.A. 2000. Seedbank diversity in grazing lands of the Northeast United States. Journal of Range Management **53**(1):114-118.

Travnicek, A.J., Lym, R.G. and Prosser, C. 2005. Fall-prescribed burn and spring-applied herbicide effects on Canada thistle control and soil seedbank in a northern mixed-grass prairie. Rangeland Ecology and Management **58**(4):413-422.

Trumble, J.T. and Kok, L.T. 1982. Integrated pest management techniques in thistle suppression in pastures of North America. Weed Research **22**:345-359

Turnbull, L.A., Rahm, S., Baudois, O., Eichenberger-Glinz, S., Wacker, L. and Schmid, B. 2005. Experimental invasion by legumes reveals non-random assembly rules in grassland communities. Journal of Ecology **93**(6):1062-1070.

Turkington, R. and Burdon, J.J. 1983. The biology of Canadian weeds: 57. *Trifolium repens* L. Canadian Journal of Plant Science **63**(1):243-266.

Turkington, R. A., Cavers, P. B., and Rempel, E. 1978. The biology of Canadian weeds: 29. *Melilotus alba* Desr. and *M. officinalis* (L.) Lam. Canadian Journal of Plant Science **58**(2):523-537.

Van Assche, J.A., Debucquoy, K.L. and Rommens, W.A. 2003. Seasonal cycles in the germination capacity of buried seeds of some Leguminosae (Fabaceae). New Phytologist **158**(2):315-323.

Van Riper, L.C. and Larson, D.L. 2009. Role of invasive *Melilotus officinalis* in two native plant communities. Plant Ecology **200**:129-139.

Van Riper, L.C., Larson, D.L. and Larson, J.L. 2010. Nitrogen-limitation and invasive sweetclover impacts vary between two Great Plains plant communities. Biological Invasions **12**(8):2735-2749.

Vaness, B.M. and Wilson, S.D. 2007. Impact and management of crested wheatgrass (*Agropyron cristatum*) in the northern Great Plains. Canadian Journal of Plant Science **87**(5):1023-1028.

Viall, E.M., Gentry, L.F., Hopkins, D.G., Ganguli, A.C. and Stahl, P. 2014. Legacy effects of oil road reclamation on soil biology and plant community composition. Restoration Ecology **22**(5):625-632.

Vujnovic, K., Wein, R., and Dale, M.R.T. 2000. Factors determining the centrifugal organization of remnant Festuca grassland communities in Alberta. Journal of Vegetation Science **11**:127-134.

Wagg, C., Boller, B., Schneider, S., Widmer, F. and van der Heijden, M.G. 2015. Intraspecific and intergenerational differences in plant–soil feedbacks. Oikos **124(**8):.994-1004.

Wagner, M., Heinrich, W., and Jetschke, G. 2006. Seed bank assembly in an unmanaged ruderal grassland recovering from long-term exposure to industrial emissions. Acta Oecologica **30**:342-352.

Wainwright, C.E., Staples, T.L., Charles, L.S., Flanagan, T.C., Lai, H.R., Loy, X., Reynolds, V.A. and Mayfield, M.M. 2017. Links between community ecology theory and ecological restoration are on the rise. Journal of Applied Ecology 00:1-12.

Wali, M.K. 1999. Ecological succession and the rehabilitation of disturbed terrestrial ecosystems. Plant and Soil **213**(1-2):195-220.

Walton, P.D., Martinez, R. and Bailey, A.W. 1981. A comparison of continuous and rotational grazing. Journal of Range Management **34**(1):19-21.

Wang, W., Liu, Y., Li, D., Hu, C. and Rao, B. 2009. Feasibility of cyanobacterial inoculation for biological soil crusts formation in desert area. Soil Biology and Biochemistry **41**(5):926-929.

Warren, R.J., Bahn, V. and Bradford, M.A. 2012. The interaction between propagule pressure, habitat suitability and density-dependent reproduction in species invasion. Oikos **121**(6):874-881.

Wellstein, C., Otte, A., and Waldhardt, R. 2007. Seed bank diversity in mesic grasslands in relation to vegetation type, management and site conditions. Journal of Vegetation Science **18**:153-162.

Werner, P.A. and Rioux, R. 1977. The biology of Canadian weeds. 24. *Agropyron repens* (L.) Beauv. Canadian Journal of Plant Science **57**(3):905-919.

Westoby, M., Walker, B., and Noy-Meir, I. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management **42**(4):266-274.

Whicker, A. D. and Detling, J. K. 1988. Ecological consequences of prairie dog disturbances. BioScience **8**(11):778-785.

White, S.R., Bork, E.W., Karst, J., and Cahill, J. 2012. Similarity between grassland vegetation and seed bank shifts with altered precipitation and clipping, but not warming. Community Ecology **13**(2):129-136.

Wienhold, C.E. and Van der Valk, A.G. 1989. The impact of duration on the seed banks of northern prairie wetlands. Canadian Journal of Botany 67(6):1878-1884.

Wiese, A. F., Sweeten, J. M., Bean, B. W., Salisbury, C. D. and Chenault, E. W., 1998. High temperature composting of cattle feedlot manure kills weed seed. Applied Engineering in Agriculture **14**(4):377-380.

Willems, J.H. and Bik, L.P.M. 1998. Restoration of high species density in calcareous grassland: the role of seed rain and soil seed bank. Applied Vegetation Science **1**(1):91-100.

Williams, E.D. 1983. Germinability and enforced dormancy in seeds of species of indigenous grassland. Annals of Applied Botany **102**:557-566.

Williams, E.D. 1984. Changes during 3 years in the size and composition of the seed bank beneath a long-term pasture as influenced by defoliation and fertilizer regime. Journal of Applied Ecology **21**:603-615.

Williams, D.W., Jackson, L.L. and Smith, D.D. 2007. Effects of frequent mowing on survival and persistence of forbs seeded into a species-poor grassland. Restoration Ecology **15**(1):24-33.

Willms, W.D. and Jefferson, P.G.1993. Production characteristics of the mixed prairie: Constraints and potential. Canadian Journal of Animal Science **73**(4):765-778.

Willms, W.D., and Quinton, D.A. 1995. Grazing effects on germinable seeds on the fescue prairie. Journal of Range Management **48**(5):423-430.

Willms, W.D., Smoliak, S., and Bailey, A.W. 1986. Herbage production following litter removal on Alberta native grasslands. Journal of Range Management **39**(6):536-540.

Willms, W.D., Smoliak, S., and Dormaar, J.F. 1985. Effects of stocking rate on a rough fescue grassland vegetation. Journal of Range Management **38**(3):220-225.

Wilsey, B.J. and Polley, H.W. 2002. Reductions in grassland species evenness increase dicot seedling invasion and spittle bug infestation. Ecology Letters **5**(5):676-684.

Wilson Jr., R.G. 1979. Germination and seedling development of Canada thistle (*Cirsium arvense*). Weed Science **27**(2): 146-151.

Wilson, S.D. 1988. The effects of military tank traffic on prairie: a management model. Environmental Management **12**(3):397-403.

Wilson, S.D. 1989. The suppression of native prairie by alien species introduced for revegetation. Landscape and Urban Planning **17**(2):113-119.

Wilson, S.D. and Pärtel, M. 2003. Extirpation or coexistence? Management of a persistent introduced grass in a prairie restoration. Restoration Ecology **11**(4):410-416.

Wilson, B. J., Wright, K. J. and Butler, R. C. 1993. The effect of different frequencies of harrowing in the autumn or spring on winter wheat, and on the control of *Stellaria media* (L.) vill., *Galium aparine* L. and *Brassica napus* L. Weed Research **33**(6):501-506.

Wolf, J. J., Beatty, S.W., Carey, and G. 2008. Invasion by sweet clover (*Melilotus*) in montane grasslands, Rocky Mountain National Park. Annals of the Association of American Geographers **93**(3): 531-543.

Wolf, J.J., Beatty, S.W. and Seastedt, T.R. 2004. Soil characteristics of Rocky Mountain National Park grasslands invaded by *Melilotus officinalis* and *M. alba*. Journal of Biogeography **31**(3):415-424.

Woodcock, B.A., Savage, J., Bullock, J.M., Nowakowski, M., Orr, R., Tallowin, J.R.B. and Pywell, R.F. 2014. Enhancing floral resources for pollinators in productive agricultural grasslands. Biological Conservation **171**:44-51.

Wright, H.A., and Bailey, A.W. 1982. Fire ecology: United States and southern Canada. John Wiley & Sons.

Wu, C.X., Guo, X.X., Li, Z.H. and Shen, Y.X. 2010. Feasibility of using the allelopathic potential of yellow sweet clover for weed control. Allelopathy Journal **25**(1):173-183.

Wuerthner, G. 1997. Viewpoint: The black-tailed prairie dog: headed for extinction? Journal of Range Management **50**(5):459-466.

[WWF] World Wildlife Fund. 2016. Plowprint report: Facts & Figures. Accessed February 22, 2017. https://c402277.ssl.cf1.rackcdn.com/publications/947/files/original/plowprint_AnnualReport_2016_Final_REV09192016.pdf

Xiao, J., Wang, Y.F., Shi, P., Yang, L. and Chen, L.D. 2014. Potential effects of large linear pipeline construction on soil and vegetation in ecologically fragile regions. Environmental Monitoring and Assessment **186**(11):8037-8048.

Yeomans, J.C. and Bremner, J.M. 1991. Carbon and nitrogen analysis of soils by automated combustion techniques. Communication in Soil Science and Plant Analysis **22**(9-10):843-850.

Young, J. E., Sánchez-Azofeifa, G. A., Hannon, S. J. and Chapman, R. 2006. Trends in land cover change and isolation of protected areas at the interface of the southern boreal mixedwood and aspen parkland in Alberta, Canada. Forest Ecology and Management **230**(1):151-161.

Zhan, X., Li, L., and Cheng, W. 2007. Restoration of *Stipa krylovii* steppes in Inner Mongolia of China: Assessment of seed banks and vegetation composition. Journal of Arid Environments **68**(2):298-307.

APPENDIX

APPENDIX A. Chapter 3.

Appendix A.1. Producer management survey from 2013.

Survey of Pasture Seed-Bank Composition in the Aspen Parkland:

Supplemental Information on Pasture Management History

*** NOTE: All information collected in this survey will remain confidential ***

Name of producer: _________________________________

Pasture location: ___ Sect ___ - TP ___ - RG ___ Address:_____________________________________

Phone: _____ Would you like a copy of the final summary results? YES NO

If Yes, E-mail address (if applicable): ____

Is the land: OWNED RENTED

For how many years have you farmed this land: _________

Land Use History:

1. To the best of your knowledge, has this pasture ever been cultivated?

YES NO UNKNOWN

If YES, approximate year it was last cultivated? ______________________

If cultivated, was the pasture seeded? YES NO (i.e. Abandoned land) DON'T KNOW

If seeded, forage mix at time of seeding (grasses, legumes, etc.)? ___________________

2. Has the pasture been sprayed with herbicide(s) in the last three years? YES NO

If YES, with what herbicide(s)? ____

At what rate was herbicide applied? _____________Date of Last Application? ________

Target weeds: ________________________________

3. Was the pasture ever burned? YES NO

If YES, why did the fire occur? WILDFIRE PRESCRIBED-BURN

Current Management:

4. Is the area grazed? YES NO

If YES, by what kind of livestock? COW/CALF PAIRS YEARLINGS HORSES

OTHER: ____

Number of animals? _____________ For how long? ____________________ months/yr

Approximate timing of grazing each year? Start: __________ End: ____________

Do you rotate pasture use during summer?

YES NO If yes, length of rest period? ___ weeks

Number of pastures in the rotation and approximate size? __________ ac or ha

Are cattle fed hay (on this pasture) over the winter? YES NO

5. Is the pasture fertilized? YES NO

If so, how often and at what time of year? ____________________

If so, at what approximate rate? _____ N _____ P _____K _____S lb/ac kg/ha

6. Has the pasture been treated with manure? YES NO

7. Do you swath or mow your pasture? YES NO

If so, when? Summer (JULY) FALL (SEPTEMBER)

8. Other management (circle all those that apply)?

AERATION HARROWING OVERSEEDING

If overseeding, how long ago and what forage mix? _____________________________

9. Common pests (circle all that apply)?

GROUND SQUIRRELS POCKET GOPHERS GRASSHOPPERS

10. Other comments on land use history of the field

____
____
____
____
____

11. Has there been pressure from the oil/gas industry (or other) to develop your land?

YES NO I ALREADY HAVE DEVELOPMENTS (ROADS, PIPELINES, WELLS, PUMPJACKS, GRAVELPITS)

Comments

____
____

Appendix A.2. Simplified tame pasture assessment form used during the pasture field survey.

Range Health Assessment

Date:
Site:

Dominant Species

Grasses & Grasslikes	*Cover*	*Forbs*	*Cover*	*Shrubs*	*Cover*	*Trees*	*Cover*

1. Pasture Composition

1 A Tame Pasture

% of the cover (relative) is introduced forage species.

Score

12	90% or Greater
9	75% to 89%
5	50% to 74%

1 B Modified Tame Pasture

% of the cover (relative) is introduced & native forage species.

Score

9	75% or greater
5	50% to 74%
0	Less than 40%

2. Plant Composition Shift

2.1 Forage Species Shift

% of forage cover (relative) is from tall, productive, introduced and native forage species.

Score

14	75% or Greater
7	40% to 74%
0	Less than 40%

2.2 Weedy & Disturbance Based Species Shift

% of cover (absolute) from weedy and disturbance induced species

Score

14	25% or Less
7	26% to 49%
0	50% or Greater

3. Hydraulic Function & Nutrient Cycling

Score

25 Distinct litter layer visible. Litter has uniform distribution across pasture with less than 5% of the pasture lacking adequate cover. Hand raked litter from a ¼ m^2 plot is estimated at 450 lbs/acre or more (~one handful).

16 Distinct litter layer visible but cover is reduced and there is no uniform layer. Litter reduced on 5% to 25% of the pasture with these areas having little or litter. Hand raked litter from ¼ m^2 plot estimated at 250 to 450 lbs/acre (~ ½ of one handful).

8 Thin litter layer present throughout or in scattered patches. About 25% to 67% of pasture has inadequate litter cover, estimated at 125 to 250 lbs/acre (~ ½ to ¼ of one handful).

0 Litter Sparse or absent over greater than 67% of the area, estimated at less than 125 lbs/acre (less than ¼ of one handful).

4. Is There Accelerated Soil Erosion? Site Normally (circle) Stable/Unstable

4.1 Erosion Evidence

Score

10 No visible macro or micro evidence of soil movement, deposition of soil/litter, plant pedestalling, coarse sand or aggregate remnants, hoof shear, soil compaction, flow patterns or scouring.

7 No macro evidence as above. Some micro evidence of hoof shear or plant pedestalling. Old erosion features may be stable and vegetated.
4 Erosion features active but limited to site with no off site movement. Flow patterns have well defined branches.
0 Macro and micro evidence of extreme soil movement with most material being carried off site. Flow patterns obvious, rills are abundant and deep, deep gullies, erosion features active, plants with exposed roots.

4.2 Bare Soil
% of area is exposed soil that is management caused.

4.2 A Dry Mixedgrass or Mixedgrass
Score
5 10% or Less
3 11% to 20%
1 21% to 59%
0 Greater than 50%

4.2 B Foothills Fescue, Foothills and Central Parkland, Montane, Boreal Forest
Score
5 5% or Less
3 6% to 10%
1 11% to 15%
0 16% or Greater

Human caused bare soil (%) _____

5. Are Noxious Weeds Present?

5.1 Cover
Score
5 None Present
3 Present with cover (absolute) <1%
1 Cover (absolute) 1% to 15%
0 Cover (absolute) >15%

5.2 Density Distribution
Score
5 None Present
3 Low Infestation (Dist. Class 1-3)
1 Moderate Infestation (Dist. Class 4-7)
0 Heavy Infestation (Dist. Class 8-13)

Dominant Species	*% Cover*	*Density Dist.*

6. Does the Site Have Woody Regrowth?

6.1 Cover
Woody regrowth present with % total cover (absolute)
Score
6 Less than 5%
3 5% to 15%
0 Greater than 15%
N/A Not Scored

6.2 Density Distribution
Score
4 Low Infestation (Dist. Class 1-3)
2 Moderate Infestation (Dist. Class 4-7)
0 Heavy Infestation (Dist. Class 8-13)
N/A Not Scored

Dominant Species	*% Cover*	*Density Dist.*

Grazing Intensity (estimated Long Term): U U-L L-M M M-H H

Observed Utilization ______ % **Vegetation Height** ______ cm

Trend (apparent): Upward Downward Stable Unknown

Overall Score

Total Score ______ Out of ______

0--50	------------------------------74	--------------------------------100
< 50%	50-74%	75-100%
Unhealthy	Healthy with problems	Healthy

Appendix A.3. Treatment Table for Chapters 3 to 5.

Table A.1. Summary of management factors, plant community types, and rangeland health questions used to analyze plant community, seed bank, and biophysical responses.

Condition	Category	Treatment Level	Notes
Community	Plant Community Type	Modified-Tame	
		Tame	
	Natural Subregion	Parkland	Central Parkland
		Boreal	Includes Central Mixedwood and Dry Mixedwood
Management	Ownership	Owned	
		Rented	
	Cultivation	Cultivated	
		Never Cultivated	
		Unknown	
	Grazing System	Abandoned (None)	
		Continuous	
		Rotational	
	Timing of Grazing	Abandoned	
		All Year	
		Growing Season	
		Winter	
	System x Timing	Abandoned	
		All Year (Continuous)	
		Growing Season (Continuous)	
		Growing Season (Rotational)	
		Winter (Rotational)	
	Herbicide	Sprayed in Last 3 Years	
		Not Sprayed Recently	
	Fertilized	Fertilized	
		Not Fertilized	
	Manure Spreading	Manured	
		Not Manured	
	Harrowed	Harrowed	
		Not Harrowed	
	Aerated	Aerated	
		Not Aerated	
	Swathed or Mowed	Swath-Mow	
		No Swath-Mow	
	Fed Hay in Pasture Sampled	Hay	
		No Hay	
	Burrowing Mammals	Present	
		Absent	
	Fire (Survey)	Present	
		Absent	
	Fire (Charcoal in Soil)	Present	
		Absent	
Rangeland Health	Cover of Tall Productive Forages	0	
		7	
		14	
	Soil Erosion	4	
		7	
		10	
	Anthropogenic Bare Soil	0	
		3	
		5	
	Noxious Weed Cover	1	
		3	
		5	
	Woody Spp Cover	3	
		6	
	Woody Spp Density	0	
		2	
		4	
	Grazing Intensity	U	No animal use.
		L	
		LM	
		M	
		MH	
		H	
	Health	Healthy	RHA Score >75%
		Healthy with Problems	RHA Score 51 to 75%
		Unhealthy	RHA Score ≤ 50%

Appendix B. Chapter 4

Appendix B.1. Plant Community.

Table B.1. Summary of plant community and soil attributes observed, their abbreviations, units, and transformation for univariate tests.

Attribute	Variable	Abbreviation	Units	Transformation
Primary Functional Groups	Total Graminoids	-	%	
	Total Broad Leaf	-	%	sqrt
	Total Introduced	-	%	*
	Total Native	-	%	*
Functional Groups	Graminoids (grass-like taxa)	-	%	*
	Introduced Ruderal Forbs	-	%	sqrt
	Legumes (native & introduced)	-	%	sqrt
	Native Perennial Forbs	-	%	*
	Native Perennial Grasses	-	%	*
	Native Ruderal Forbs	-	%	*
	Noxious Weeds	-	%	*
	Ruderal Grasses	-	%	*
	Seeded (introduced) Grasses	-	%	
	Woody (shrubs and trees)	-	%	*
Indices	Species Richness	-	n/a	*
	Shannon's Diversity	-	n/a	
	Simpson's Diversity	-	n/a	x^2
	Pielou's Evenness	-	n/a	Log
Ground Cover Attributes	Basal Vegetation Cover	-	%	Log
	Litter Cover	-	%	
	Litter Depth	-	cm	Log
	Bare Ground Cover	-	%	*
	Manure Cover	-	%	*
Soil Properties	Total Carbon	C	%	*
	Total Nitrogen	N	%	*
	C:N Ratio	C:N	n/a	
	Organic Matter	OM	%	Log
	pH	pH	n/a	
	Electrical Conductivity	EC	µS/cm	
	Soil Surface Compaction	-	kg/cm^3	Sqrt
Soil Texture	Sand	-	%	Log
	Silt	-	%	
	Clay	-	%	Log

*Variables analysed with nonparametric tests

Appendix B.1.1. Summary tables for plant community NMDS ordinations.

Table B.1.1.1. Summary of significant management centroids arising from the NMDS ordination of plant community composition (P < 0.1) (Figure 4.3).

Management Factor	**r^2**	**P Value**	**Centroid**	**MDS 1**	**MDS 2**
Cultivation	0.21	**0.001**	Cultivated	-0.08	0.00
			Not Cultivated	1.13	0.11
			Unknown	-0.18	-0.04
Feeding Hay	0.09	**0.003**	Hay	-0.32	-0.05
			No Hay	0.13	0.17
			Unknown	-0.02	-0.15
Fertilization	0.05	**0.016**	Fertilized	-0.36	-0.36
			Not Fertilized	0.04	0.03
Fire (Survey)	0.03	0.075	Fire	0.26	0.12
			No Fire	-0.05	-0.02
Grazing Intensity	0.08	0.092	U	0.60	-0.18
			L	0.15	0.22
			LM	0.02	-0.06
			M	0.05	0.04
			MH	-0.15	-0.12
			H	-0.32	0.18
Harrowed	0.04	**0.018**	Harrowed	-0.20	0.06
			Not Harrowed	0.10	-0.03
Manure	0.05	**0.015**	Manured	-0.26	0.07
			Not Manured	0.08	-0.02

Table B.1.1.2. Significant biplot vectors for NMDS ordination of plant community composition (Figure 4.3).

Biplot		MDS 1	MDS 2	r^2	P Value
Soil Properties	OM	-1.00	-0.01	0.10	**0.007**
	EC	-0.72	-0.69	0.07	**0.041**
	pH	-0.42	-0.91	0.02	0.387
	N	-0.95	-0.31	0.10	**0.012**
	C	-1.00	0.00	0.11	**0.008**
	C:N Ratio	0.02	1.00	0.08	**0.020**
	Sand	0.87	-0.50	0.05	0.061
	Clay	-0.97	0.24	0.02	0.308
	Silt	-0.78	0.62	0.05	0.083
	Compaction	-0.77	-0.63	0.09	0.130
Litter Depth	Depth	-0.27	-0.96	0.00	0.799
Basal Cover	Vegetation	-0.06	1.00	0.00	0.787
	Litter	-0.02	-1.00	0.06	**0.049**
	Bare Ground	0.05	1.00	0.06	**0.049**
	Manure	-0.92	0.39	0.02	0.297
	Rock	-0.88	-0.48	0.02	0.294
	Lichen	0.80	-0.60	0.12	**0.024**
	Moss	0.94	-0.35	0.06	0.077
	Wood	0.38	0.93	0.11	**0.017**
Pasture Characteristics	Years Farmed	0.79	-0.61	0.07	0.119
	Pasture Age	0.62	-0.79	0.18	**0.001**
Rangeland Health	Total RHA Score	-0.13	-1.00	0.06	**0.048**
	Forage Cover	-0.45	-0.89	0.12	**0.002**
	Cover of Tall Productive Forages	-0.35	-0.94	0.06	**0.049**
	Weedy & Ruderal Cover	0.59	-0.81	0.02	0.377
	Hydraulic Function & Litter	0.00	-1.00	0.03	0.206
	Soil Erosion	0.51	0.86	0.04	0.109
	Anthropogenic Bare Soil	0.06	-1.00	0.03	0.230
	Noxious Weed Cover	0.93	0.37	0.02	0.348
	Noxious Weed Density	0.77	-0.64	0.02	0.442
	Woody Spp Cover	-0.96	-0.29	0.14	**0.003**
	Woody Spp Density	-0.55	-0.83	0.09	**0.003**
Similarity	Sorensen's	-0.25	0.97	0.01	0.753
Plant Community	Shannon's Diversity	0.70	0.71	0.61	**0.001**
	Simpson's Diversity	0.58	0.81	0.41	**0.001**
	Pielou's Evenness	-0.97	0.26	0.14	**0.001**
	Richness	0.92	0.40	0.68	**0.001**
	Total Veg. Cover	0.99	0.11	0.02	0.413
	Total Graminoids	-0.25	-0.97	0.12	**0.002**
	Total Broad Leaf	0.38	0.93	0.14	**0.001**
	Total Native	0.97	0.24	0.56	**0.001**
	Total Introduced	-0.97	-0.26	0.35	**0.001**
	Noxious Weeds	-1.00	0.03	0.01	0.765
	Legumes	-0.09	1.00	0.07	**0.027**
	Woody	0.95	-0.30	0.19	**0.002**
	Native Ruderal Forbs	0.72	0.69	0.12	**0.007**
	Native Perennial Forbs	0.96	-0.28	0.58	**0.001**
	Introduced Ruderal Forbs	-0.45	0.89	0.19	**0.001**
	Seeded Graminoids	-0.49	-0.87	0.31	**0.001**
	Native Grasses	0.61	0.79	0.19	**0.001**
	Ruderal Grasses	-0.20	0.98	0.05	0.075
	Graminoids	0.99	-0.11	0.44	**0.001**

Table B.1.1.3. Individual plant species' relationship to NMDS for plant community composition (P < 0.1) (Figure 4.3).

Species	MDS 1	MDS 2	r^2	P Value
Achillea millefolium	0.31	-0.19	0.41	**0.001**
Achnatherum hymenoidea	1.72	2.64	0.19	**0.019**
Agropyron pectiniforme	-0.18	-0.75	0.07	0.054
Agrostis scabra	3.33	-0.38	0.19	**0.010**
Alnus viridis	2.49	-1.12	0.13	**0.028**
Amelanchier alnifolia	2.49	-1.12	0.13	**0.028**
Androsace septentrionalis	0.32	-0.66	0.16	**0.006**
Antennaria parvifolia	1.28	-0.60	0.14	**0.007**
Antennaria rosea	3.33	-0.38	0.19	**0.010**
Arabis hirsuta	1.22	1.59	0.19	**0.002**
Arabis holboellii var. *retrofracta*	1.26	-1.01	0.05	0.092
Artemisia absinthium	0.57	-0.92	0.06	0.085
Artemisia frigida	1.29	-0.51	0.21	**0.002**
Artemisia ludoviciana	3.10	-0.10	0.27	**0.001**
Astragalus agrestis	2.97	0.06	0.27	**0.001**
Axyris amaranthoides	-0.15	-1.08	0.09	**0.039**
Botrychium lunaria	1.26	-1.01	0.05	0.092
Bouteloua gracilis	2.49	-1.12	0.13	**0.028**
Bromus biebersteinii	0.44	-0.19	0.12	**0.006**
Bromus inermis	-0.14	-0.34	0.22	**0.001**
Calamovilfa longifolia	2.49	-1.12	0.13	**0.028**
Campanula rotundifolia	1.27	0.32	0.08	**0.045**
Capsella bursa-pastoris	-0.17	0.70	0.05	0.076
Carex aenea	0.41	-0.22	0.07	**0.028**
Carex atherodes	0.71	-0.32	0.12	**0.031**
Carex aurea	2.32	0.11	0.17	**0.011**
Carex bebbii	1.14	0.76	0.16	**0.004**
Carex filifolia	2.49	-0.19	0.32	**0.001**
Carex pensylvanica	1.26	-1.01	0.05	0.092
Carex praegracilis	2.75	-0.03	0.30	**0.001**
Cerastium arvense	1.46	-0.50	0.21	**0.002**
Chenopodium album	-0.68	-0.20	0.10	**0.027**
Chenopodium pratericola	1.72	2.64	0.19	**0.019**
Comandra umbellata	2.68	-0.46	0.24	**0.002**
Crepis tectorum	0.07	1.07	0.22	**0.002**
Dactylis glomerata	-0.22	0.62	0.12	**0.007**
Danthonia intermedia	1.62	0.03	0.23	**0.001**
Deschampsia cespitosa	0.16	0.31	0.06	0.066
Draba nemorosa	-0.20	1.06	0.07	0.055
Elymus lanceolatus	1.72	2.64	0.19	**0.019**
Elymus trachycaulus ssp. *subsecendus*	2.55	-0.37	0.22	**0.004**
Elymus tracychaulus ssp. *trachycaulus*	1.72	2.64	0.19	**0.019**
Elytrigia repens	-0.42	-0.17	0.14	**0.004**
Equisetum arvense	0.08	-0.16	0.06	0.061
Festuca hallii	2.70	-0.11	0.27	**0.001**
Festuca rubra	-0.26	0.34	0.06	0.060
Festuca saximontana	1.72	2.64	0.19	**0.019**
Fragaria virginiana	0.49	-0.04	0.11	**0.013**
Galeopsis tetrahit	-1.25	-0.73	0.08	**0.038**
Galium boreale	1.02	0.11	0.24	**0.001**
Geum triflorum	2.81	-0.48	0.31	**0.001**
Hesperostipa comata	2.59	-0.29	0.22	**0.001**
Hesperostipa curtiseta	1.78	-0.36	0.06	0.082
Heterotheca villosa	2.32	-0.68	0.35	**0.001**
Heuchera richardsonii	3.33	-0.38	0.19	**0.010**
Hierochloe odorata	3.33	-0.38	0.19	**0.010**
Houstonia longifolia	1.83	-0.70	0.12	**0.024**
Juncus acrticus ssp. b*alticus*	2.06	-0.26	0.36	**0.001**
Juncus tenuis	3.33	-0.38	0.19	**0.010**
Kochia scoparia	2.68	0.42	0.12	**0.035**
Koeleria macrantha	2.40	-0.16	0.21	**0.002**
Linaria vulgaris	1.26	-1.01	0.05	0.092
Lithospermum incisum	2.49	-1.12	0.13	**0.028**
Medicago sativa	-0.43	0.10	0.08	**0.029**
Melilotus alba	1.46	1.82	0.21	**0.003**
Melilotus officinalis	1.72	2.64	0.19	**0.019**

Moehringia laterifolia	1.26	-1.01	0.05	0.092
Nassella viridula	2.15	1.64	0.24	**0.002**
Pascopyrum smithii	0.23	0.25	0.07	0.057
Penstemon procerus	3.33	-0.38	0.19	**0.010**
Phleum pratense	-0.10	0.16	0.05	0.084
Poa pratensis	-0.15	-0.21	0.13	**0.002**
Poa secunda	2.10	1.76	0.26	**0.002**
Potentilla gracilis	3.05	-0.03	0.30	**0.001**
Potentilla pensylvanica	0.89	-0.07	0.07	0.051
Pulsatilla patens	2.49	-1.12	0.13	**0.028**
Ranunculus rhomboideus	1.83	-0.73	0.13	**0.013**
Rosa acicularis	0.83	0.13	0.08	**0.032**
Sisyrinchium montanum	0.56	0.24	0.09	**0.018**
Solidago missouriensis	1.54	-0.24	0.20	**0.004**
Sonchus arvensis	0.44	0.32	0.06	0.062
Spergula arvensis	-0.84	-0.29	0.05	0.074
Stellaria longifolia	1.14	-0.07	0.25	**0.001**
Symphoricarpos occidentalis	1.54	-0.76	0.24	**0.001**
Taraxacum officinale	-0.17	0.08	0.15	**0.004**
Thalaspi arvense	-0.48	0.53	0.07	0.053
Thermopsis rhombifolia	2.31	-1.09	0.14	**0.020**
Thinopyrum intermedium	1.55	-1.04	0.08	**0.048**
Tragopogon dubius	1.84	-0.27	0.13	**0.018**
Trifoloium hybridum	-0.06	0.17	0.07	**0.036**
Vicia americana	0.33	0.05	0.07	0.055
Viola adunca	1.19	-0.83	0.16	**0.010**

Appendix B.1.2. Indicator species analysis (ISA) tables for plant community responses to management and rangeland health assessment questions.

Table B.1.2.1. Indicator analysis linking plant community species association with various management factors (P < 0.05).

Management	Category	Species	A	B	P value
Ownership	Owned	*Bromus bieberstienii*	1.00	0.50	0.045
	Rented	*Equisetum arvense*	0.55	0.60	0.034
		Stellaria longipes	0.86	0.20	0.044
Cultivation	Never Cultivated	*Achillea millefolium*	0.81	0.88	0.002
		Antennaria parvifolia	0.91	0.38	0.005
		Artemisia frigida	0.83	0.38	0.010
		Artemisia ludoviciana	1.00	0.25	0.006
		Astragalus agrestis	1.00	0.25	0.006
		Campanula rotundifolia	1.00	0.38	0.001
		Carex aurea	1.00	0.25	0.008
		Carex bebbii	0.88	0.25	0.010
		Carex filifolia	0.88	0.50	0.001
		Carex praegracilis	1.00	0.38	0.001
		Carex praticola	0.84	0.25	0.022
		Cerastium arvense	0.95	0.38	0.005
		Comandra umbellate	0.86	0.25	0.021
		Danthonia intermedia	0.93	0.50	0.001
		Elymus trachycaulus ssp. *subsecundus*	1.00	0.25	0.005
		Festuca hallii	1.00	0.38	0.001
		Fragaria virginiana	0.76	0.50	0.015
		Galium boreale	0.96	0.50	0.004
		Geum triflorum	0.98	0.25	0.013
		Heterotheca villosa	0.95	0.25	0.005
		Juncus arcticus ssp. *balticus*	0.97	0.25	0.005
		Koeleria macrantha	0.98	0.25	0.010
		Nassella viridula	1.00	0.25	0.005
		Pascopyrum smithii	0.67	0.63	0.005
		Poa secunda	1.00	0.25	0.005
		Potentilla gracilis	1.00	0.25	0.006
		Sisyrinchium montanum	0.70	0.50	0.008
		Solidago missouriensis	0.84	0.63	0.001
		Stellaria longifolia	0.91	0.38	0.003
		Viola adunca	0.89	0.25	0.023
	Cultivated	*Phleum pretense*	0.76	0.58	0.030
	Unknown	*Elytrigia repens*	0.62	0.94	0.008
Grazing System	None (Abandoned)	*Agrostis scabra*	1.00	0.25	0.032
		Antennaria rosea	1.00	0.25	0.032
		Artemisia ludoviciana	0.97	0.25	0.017
		Carex Spp.	0.93	0.25	0.040
		Danthonia intermedia	0.99	0.50	0.003
		Elymus trachycaulus ssp. *subsecundus*	0.93	0.25	0.031
		Festuca hallii	0.97	0.25	0.018
		Galium boreale	0.96	0.25	0.027
		Geum triflorum	0.97	0.25	0.011
		Heterotheca villosa	0.86	0.25	0.035
		Heuchera richardsonii	1.00	0.25	0.032
		Hierochloe odorata	1.00	0.25	0.032
		Juncus arcticus ssp. *balticus*	0.90	0.25	0.014
		Juncus tenuis	1.00	0.25	0.032
		Penstemon procerus	1.00	0.25	0.032
		Potentilla gracilis	0.95	0.25	0.017
		Sisyrinchium montanum	0.76	0.50	0.033
		Stellaria longipes	0.95	0.50	0.005
		Vicia americana	0.61	0.75	0.021
	Continuous + Rotational	*Trifolium repens*	0.98	0.76	0.017
Timing of Grazing	Never (Abandoned)	*Danthonia intermedia*	0.99	0.50	0.013
		Stellaria longipes	0.97	0.50	0.004
		Vicia americana	0.72	0.75	0.037
	All Year	*Plantago major*	0.91	0.67	0.035

	Winter	*Astragalus cicer*	0.98	0.50	0.022
		Hordeum vulgatum	1.00	0.33	0.030
		Pascopyrum smithii	0.91	1.00	0.005
	Abandoned + All Year + Winter	*Elytrigia repens*	0.84	0.93	0.029
Gr. System x Timing of Gr.	Never (Abandoned)	*Danthonia intermedia*	0.99	0.50	0.013
		Stellaria longipes	0.97	0.50	0.004
		Sisyrichium montanum	0.68	0.50	0.049
		Vicia americana	0.72	0.75	0.037
	All Year	*Plantago major*	0.91	0.67	0.035
	Winter	*Astragalus cicer*	0.98	0.50	0.022
		Hordeum vulgatum	1.00	0.33	0.030
		Pascopyrum smithii	0.91	1.00	0.005
	All Year + Continuous + Rotational + Winter	*Trifolium repens*	0.99	0.76	0.045
Herbivore Type	Multiple Herbivores	*Agropyron pectiniforme*	0.92	0.50	0.015
	Sheep/Alpaca	*Bromus anomalus*	0.97	0.50	0.002
	No Livestock (Abandoned)	*Danthonia intermedia*	0.97	0.50	0.009
		Stellaria longipes	0.97	0.50	0.007
		Vicia americana	0.69	0.75	0.016
Herbicide	Sprayed	*Cirsium arvense*	0.74	0.62	0.015
		Festuca rubra	0.78	0.63	0.007
		Schedonorus arundinaceus	1.00	0.13	0.021
Fertilization	Fertilized	*Bromus bieberstienii*	0.75	0.67	0.031
	Not Fertilized	*Trifolium hybridum*	0.98	0.84	0.001
		Trifolium repens	0.96	0.78	0.002
Manure	Manure Spread	*Lepidium densiflorum*	1.00	0.12	0.022
		Lollium perenne	1.00	0.12	0.022
		Silene latifolia alba	0.70	0.16	0.044
		Thalapsi arvense	0.86	0.20	0.040
	None	*Fragaria virginiana*	0.97	0.25	0.037
		Pascopyrum smithii	0.98	0.27	0.030
		Trifolium pretense	0.92	0.29	0.043
Harrowed	Harrowed	*Plantago major*	0.91	0.38	0.001
		Polygonum convolvulus	0.82	0.21	0.032
		Silene latifolia ssp. Alba	0.93	0.12	0.046
	Not Harrowed	*Agropyron pectiniforme*	1.00	0.15	0.044
		Galium boreale	0.99	0.18	0.048
Aeration	Aerated	*Poa palustris*	0.89	0.75	0.006
		Hordeum vulgatum	1.00	0.25	0.038
		Symphyotrichum leave	0.98	0.25	0.040
		Silene latifolia ssp. *Alba*	0.96	0.25	0.490
Swathed or Mowed	Swath/Mowed	*Medicago sativa*	0.89	0.67	0.001
		Trifolium pretense	0.85	0.44	0.024
		Spergula arvensis	0.97	0.22	0.019
Fed Hay (in pasture)	Hay	*Carex praticola*	0.90	0.13	0.048
		Chenopodium album	0.79	0.56	0.001
		Descurainia Sophia	0.69	0.25	0.011
		Erysimum cheiranthoides	0.89	0.25	0.003
		Lepidium densiflorum	0.96	0.13	0.038
		Thalapsi arvense	0.82	0.25	0.031
	No Hay	*Symphyotrichum laeve*	1.00	0.12	0.038
		Dactylis glomerata	0.86	0.37	0.006
Burrowing Mammals	Absent	*Fragaria virginiana*	0.84	0.26	0.030
		Lathyrus ochroleucus	0.99	0.17	0.004
		Rosa acicularia	0.91	0.14	0.017
Recent Fire	Fire (Survey)	*Alopecurus pratensis*	0.78	0.33	0.015
		Arabis hirsute	1.00	0.13	0.021
		Aster ciliates	1.00	0.13	0.019
		Bromus inermis pumpelianus	0.70	0.13	0.043

		Dactylis glomerata	0.76	0.40	0.035
		Fragaria virginiana	0.88	0.53	0.001
		Galium boreale	0.67	0.40	0.010
		Lathyrus ochroleucus	0.99	0.40	0.001
		Lathyrus venosus	1.00	0.13	0.016
		Phleum pretense	0.78	0.67	0.013
		Rosa acicularis	0.95	0.27	0.002
		Sonchus arvensis	0.88	0.33	0.004
		Thalictrum venulosum	1.00	0.13	0.019
		Trifolium pretense	0.72	0.53	0.008
Historical Fire	Fire (Charcoal in Soil)	*Fragaria virginiana*	0.86	0.35	0.004
		Lathyrus ochroleucus	0.99	0.23	0.001
		Rosa acicularis	0.88	0.13	0.037
		Trifolium pretense	0.65	0.39	0.040
		Vicia americana	0.80	0.39	0.003

ISA ran in R using *indicspecies:multipatt* (Caceres and Legendre, 2009).
A = Probability of occurring, B = Fidelity
Permutations = 999

Table B.1.2.2. Indicator plant species analysis assessing significant Rangeland Health Assessment (RHA) categories describing shifts in plant community composition (P < 0.05).

Management	Category	Species	A	B	P value
Plant Community	Modified-Tame	*Achillea millefolium*	0.86	0.83	**0.002**
		Androsace septentrionalis	0.87	0.33	**0.016**
		Antennaria parvifolia	0.99	0.42	**0.001**
		Artemisia frigida	0.98	0.42	**0.001**
		Artemisia ludoviciana	1.00	0.17	**0.016**
		Astragalus agrestis	1.00	0.17	**0.016**
		Campanula rotundifolia	1.00	0.25	**0.002**
		Carex aurea	1.00	0.17	**0.012**
		Carex bebbii	1.00	0.25	**0.001**
		Carex filifolia	1.00	0.50	**0.001**
		Carex praegracilis	1.00	0.25	**0.001**
		Cerastium arvense	1.00	0.42	**0.001**
		Comandra umbellate	1.00	0.25	**0.001**
		Danthonia intermedia	1.00	0.42	**0.001**
		Elymus trachycaulus ssp. *Subsecundus*	1.00	0.17	**0.019**
		Festuca hallii	1.00	0.25	**0.004**
		Fragaria virginiana	0.78	0.50	**0.006**
		Galium boreale	0.95	0.42	**0.002**
		Geum triflorum	1.00	0.33	**0.001**
		Hesperostipa comate	1.00	0.17	**0.012**
		Heterotheca villosa	1.00	0.33	**0.001**
		Houstonia longifolia	1.00	0.17	**0.009**
		Juncus balticus	0.98	0.25	**0.003**
		Koeleria macrantha	1.00	0.25	**0.001**
		Nassella viridula	1.00	0.17	**0.007**
		Pascopyrum smithii	0.72	0.58	**0.009**
		Poa secunda	1.00	0.17	**0.007**
		Potentilla gracilis	1.00	0.17	**0.016**
		Ranunculus rhomboids	1.00	0.17	**0.009**
		Rosa acicularis	0.70	0.25	**0.036**
		Sisyrinchium montanum	0.89	0.50	**0.001**
		Solidago missouriensis	0.99	0.50	**0.001**
		Stellaria longifolia	0.95	0.33	**0.002**
		Stellaria longipes	0.92	0.17	**0.032**
		Symphoricarpos occidentalis	0.98	0.25	**0.001**
		Thermopsis rhombifolia	1.00	0.17	**0.015**
		Thinopyrum intermedium	1.00	0.17	**0.012**
		Viola adunca	0.98	0.33	**0.001**
Forage Cover	Score 9	*Artemisia frigida*	0.97	0.22	**0.017**
		Carex filifolia	0.90	0.19	**0.045**
		Carex praegracilis	1.00	0.11	**0.038**
		Comandra umbellate	1.00	0.11	**0.046**
		Festuca hallii	1.00	0.11	**0.036**
		Galium boreale	0.88	0.26	**0.044**
		Geum triflorum	1.00	0.15	**0.037**
		Heterotheca villosa	1.00	0.15	**0.035**
		Juncus bufonius	1.00	0.11	**0.037**
		Koeleria macrantha	1.00	0.11	**0.038**
		Solidago missouriensis	0.97	0.22	**0.012**
		Stellaria longifolia	0.94	0.19	**0.027**
	Score 9 + 5	*Cerastium arvense*	1.00	0.19	**0.029**
		Danthonia intermedia	1.00	0.14	**0.047**
		Fragaria virginiana	0.80	0.36	**0.028**
		Pascopyrum smithii	0.74	0.39	**0.041**
		Sisyrinchium montanum	0.93	0.22	**0.045**
	Score 5	*Hordeum jubatum*	0.95	0.22	**0.040**
		Stellaria longipes	0.92	0.22	**0.008**
		Thalapsi arvense	0.94	0.44	**0.001**
Cover of Tall Productive Forages	Score 0	*Amaranthus blitoides*	1.00	0.50	**0.021**
		Capsella bursa-pastoris	0.93	1.00	**0.002**
		Chenopodium album	0.86	1.00	**0.003**
		Descurainia Sophia	0.93	1.00	**0.003**
		Gnaphalium uliginosum	1.00	0.50	**0.039**
		Hordeum jubatum	0.99	0.50	**0.023**

		Lepidium densiflorum	0.99	0.50	**0.034**
		Plantago major	0.92	1.00	**0.008**
		Senecio vulgaris	1.00	0.50	**0.020**
		Thalapsi arvense	0.96	1.00	**0.005**
Weedy & Ruderal Cover	Score 7	*Elytrigia repens*	0.65	1.00	**0.031**
		Hordeum jubatum	0.98	0.20	**0.030**
		Taraxacum officinale	0.73	1.00	**0.001**
		Thalapsi arvense	0.97	0.40	**0.002**
Hydraulic Function & Litter	Score 0	*Alopecurus aequalis*	1.00	0.25	**0.043**
		Amaranthus blitoides	1.00	0.25	**0.042**
		Capsella bursa-pastoris	0.85	0.50	**0.006**
		Chenopodium album	0.60	0.75	**0.014**
		Descurainia Sophia	0.78	0.25	**0.048**
		Medicago sativa	0.72	0.75	**0.016**
		Poa palustris	0.71	0.50	**0.024**
		Plagiobothrys scouleri	0.99	0.25	**0.043**
		Plantago major	0.82	0.50	**0.026**
		Polygonum aviculaire	0.79	0.50	**0.023**
		Senecio vulgaris	1.00	0.25	**0.034**
		Spergula arvensis	0.88	0.25	**0.028**
	Score 0 + 8	*Hordeum jubatum*	1.00	0.18	**0.048**
Soil Erosion	Score 10	*Lathyrus ochroleucus*	0.97	0.15	**0.045**
		Vicia americana	0.94	0.37	**0.004**
	Score 7 + 4	*Plantago major*	0.96	0.34	**0.023**
	Score 4	*Agropyron cristatum*	0.63	0.27	**0.035**
		Erysimum cheirantoides	0.88	0.20	**0.007**
		Juncus bufonius	0.65	0.13	**0.041**
Anthropogenic Bare Soil	Score 0	*Achnatherum hymenoides*	1.00	0.20	**0.049**
		Amaranthus blitoides	1.00	0.20	**0.043**
		Arabis hirsute	0.98	0.20	**0.041**
		Capsella bursa-pastoris	0.91	0.40	**0.010**
		Carex bebbii	0.93	0.20	**0.035**
		Chenopodium album	0.70	0.80	**0.007**
		Chenopodium pratericola	1.00	0.20	**0.049**
		Crepis tectorum	0.92	0.20	**0.043**
		Elymus lanceolatus	1.00	0.20	**0.049**
		Elymus trachycaulus ssp. *trachycaulus*	1.00	0.20	**0.049**
		Festuca saximontana	1.00	0.20	**0.049**
		Melilotus alba	0.96	0.20	**0.038**
		Melilotus officinalis	1.00	0.20	**0.049**
		Plantago major	0.89	0.60	**0.012**
		Poa palustris	0.73	0.60	**0.014**
		Poa secunda	0.95	0.20	**0.035**
		Polygonum aviculaire	0.93	0.60	**0.001**
		Senecio vulgaris	1.00	0.20	**0.034**
		Sonchus arvensis	0.65	0.40	**0.037**
		Spergula arvensis	0.85	0.20	**0.050**
Noxious Weed Cover	Score 5	*Astragalus cicer*	0.97	0.12	**0.047**
		Axyris amaranthoides	0.68	0.18	**0.043**
		Comandra umbellate	0.99	0.12	**0.047**
	Score 1	*Cirsium arvense*	0.86	0.71	**0.001**
		Tanacetum vulgare	0.98	0.12	**0.023**
Noxious Weed Density	Score 5	*Astragalus cicer*	0.91	0.12	**0.032**
		Comandra umbellate	0.98	0.12	**0.027**
		Hesperostipa comate	1.00	0.12	**0.023**
	Score 1	*Descurainia Sophia*	0.69	0.24	**0.021**
		Thalapsi arvense	0.78	0.29	**0.030**
	Score 0	*Bromus anomalus*	0.99	0.12	**0.049**
		Cirsium arvense	0.82	0.70	**0.001**
	Score 1 + 5	*Axyris amaranthoides*	1.00	0.13	**0.035**
		Capsella bursa-pastoris	0.98	0.18	**0.017**
Woody Spp Cover	Score 3	*Achillea millefolium*	0.72	0.79	**0.003**
		Antennaria parvifolia	0.84	0.29	**0.009**

		Arabis hirsute	1.00	0.14	**0.020**
		Artemisia frigida	0.97	0.21	**0.011**
		Campanula rotundifolia	0.89	0.14	**0.039**
		Carex bebbii	0.96	0.14	**0.016**
		Carex filifolia	0.91	0.29	**0.006**
		Cerastium arvense	0.97	0.36	**0.001**
		Fragaria virginiana	0.87	0.50	**0.003**
		Galium boreale	0.68	0.50	**0.006**
		Heterotheca villosa	0.93	0.21	**0.003**
		Houstonia longifolia	1.00	0.14	**0.016**
		Koeleria macrantha	0.75	0.14	**0.040**
		Lathyrus ochroleucus	0.95	0.21	**0.013**
		Lathyrus venosus	1.00	0.14	**0.023**
		Potentilla pensylvanica	1.00	0.14	**0.018**
		Ranunculus rhomboids	1.00	0.14	**0.016**
		Rosa acicularis	0.98	0.43	**0.001**
		Solidago missouriensis	0.92	0.21	**0.023**
		Stellaria longifolia	0.81	0.21	**0.036**
		Symphoricarpos occidentalis	0.98	0.21	**0.002**
		Symphyotrichum leave	0.69	0.36	**0.007**
		Thalictrum venulosum	1.00	0.14	**0.014**
		Thermopsis rhombifolia	1.00	0.14	**0.023**
		Vicia americana	0.83	0.43	**0.016**
		Viola adunca	0.91	0.21	**0.011**
Wood Density	Score 2	*Antennaria parvifolia*	0.86	0.25	**0.036**
		Carex filifolia	0.90	0.25	**0.026**
		Geum triflorum	0.66	0.25	**0.028**
		Heterotheca villosa	0.66	0.25	**0.046**
		Symphoricarpos occidentalis	0.89	0.25	**0.026**
		Thermopsis rhombifolia	1.00	0.25	**0.008**
	Score 0	*Bromus anomalus*	0.99	0.19	**0.034**
		Carex bebbii	1.00	0.14	**0.022**
	Score 0 + 2	*Fragaria virginiana*	0.90	0.41	**0.013**
		Rosa acicularis	1.00	0.24	**0.014**
		Vicia americana	0.89	0.38	**0.026**
Grazing Intensity	U	*Agrostis scabra*	1.00	0.25	**0.034**
		Antennaria rosea	1.00	0.25	**0.034**
		Artemisia ludoviciana	0.96	0.25	**0.020**
		Astragalus agrestis	0.83	0.25	**0.047**
		Danthonia intermedia	0.97	0.50	**0.001**
		Festuca hallii	0.96	0.25	**0.019**
		Geum triflorum	0.94	0.25	**0.019**
		Heuchera richardsonii	1.00	0.25	**0.034**
		Hierochloe odorata	1.00	0.25	**0.034**
		Juncus arcticus ssp. *Balticus*	0.83	0.25	**0.027**
		Juncus tenuis	1.00	0.25	**0.034**
		Penstemon procerus	1.00	0.25	**0.034**
		Potentilla gracilis	0.93	0.25	**0.020**
		Stellaria longipes	0.90	0.50	**0.005**
	L	*Melilotus alba*	1.00	0.22	**0.039**
	H	*Hordeum jubatum*	0.95	0.25	**0.046**
		Lepidium densiflorum	0.92	0.25	**0.050**
		Thalapsi arvense	0.85	0.75	**0.035**
	L+LM+M+MH+H	*Trifolium repens*	0.99	0.76	**0.030**

ISA ran in R using *indicspecies:multipatt* (Caceres and Legendre, 2009).
A = Probability of occurring, B = Fidelity
Permutations = 999

Appendix B.2 Rangeland Health Assesment

Appendix B.2.1. Summary tables for the NMDS ordinations of rangeland health assesment scores.

Table B.2.1.1. Summary of significant management factor centroids arising from the NMDS ordination of rangeland health assessment scores (Figure 4.5).

Management Factor	r^2	P Value	Centroid	NMDS 1	NMDS 2
Grazing Intensity	0.14	**0.002**	U	0.02	-0.04
			L	0.02	-0.02
			LM	0.02	0.02
			M	0.00	-0.01
			MH	0.00	0.00
			H	-0.09	0.01
Fire (Survey)	0.08	**0.001**	No Fire	-0.01	0.00
			Fire	0.05	-0.01
Harrowing	0.03	0.057	Not Harrowed	0.01	-0.01
			Harrowed	-0.01	0.01
Fed Hay (in pasture)	0.08	**0.007**	No Hay	0.02	0.01
			Fed Hay	-0.01	0.01
			Unknown	-0.02	-0.01

Table B.2.1.2. Summary of biplot scores from the NMDS ordination of various pasture soil and vegetation properties, including rangeland health assessment scores (Figure 4.5).

Factor		NMDS 1	NMDS 2	r^2	P Value
Soil Properties	OM	-0.98	0.20	0.02	0.375
	EC	-0.97	0.26	0.05	0.056
	pH	-0.99	0.10	0.04	0.109
	N	-0.98	0.21	0.02	0.274
	C	-0.98	0.22	0.02	0.339
	C:N Ratio	0.99	-0.14	0.04	0.135
	Sand	0.83	-0.56	0.03	0.240
	Clay	-0.97	0.22	0.03	0.274
	Silt	-0.61	0.79	0.02	0.383
Litter Depth	Depth	0.94	-0.35	0.07	**0.035**
Basal Cover	Vegetation	0.32	0.95	0.10	**0.008**
	Litter	0.56	-0.83	0.02	0.388
	Bare Soil	-0.67	-0.74	0.13	**0.001**
	Manure	-0.54	-0.84	0.06	**0.043**
	Rock	-0.61	0.80	0.02	0.403
	Lichen	0.29	-0.96	0.02	0.377
	Moss	-0.43	-0.90	0.03	0.277
	Wood	0.55	-0.83	0.03	0.179
Pasture Characteristics	Years Farmed	0.67	-0.74	0.03	0.394
	Pasture Age	0.79	-0.62	0.01	0.692
	RHA Score	0.94	-0.34	0.12	**0.001**
Similarity	Sorensen's	-0.63	0.78	0.01	0.626
Seed Bank	Shannon's Diversity	0.31	-0.95	0.02	0.367
	Simpson's Diversity	0.29	-0.96	0.01	0.494
	Pielou's Evenness	0.16	-0.99	0.00	0.800
	Richness	0.46	-0.89	0.00	0.935
	Abundance	-0.67	0.74	0.03	0.250
	Total Graminoids	-0.69	-0.72	0.00	0.805
	Total Broad Leaf	-0.53	0.85	0.03	0.185
	Total Native	-0.44	-0.90	0.04	0.139
	Total Introduced	-0.29	0.96	0.06	**0.046**
	Noxious Weeds	0.36	0.93	0.08	**0.019**
	Legumes	0.95	0.32	0.01	0.738
	Woody	0.99	0.10	0.02	0.434
	Native Ruderal Forbs	-0.80	-0.60	0.04	0.175
	Native Perennial Forbs	0.39	-0.92	0.01	0.491
	Introduced Ruderal Forbs	-0.45	0.89	0.04	0.104
	Seeded Graminoids	-0.23	0.97	0.00	0.801
	Native Grasses	0.67	-0.74	0.00	0.829
	Ruderal Grasses	-0.18	-0.98	0.09	**0.012**
	Graminoids	-0.60	-0.80	0.01	0.707
Plant Community	Shannon's Diversity	0.82	-0.57	0.02	0.430
	Simpson's Diversity	0.98	0.18	0.00	0.841
	Pielou's Evenness	-0.99	0.16	0.05	0.109
	Richness	0.85	-0.51	0.06	**0.046**
	Total Veg. Cover	0.47	0.88	0.09	**0.011**
	Total Graminoids	0.89	0.46	0.03	0.270
	Total Broad Leaf	-0.51	0.86	0.01	0.650
	Total Native	0.38	-0.93	0.11	**0.005**
	Total Introduced	-0.10	1.00	0.15	**0.002**
	Noxious Weeds	-0.13	0.99	0.30	**0.001**
	Legumes	0.99	0.16	0.01	0.784
	Woody	0.84	-0.54	0.11	**0.008**
	Native Ruderal Forbs	-0.45	-0.89	0.05	0.090
	Native Perennial Forbs	0.57	-0.82	0.04	0.142
	Introduced Ruderal Forbs	-0.82	0.57	0.15	**0.001**
	Seeded Graminoids	0.36	0.93	0.06	0.062
	Native Grasses	0.34	-0.94	0.08	**0.020**
	Ruderal Grasses	-1.00	0.03	0.09	**0.015**
	Graminoids	0.02	-1.00	0.05	0.097

Appendix B.3 Defining Plant Communities

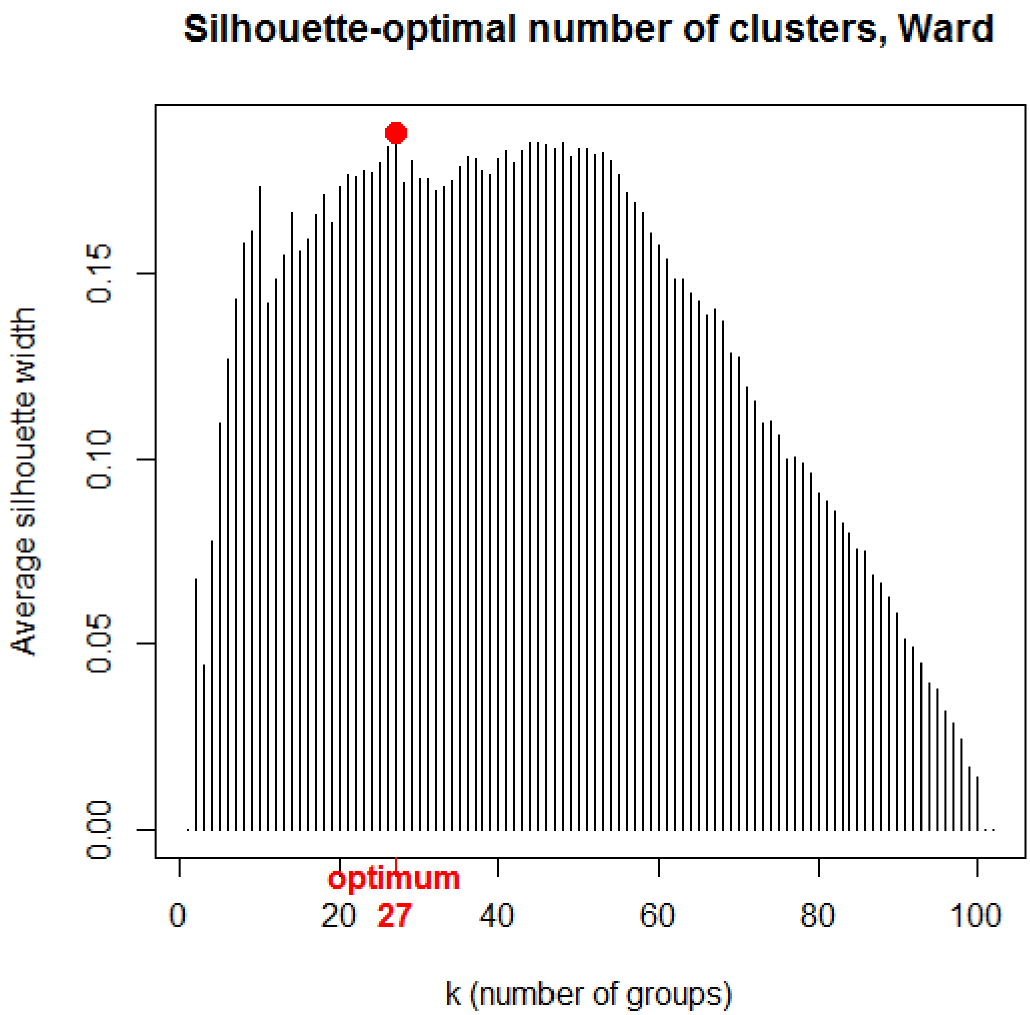


Figure B.3.1. Silhouette widths for plant community groups determine the relatedness of clusters when choosing the partition criteria (Borcard et al. 2011). Optimally we have 27 plant communities, but a meaningful peak of 10 groupings was chosen to simplify descriptions and reduce complexity for pastures in north central Alberta.

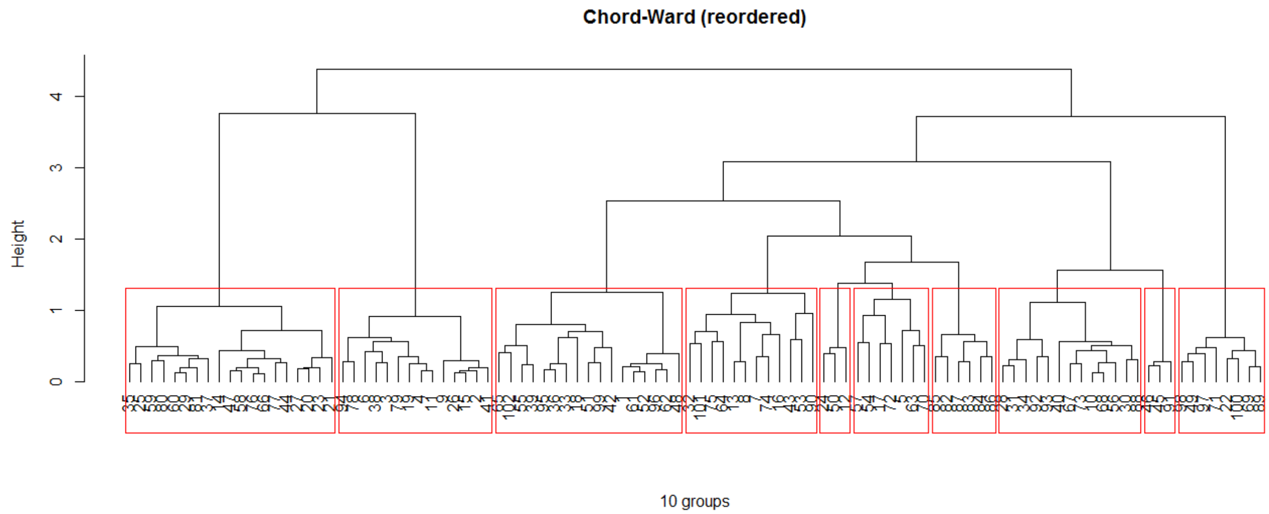


Figure B.3.2. Summary of final cluster analysis revealing 10 different communities comprised of groupings ranging from 3 to 19 sites for pastures in north central Alberta.

Table B.3.1 Indicator species analysis and dominant species cover for each of 10 plant communities identified through a cluster analysis.

Community Type	n	Indicator Species	A	B	P value	Dominant Species	Cover (%)
1	19	*Poa pratensis*	0.26	1.00	**0.001**	*Poa pratensis*	51.5
						Bromus inermis	13.6
						Trifolium repens	6.7
						Taraxacum officinale	6.6
2	14	*Bromus inermis*	0.33	1.00	**0.001**	*Bromus inermis*	42.4
						Poa pratensis	20.4
						Elytrigia repens	7.3
						Taraxacum officinale	6.5
3	17	*Trifolium repens*	0.46	0.94	**0.001**	*Poa pratensis*	24.6
						Trifolium repens	21.3
						Taraxacum officinale	12.6
						Bromus inermis	10.0
4	12	*Achillea millefolium*	0.45	0.75	**0.038**	*Poa pratensis*	23.9
		Androsace septentrionalis	0.60	0.50	**0.028**	*Agropyron cristatum*	6.3
		Antennaria parvifolia	0.96	0.42	**0.015**	*Bromus inermis*	5.2
		Artemisia frigida	0.93	0.50	**0.016**	*Elytrigia repens*	4.1
		Carex filifolia	1.00	0.50	**0.009**		
		Cerastium arvense	0.98	0.42	**0.012**		
		Geum triflorum	1.00	0.33	**0.007**		
		Heterotheca villosa	1.00	0.33	**0.011**		
		Solidago missourienses	0.85	0.33	**0.050**		
		Violoa adunca	0.88	0.33	**0.018**		
5	3	*Phalaris aurundinacea*	0.93	1.00	**0.001**	*Phalaris aurundinacea*	32.4
		Carex rostrata	0.90	0.67	**0.002**	*Bromus inermis*	17.8
		Lotus corniculatus	1.00	0.33	0.073	*Poa pratensis*	15.7
						Taraxacum officinale	11.0
6	7	*Medicago sativa*	0.42	0.86	**0.027**	*Dactylis glomerata*	15.1
		Dactylis glomerata	0.57	0.57	**0.044**	*Medicago sativa*	11.9
						Festuca rubra	10.8
						Bromus inermis	8.1
7	6	*Lathyrus ochroleucus*	0.98	0.83	**0.001**	*Bromus inermis*	23.2
		Fragaria virginiana	0.69	1.00	**0.001**	*Poa pratensis*	17.5
		Vicia americana	0.68	1.00	**0.002**	*Trifolium hybridum*	10.0
		Rosa acicularis	0.88	0.67	**0.001**	*Alopecurus pratensis*	6.9
		Trifolium hybridum	0.45	1.00	**0.001**		
		Alopecurus pratensis	0.54	0.67	**0.026**		
		Lathyrus venosus	1.00	0.33	**0.005**		
		Thalictrum venulosum	1.00	0.33	**0.004**		
		Phleum pratense	0.40	0.83	**0.026**		
		Trifolium pratense	0.40	0.83	**0.038**		
		Galium boreale	0.37	0.83	**0.046**		
8	13	*Taraxacum officinale*	0.21	1.00	**0.043**	*Poa pratensis*	26.5
						Elytrigia repens	23.9
						Taraxacum offficinale	16.6
						Trifolium repens	8.2
9	3	*Chenopodium album*	0.74	1.00	**0.001**	*Elytrigia repens*	57.6
		Elytrigia repens	0.52	1.00	**0.001**	*Poa pratensis*	8.8
						Lolium perenne	2.9
						Medicago sativa	2.8

10	8	*Bromus biebersteinii*	0.75	1.00	**0.001**	*Bromus biebersteinii*	42.6
		Polygonum convolvulus	0.45	0.50	**0.049**	*Taraxacum officinale*	10.0
						Medicago sativa	8.5
						Poa pratensis	8.0

n = Number of sites (pastures)
A = Probability of occurring, B = Fidelity
Permutations = 999

Table B.3.2. perMANOVA of plant community composition among natural subregions, plant community types, and defined communities through clustering.

RHA Category	Mean Square	F Model	R^2	P Value
Plant Community Type	0.70	3.24	0.03	**0.002**
Natural Subregion	0.29	1.34	0.01	0.191
Defined Plant Communities	2.52	12.82	0.11	**0.001**

Table B.3.3. Plant community indicator species for the natural regions in central Alberta.

Management	Category	Species	A	B	P value
Natural Region	*Boreal	*Alopecurus pratensis*	0.83	0.19	0.051
		Lathyrus ochroleucus	0.95	0.12	0.093
		Rosa acicularis	0.98	0.12	**0.019**
	Parkland	*Carex praegracilis*	1.00	0.06	0.100
		Capsella bursa-pastoris	0.99	0.14	**0.007**
		Polygonum aviculare	0.95	0.18	**0.011**

A = Probability of occurring, B = Fidelity
Permutations = 999
*Sites from two boreal natural subregions were combined

Appendix B.4. Characteristics of Plant Communities and Soil under Rangeland Health Parameters.

B.4.1. Results and Brief Discussion

The rangeland health assessment's categorical scores were useful predictors of plant community shifts, as well as the abundance and diversity of functional plant groups. As mentioned above in Chapter 4, plant community type (tame vs. modified-tame) significantly defined plant communities ($P = 0.002$; Table B.4.1), which reflected communities similar to those with a history of cultivation and non-cultivation, respectively. Native grasses and forbs were strong indicators of modified-tame communities (Table B.4.3). Richness and diversity were significantly higher in modified than tame communities ($P < 0.001$; Table B.4.8 and B.4.9), with modified communities having nearly two-fold more richness than tame grasslands. High richness and diversity in turn, corresponded with lower evenness ($P = 0.005$; Table B.4.8 and B.4.9). Tame pastures were more nutrient rich, having higher C, N, and OM and were also associated with greater soil compaction (Ps < 0.028; Table B.4.10 and B.4.11). Soils in modified-tame communities were associated with higher proportions of sand and lower amounts of clay and silt (Ps < 0.05; Table B.5.1 and B.5.2).

Relative forage cover score significantly defined plant communities ($P = 0.001$; Table B.4.1). Pastures receiving lower scores of 5 & 9 were similar in composition ($P = 0.075$; Table B.4.2), while pastures scoring the highest score, 12 with 90% relative forage cover, were significantly different from pastures receiving lower scores (Ps < 0.011; Table B.4.2). Native forbs and grasses were representative of pastures scoring 9, as this was the highest possible score for modified-tame pastures (Ps < 0.05; Table B.4.3). Pastures scoring 5 included disturbance-induced species like foxtail barley (*Hordeum jubatum*) and stinkweed (*Thalapsi arvense*) (Table B.1.2.2). The relationship between richness and diversity and forage cover scores was nonlinear, but unimodal in nature with richness and diversity peaking at lower scores ($P < 0.02$; Table B.4.8 and B.4.9). No pastures scored less than 5, which would have indicated less than 40% cover. Pastures that scored lower than the maximum score had significantly more bare ground

(P < 0.021; Table B.4.12 and B.4.13). Soil compaction also decreased with lower forage cover score (P = 0.002; B.4.10 and B.4.11), which is counter intuitive.

A decrease in the relative cover of tall productive forage species caused a significant shift in plant communities (P = 0.048; Table B.4.1); where pastures with ≥75% (relative) tall productive forage cover (RHA score = 14) differed from pastures with 40% to 74% cover (RHA score = 7) (P = 0.049; Table B.4.2). No significant (P < 0.05) indicator species emerged for those groups, but pastures with the lowest possible score (with less than 40% cover) included disturbance-induced ruderal forbs like lambsquarters (*Chenopodium album*), shepherds purse (*Capsella bursa-pastoris*), and flixweed (*Descurainia sophia*) (P < 0.003) (Table B.1.2.2). Richness induced by removal of more competitive forages corresponded with higher richness and diversity at lower scores (P < 0.05; B.4.8 and B.4.9). Not surprisingly, decreased litter cover was detected in pastures scoring lower (P = 0.001; Table B.4.12 and B.4.13). Hence, reductions in forage cover are consistent with declines in range health associated with high intensity use, reducing hydraulic function and increasing plant community richness by favouring greater cover of weedy ruderals.

Differences in weedy and disturbance-based plant species shifts were detected (P = 0.01; Table B.4.1); the lowest score (7) was assigned to pastures indicating 26% to 49% cover of the ruderal species was primarily associated with dandelion (*Taraxacum officinale*) (P = 0.001) cover, with additional cover of the disturbance-adapted grasses like quackgrass (*Elytrigia repens*) and foxtail barley (*Hordeum jubatum*) (P < 0.031; Table B.1.2.2).

Litter quantity, a measure of hydraulic function, also explained shifts in plant communities (P = 0.049; Table B.4.1). Pastures with sparse or absent litter (RHA score = 0) were associated primarily with introduced annuals (P < 0.05) and the disturbance-adapted grasses fowl bluegrass (*Poa palustris*) and shortawned foxtail (*Alopecurus aequalis*) (Table B.1.2.2). The legume *Medicago sativa* (P = 0.016) was also an indicator of low litter scores (Table B.1.2.2); no indicator species were detected for higher scoring pastures. Scores based on litter quantity had significant effects on plant basal cover with litter depth and cover responding positively to increasing scores (P < 0.001; B.4.11 and B.4.12), while bare ground and manure cover were lowest for higher scoring pastures (P < 0.041; B.4.11 and B.4.12). Basal vegetation cover had a non-linear response (P = 0.01), peaking at a lower intermediate score of 8/25, which is described as a thin litter layer, with 25% to 67% of the pasture having inadequate litter (Table B.4.13).

Soil erosion scores did not define distinct plant community composition (P = 0.253; Table B.4.1). The legumes cream peavine (*Lathyrus ochroleucus*) and American vetch (*Vicia americana*) were associated with stable (i.e. non-eroding) communities while introduced ruderals were associated with eroded pastures (P < 0.05; Table B.1.2.2). Pastures losing points for the presence of erosion were more saline (P = 0.014; Table B.4.10 and B.4.11), with higher basal cover from bare ground and manure (P < 0.01), and a thinner litter layer (P < 0.001; Table B.4.12 and B.4.13). Unlike erosion, scores for anthropogenic bare soil did not detect shifts in plant communities (P = 0.498; Table B.4.1).

Noxious weed cover and density scores were not associated with shifts in plant communities (Ps > 0.1; Table B.4.1); however, select noxious weed species were indicators for pastures with lower scores. Where absolute noxious weed cover was 1% to 15% (RHA score = 1/5), Canada thistle (*Cirsium arvense*) and common tansy (*Tanacetum vulgare*) were indicators (P < 0.023; B.1.2.2). When noxious weed density was high (RHA Score = 0), indicating a heavy infestation, Canada thistle was a strong indicator (P = 0.001). Ruderal introduced forbs were also indicative of higher noxious weed cover (Ps < 0.05; Table B.4.3). Scores for noxious weed cover and density did not reflect differences in soil properties or plant community diversity. Estimated basal vegetative cover increased with decreasing score for noxious cover and density, while basal litter cover and depth decreased (P < 0.05; Table B.4.12 and B.4.13).

Woody cover and density was not a strong indicator of shifts in plant communities, likely resulting from the avoidance of pastures with abundant brush during our survey. Woody cover had a near significant effect on plant communities (P = 0.093; Table B.4.1); indicator species of pastures with moderate cover (5% to 15%; RHA score = 3) tended to contain native plants from various taxa (grasses, sedges, forbs) ($P < 0.05$) (Table B.1.2.2) and may therefore reflect a decreased cultivation history. Native shrubs were indicators for woody species density distribution scores, with western snowberry (*Symphoricarpos occidentalis*) indicative of moderate infestations ($P = 0.026$) (RHA score = 2), and prickly rose (*Rosa acicularis*) heavier infestations ($P = 0.014$) (RHA scores 0 and 2) (Table B.1.2.2). Richness and diversity were higher when pastures scored lower for woody cover and density ($P < 0.05$; Table B.4.8 and B.4.9). Carbon to nitrogen ratios were higher for pastures with woody cover at moderate to high infestations ($P < 0.05$; Table B.4.10 and B.4.11), reflecting the accumulation of more recalcitrant woody material. Pastures with woody cover exceeding 5% were associated with sandier soil containing less clay ($P < 0.05$; Table B.5.1 and B.5.2), more basal vegetation cover ($P = 0.011$), and less litter cover ($P = 0.038$), and could reflect reduced cultivation on less productive ecosites (Table B.4.12 and B.4.13).

RHA categories describing overall health (i.e. healthy, healthy with problems, and unhealthy) were associated with plant community and soil responses. Total richness, bare ground, and manure cover increased with decreasing health, while compaction, litter cover and litter depth were highest in pastures defined as healthy ($Ps < 0.05$).

Many of these significant effects align with the expected plant community shifts we expect to observe in central Alberta's tame grasslands under heavy grazing pressure.

Table B.4.1. PerMANOVA of plant community composition responses to individual rangeland health metric categories in north central Alberta.

RHA Category	**Mean Square**	**F Model**	**R^2**	**P Value**
Plant Community Type	0.70	3.24	0.03	**0.002**
Forage Cover	0.52	2.44	0.05	**0.001**
Cover of Tall Productive Forages	0.35	1.61	0.03	**0.048**
Weedy & Ruderal Cover	0.58	2.68	0.03	**0.010**
Hydraulic Function & Litter	0.33	1.51	0.04	**0.049**
Soil Erosion	0.26	1.17	0.02	0.253
Anthropogenic Bare Soil	0.21	0.94	0.02	0.498
Noxious Weed Cover	0.25	1.12	0.02	0.316
Noxious Weed Density	0.27	1.25	0.04	0.176
Woody Spp Cover	0.36	1.63	0.02	0.093
Woody Spp Density	0.23	1.05	0.02	0.391
Grazing Intensity	0.28	1.27	0.06	0.116
Health	0.34	1.57	0.03	**0.048**

Distance = Bray-Curtis, Permutations = 999

Table B.4.2. PerMANOVA contrasts of management factors and RHA categories affecting plant community composition in north central Alberta.

Rangeland Health	**Scores**	**Mean Square**	**F Model**	**R^2**	**P Value**
Forage Cover	5 vs 9	0.35	1.67	0.05	0.075
	5 vs 12	0.59	2.74	0.04	**0.011**
	9 vs 12	0.53	2.50	0.03	**0.008**
Cover of Tall Productive Forages	0 vs 7	0.22	0.93	0.05	0.476
	0 vs 14	0.33	1.53	0.02	0.116
	7 vs 14	0.39	1.81	0.02	**0.049**
Hydraulic Function & Litter	0 vs 8	0.28	1.35	0.05	0.180
	0 vs 16	0.27	1.31	0.04	0.203
	0 vs 25	0.26	1.08	0.02	0.362
	8 vs 16	0.17	0.90	0.02	0.523
	8 vs 25	0.42	1.92	0.03	**0.047**
	16 vs 25	0.43	1.97	0.03	**0.041**
Health	Healthy vs Problems	0.32	1.50	0.02	0.129
	Healthy vs Unhealthy	0.38	1.65	0.02	0.078
	Problems vs Unhealthy	0.32	1.60	0.05	0.098

Distance = Bray-Curtis, Permutations = 999

Table B.4.3. Indicator species analysis of plant community functional group association with rangeland health metrics. Results with P < 0.1 are shown, significant results (P < 0.05) are bolded.

Rangeland Health	Category	Species	A	B	P value
Plant Community Type	Modified-Tame	Native Perennial Forbs	0.92	1.00	**0.001**
		Graminoids	0.97	0.83	**0.001**
		Native Perennial Grasses	0.83	0.75	**0.006**
		Native Ruderal Forbs	0.79	0.42	**0.026**
		Woody Species	0.85	0.33	**0.011**
Forage Cover	Score 12	Introduced Species	0.36	1.00	**0.007**
		Seeded (Introduced) Grasses	0.43	1.00	**0.001**
		Total Grasses + Graminoids	0.39	1.00	**0.001**
	Score 9	Graminoids	0.84	0.41	**0.027**
		Native Perennial Forbs	0.61	0.74	0.083
	Score 5	Introduced Ruderal Forbs	0.56	1.00	**0.001**
		Ruderal Grasses	0.66	0.44	**0.039**
		Total Broad Leaf Plants	0.41	1.00	0.076
Cover of Tall Productive Forages	Score 14	Seeded (Introduced) Grasses	0.44	1.00	**0.005**
		Total Grasses + Graminoids	0.41	1.00	**0.002**
	Score 7	Native Perennial Forbs	0.84	0.76	0.071
	Score 0	Ruderal Grasses	0.76	0.50	0.099
		Introduced Ruderal Forbs	0.58	1.00	**0.025**
Weedy & Ruderal Cover	Score 14	Seeded (Introduced) Grasses	0.56	1.00	**0.043**
		Total Grasses + Graminoids	0.56	1.00	**0.008**
	Score 7	Introduced Ruderal Forbs	0.75	1.00	**0.001**
		Noxious Weeds	0.71	0.70	0.079
		Ruderal Grasses	0.86	0.40	**0.046**
		Total Broad Leaf Plants	0.61	1.00	**0.030**
Hydraulic Function & Litter	Score 25	Total Grasses + Graminoids	0.28	1.00	0.059
	Score 0	Ruderal Grasses	0.64	0.50	**0.037**
Erosion	Score 10	Legumes	0.44	0.98	**0.034**
	Score 7	Graminoids	0.67	0.34	0.094
Anthropogenic Bare Soil	Score 5	Introduced Species	0.37	1.00	**0.022**
		Seeded (Introduced) Grasses	0.39	1.00	**0.037**
		Total Grasses + Graminoids	0.37	1.00	0.061
	Score 0 + 3	Native Ruderal Forbs	0.73	0.40	0.084
		Ruderal Grasses	0.61	0.60	**0.049**
Noxious Weed Cover	Score 5	Graminoids	0.84	0.41	**0.016**
		Woody Species	0.73	0.21	0.095
	Score 1	Noxious Weeds	0.90	0.94	**0.001**
Noxious Weed Density	Score 5	Graminoids	0.69	0.41	**0.021**
	Score 0	Introduced Species	0.27	1.00	**0.003**
		Noxious Weeds	0.80	0.88	**0.001**
Woody Cover	Score 3	Native Perennial Forbs	0.83	0.93	**0.001**
		Native Ruderal Forbs	0.67	0.50	**0.017**
		Woody Species	0.97	0.50	**0.001**
Wood Density	Score 0 + 2	Native Perennial Forbs	0.86	0.83	**0.003**
		Woody Species	0.98	0.31	**0.005**
Grazing Intensity	U + L + LM	Native Perennial Grasses	0.83	0.57	**0.043**
	U + L + LM + M + MH	Native Perennial Forbs	1.00	0.68	**0.027**

A = Probability of occurring, B = Fidelity
Permutations = 999

Table B.4.4. Significant effects of rangeland health metrics on the abundance of various primary vegetation cover groupings.

	Graminoids		**Broad Leaf**		**Native**		**Introduced**	
Rangeland Health	**F Value**	**P Value**	**F Value**	**P Value**	X^2	**P Value**	X^2	**P Value**
Plant Community Type	3.886	0.051	3.881	0.052	**18.562**	**<0.001**	**8.488**	**0.004**
Forage Cover	**14.127**	**<0.001**	**11.081**	**<0.001**	**13.126**	**0.001**	5.189	0.075
Cover of Tall Productive Forages	**11.555**	**<0.001**	**6.995**	**0.001**	3.196	0.202	4.090	0.129
Weedy & Ruderal Cover	**7.971**	**0.006**	**7.566**	**0.007**	0.099	0.753	0.003	0.955
Hydraulic Function & Litter	**4.716**	**0.004**	2.359	0.076	1.950	0.583	7.117	0.068
Soil Erosion	0.054	0.947	1.839	0.164	1.514	0.469	2.346	0.310
Anthropogenic Bare Soil	**3.587**	**0.031**	0.198	0.821	2.795	0.247	**7.570**	**0.023**
Noxious Weed Cover	0.883	0.417	0.972	0.382	3.786	0.151	5.654	0.059
Noxious Weed Density	1.156	0.331	1.246	0.297	3.885	0.274	**8.155**	**0.043**
Woody Spp Cover	1.417	0.237	1.855	0.176	**6.134**	**0.013**	3.307	0.069
Woody Spp Density	0.771	0.465	1.616	0.204	5.280	0.071	1.042	0.594
Grazing Intensity	2.255	0.055	0.810	0.545	10.263	0.068	1.719	0.886
Health	**13.905**	**<0.001**	**7.890**	**0.001**	0.378	0.828	2.531	0.282

Bold: $P < 0.05$, Black: $P < 0.1$, Grey: $P > 0.1$

Table B.4.5. Summary LS means (±SE) for all significant management effects on the cover of primary vegetation groups. Within a column and management factor, means with different letters differ, $P < 0.05$ after Bonferroni correction.

Rangeland Health	Score	Graminoids	Broadleaf	Native	Introduced
Plant Community Type	Modified-Tame	58.2 (±4.2)	34.0 (±4.3)	33.0 (±3.4) a	59.2 (±4.2) b
	Tame	67.1 (±1.5)	24.6 (±1.6)	5.2 (±1.2) b	86.5 (±1.5) a
Forage Cover	5	50.2 (±4.4) b	37.6 (±4.6) a	12.5 (±4.7) a	75.2 (±5.4)
	9	59.5 (±2.6) b	33.4 (±2.7) a	16.3 (±2.7) a	76.6 (±3.1)
	12	70.9 (±1.6) a	21.0 (±1.7) b	4.8 (±1.7) b	87.2 (±2.0)
Cover of Tall Productive Forages	0	45.3 (±9.6) b	36.5 (±10.1) ab		
	7	53.7 (±3.3) b	37.5 (±3.5) a		
	14	69.1 (±1.5) a	23.1 (±1.6) b		
Weedy & Ruderal Cover	7	53.9 (±4.6) b	38.7 (±4.6) a		
	14	67.4 (±1.5) a	24.3 (±1.5) b		
Hydraulic Function & Litter	0	46.8 (±7.1) b	33.2 (±7.5)		75.4 (±8.5)
	8	60.6 (±2.9) b	31.1 (±3.0)		83.8 (±3.5)
	16	68.0 (±2.6) ab	26.1 (±2.8)		85.4 (±3.1)
	25	69.5 (±2.1) a	22.0 (±2.2)		82.3 (±2.5)
Anthropogenic Bare Soil	0	51.4 (±6.5) b			64.5 (±7.3) b
	3	62.5 (±3.5) ab			80.9 (±4.0) ab
	5	67.7 (±1.6) a			85.0 (±1.8) a
Noxious Weed Cover	1				89.4 (±4.0)
	3				83.6 (±2.0)
	5				76.2 (±4.0)
Noxious Weed Density	0				89.9 (±2.8) a
	1				81.4 (±3.6) ab
	3				81.4 (±2.9) ab
	5				76.2 (±4.0) b
Woody Spp Cover	3			19.6 (±3.8) a	73.0 (±4.4)
	6			6.8 (±1.5) b	84.9 (±1.7)
Woody Spp Density	0			11.6 (±3.2)	
	2			13.0 (±5.2)	
	4			7.1 (±1.7)	
Grazing Intensity	U	68.8 (±7.2)		23.1 (±7.3)	
	L	61.0 (±4.8)		9.6 (±4.9)	
	LM	72.9 (±2.9)		11.1 (±3.0)	
	M	64.5 (±2.5)		8.1 (±2.5)	
	MH	66.5 (±3.0)		5.4 (±3.0)	
	H	55.5 (±5.1)		2.6 (±5.2)	
Health	Healthy	71.0 (±1.6) a	21.6 (±1.7) b		
	Problems	57.4 (±2.4) b	34.1 (±2.5) a		
	Unhealthy	50.7 (±6.6) b	30.7 (±7.1) ab		

Table B.4.6. Significant effects of rangeland health metrics on the cover of specific plant functional groups in north central Alberta.

	Native & Introduced				Introduced						Native									
	Legumes		Ruderal Grasses		*Noxious Weeds		Ruderal Forbs		Seeded Graminoids		Ruderal Forbs		Perennial Forbs		Perennial Grasses		Graminoids		Woody Spp.	
Rangeland Health	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**
Plant Community Type	0.061	0.805	0.594	0.441	2.012	0.156	1.715	0.193	**19.067**	**<0.001**	**3.867**	**0.049**	**18.863**	**<0.001**	**8.959**	**0.003**	**28.628**	**<0.001**	**7.451**	**0.006**
Forage Cover	0.593	0.554	3.432	0.180	1.189	0.552	**18.285**	**<0.001**	**21.147**	**<0.001**	3.058	0.217	**6.082**	**0.048**	**8.347**	**0.015**	**8.012**	**0.018**	0.716	0.699
Cover of Tall Productive Forages	0.385	0.682	3.305	0.192	0.595	0.743	**9.129**	**<0.001**	**13.363**	**<0.001**	4.158	0.125	2.539	0.281	3.382	0.184	1.940	0.379	1.031	0.597
Weedy & Ruderal Cover	0.055	0.815	2.404	0.121	1.252	0.263	**32.202**	**<0.001**	**4.663**	**0.033**	0.053	0.817	0.001	0.972	0.906	0.341	1.255	0.263	0.013	0.908
Hydraulic Function & Litter	1.053	0.373	4.696	0.195	2.076	0.557	2.303	0.082	2.550	0.060	2.173	0.537	2.022	0.568	2.691	0.442	1.517	0.678	0.896	0.826
Soil Erosion	**4.589**	**0.012**	3.892	0.143	0.094	0.954	0.062	0.940	0.770	0.466	0.813	0.666	2.426	0.297	0.182	0.913	4.234	0.120	0.502	0.778
Anthropogenic Bare Soil	0.617	0.542	**7.554**	**0.022**	1.397	0.497	0.131	0.877	**4.921**	**0.009**	4.074	0.130	0.406	0.816	4.962	0.084	5.270	0.072	1.256	0.534
Noxious Weed Cover	0.349	0.706	1.416	0.493	**41.282**	**<0.001**	1.692	0.189	2.051	0.134	0.829	0.768	0.602	0.740	1.598	0.450	**6.479**	**0.039**	0.560	0.756
Noxious Weed Density	1.308	0.276	1.480	0.687	**43.603**	**<0.001**	1.217	0.308	1.963	0.125	4.991	0.172	2.406	0.492	5.123	0.163	5.721	0.126	1.089	0.780
Woody Spp Cover	0.860	0.356	0.521	0.471	1.346	0.246	3.729	0.056	1.588	0.211	**6.859**	**0.009**	**12.104**	**0.001**	1.227	0.268	1.401	0.237	**27.245**	**<0.001**
Woody Spp Density	1.001	0.371	2.720	0.257	3.529	0.171	0.634	0.533	0.307	0.736	1.712	0.425	**9.244**	**0.010**	0.046	0.977	0.932	0.628	**18.087**	**<0.001**
Grazing Intensity	2.110	0.071	5.499	0.358	3.188	0.671	1.535	0.186	1.437	0.218	4.877	0.431	10.678	0.058	7.403	0.192	4.994	0.417	5.118	0.402
Health	1.716	0.185	**7.195**	**0.027**	2.205	0.332	**8.902**	**<0.001**	**9.721**	**<0.001**	5.035	0.081	4.321	0.115	2.123	0.346	0.095	0.954	1.614	0.446

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

*Note noxious weeds include 1 graminoid species

Table B.4.7. Summary of LS mean (±SE) cover values of various plant functional groups with significant responses to various management factors.

		Native & Introduced		Introduced			Native				
Rangeland Health	Score	Legumes	Ruderal Grasses	Noxious Weeds	Ruderal Forbs	Seeded Grasses	Ruderal Forbs	Perennial Forbs	Perennial Grasses	Graminoids	Woody Spp.
Plant Community Type	Modified-Tame					41.9 (±4.8) b	0.29 (±0.09) a	14.2 (±1.5) a	11.4 (±2.4) a	4.5 (±0.7) a	0.7 (±0.2) a
	Tame					64.0 (±1.7) a	0.08 (±0.03) b	1.2 (±0.6) b	2.3 (±0.9) b	0.2 (±0.2) b	0.1 (±0.1) b
Forage Cover	5				23.8 (±2.1) a	41.4 (±5.0) b		2.9 (±2.1) ab	6.2 (±2.9) a	0.2 (±0.9) ab	
	9				11.7 (±1.2) b	51.1 (±2.9) b		6.5 (±1.2) a	5.6 (±1.7) ab	2.0 (±0.5) a	
	12				6.7 (±0.8) c	68.3 (±1.9) a		1.2 (±0.8) b	2.1 (±1.1) b	0.2 (±0.3) b	
Cover of Tall Productive Forages	0				30.8 (±4.9) a	39.7 (±11.3) b					
	7				14.6 (±1.7) b	44.8 (±3.9) ab					
	14				8.0 (±0.8) c	65.3 (±1.8) a					
Weedy & Ruderal Cover	7				23.4 (±2.0) a	50.0 (±5.6) b					
	14				8.0 (±0.7) b	62.6 (±1.8) a					
Hydraulic Function & Litter	0				14.8 (±3.9)	43.0 (±8.7)					
	8				12.2 (±1.6)	56.7 (±3.6)					
	16				8.9 (±1.4)	63.4 (±3.2)					
	25				8.0 (±1.2)	64.2 (±2.6)					
Soil Erosion	4	10.5 (±3.0) ab							1.3 (±2.3)		
	7	9.5 (±1.8) b							4.5 (±1.4)		
	10	15.8 (±1.7) a							3.0 (±1.3)		
Anthropogenic Bare Soil	0		3.0 (±0.8) a			38.1(±7.7) b				0.8 (±1.2)	
	3		1.6 (±0.5) ab			60.6 (±4.2) a				0.1 (±0.7)	
	5		0.3 (±0.2) b			63.0 (±1.9) a				0.8 (±0.3)	
Noxious Weed Cover	1			2.9 (±0.2) a						0.0 (±0.6) b	
	3			0.3 (±0.1) b						0.4 (±0.3) ab	
	5			0.0 (±0.2) c						2.2 (±0.6) a	
Noxious Weed Density	0			1.8 (±0.2) a							
	1			0.3 (±0.3) b							
	3			0.2 (±0.2) b							
	5			0.0 (±0.3) c							
Woody Spp Cover	3				6.2(±2.1)		0.19 (±0.08) a	8.8 (±1.7) a			1.1 (±0.2) a
	6				10.0 (±0.8)		0.09 (±0.03) b	1.8 (±0.7) b			0.0 (±0.1) b
Woody Spp Density	0							4.3 (±1.4) a			0.5 (±0.1) a
	2							6.8 (±2.3) a			0.8 (±0.2) ab
	4							1.9 (±0.8) b			0.0 (±0.1) b
Grazing Intensity	U	6.9 (±5.8)						8.6 (±3.4)			
	L	20.4 (±3.8)						0.9 (±2.2)			
	LM	8.5 (±2.4)						2.9 (±1.4)			
	M	15.4 (±2.0)						3.1 (±1.1)			
	MH	9.6 (±2.4)						2.6 (±1.4)			
	H	13.8 (±4.1)						8.6 (±3.4)			
Health	Healthy		0.3 (±0.2) b		7.4 (±0.9) c	65.8 (±2.0) a	0.04 (±0.04)				
	Healthy with Problems		1.1 (±0.3) ab		12.3 (±1.3) b	55.2 (±3.0) b	0.19 (±0.05)				
	Unhealthy		3.4 (±0.9) a		23.2 (±3.6) a	34.9 (±8.2) c	0.47 (±0.15)				

Table B.4.8. Significant effects of rangeland health on plant richness, diversity, and evenness within parkland pastures of north central Alberta.

	Richness		Shannon's Diversity		Simpson's Diversity		Pielou's Evenness	
Rangeland Health	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Plant Community Type	**18.206**	**0.000**	**25.777**	**0.000**	**12.315**	**0.001**	**8.141**	**0.005**
Forage Cover	**17.754**	**0.000**	**15.345**	**0.000**	**15.903**	**0.000**	0.116	0.890
Cover of Tall Productive Forages	**6.343**	**0.042**	**4.050**	**0.020**	**4.249**	**0.017**	0.038	0.963
Weedy & Ruderal Cover	0.307	0.580	1.000	0.320	2.056	0.155	1.434	0.234
Hydraulic Function & Litter	3.311	0.346	0.585	0.627	0.890	0.449	0.027	0.994
Soil Erosion	2.204	0.332	1.326	0.270	0.718	0.490	0.935	0.396
Anthropogenic Bare Soil	3.927	0.140	0.379	0.685	0.172	0.843	1.423	0.246
Noxious Weed Cover	1.553	0.460	0.059	0.943	0.178	0.837	0.461	0.632
Noxious Weed Density	4.972	0.174	0.629	0.598	0.665	0.576	0.496	0.686
Woody Spp Cover	**14.105**	**0.000**	**10.156**	**0.002**	**6.074**	**0.015**	**8.038**	**0.006**
Woody Spp Density	**14.855**	**0.001**	**3.695**	**0.028**	3.041	0.052	2.801	0.066
Grazing Intensity	1.487	0.915	0.298	0.913	0.476	0.793	0.446	0.815
Health	**7.033**	**0.030**	2.209	0.115	2.629	0.077	0.170	0.844

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1

Table B.4.9. Summary LS mean (±SE) values of plant richness, diversity, and evenness, for pastures sampled in relation to the management factors.

Rangeland Health	Score	Richness	Shannon's Diversity	Simpson's Diversity	Pielou's Evenness
Plant Community Type	Modified-Tame	23.3 (±1.3) a	2.20 (±0.11) a	0.82 (±0.04) a	0.101 (±0.008) b
	Tame	13.1 (±0.5) b	1.59 (±0.04) b	0.70 (±0.01) b	0.125 (±0.003) a
Forage Cover	5	16.3 (±1.7) a	1.88 (±0.13) a	0.78 (±0.04) a	
	9	17.8 (±1.0) a	1.97 (±0.07) a	0.80 (±0.02) a	
	12	12.6 (±0.6) b	1.50 (±0.05) b	0.67 (±0.01) b	
Cover of Tall Productive Forages	0	17.5 (±3.9) a	2.03 (±0.30) ab	0.82 (±0.09) a	
	7	17.1 (±1.3) a	1.89 (±0.10) a	0.78 (±0.03) a	
	14	13.7 (±0.6) b	1.60 (±0.05) b	0.70 (±0.01) b	
Woody Spp. Cover	3	20.4 (±1.4) a	1.99 (±0.11) a	0.78 (±0.03) a	0.102 (±0.007) b
	6	13.3 (±0.5) b	1.61 (±0.04) b	0.70 (±0.01) b	0.126 (±0.003) a
Woody Spp. Density	0	17.5 (±1.2) a	1.83 (±0.09) a	0.75 (±0.03)	0.107 (±0.006)
	2	16.5 (±1.9) ab	1.87 (±0.15) ab	0.78 (±0.04)	0.125 (±0.010)
	4	13.2 (±0.6) b	1.59 (±0.05) b	0.70 (±0.01)	0.126 (±0.003)
Health	Healthy	13.7 (±0.7) b		0.51 (±0.02)	
	Healthy with Problems	15.3 (±1.0) a		0.56 (±0.03)	
	Unhealthy	18.0 (±2.8) a		0.66 (±0.08)	

Table B.4.10. Significant effects of rangeland health metrics on various soil properties found across parkland pastures in north central Alberta.

Rangeland Health	**C (%)**		**N (%)**		**C:N**		**OM (%)**		**pH**		**EC (µS/cm)**		**Compaction (kg/cm^2)**	
	X^2	**P Value**	**X^2**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Plant Community Type	**5.413**	**0.020**	**4.803**	**0.028**	0.071	0.790	**7.856**	**0.006**	0.043	0.837	3.013	0.086	**6.867**	**0.012**
Forage Cover	2.593	0.273	2.683	0.261	1.177	0.312	1.364	0.260	1.713	0.186	2.811	0.065	**13.67**	**0.002**
Cover of Tall Productive Forages	3.636	0.162	4.222	0.121	0.475	0.623	2.221	0.114	0.650	0.524	1.255	0.290	3.885	0.055
Weedy & Ruderal Cover	0.010	0.919	0.099	0.753	0.049	0.826	0.048	0.827	1.112	0.294	2.017	0.159	2.331	0.134
Hydraulic Function & Litter	2.954	0.399	3.095	0.377	1.301	0.278	0.800	0.497	0.313	0.816	0.176	0.912	1.606	0.212
Soil Erosion	2.851	0.240	2.069	0.355	2.414	0.095	0.984	0.377	1.672	0.193	**4.429**	**0.014**	0.076	0.784
Anthropogenic Bare Soil	2.540	0.281	3.073	0.215	0.882	0.417	0.314	0.732	0.191	0.827	1.614	0.204	3.073	0.087
Noxious Weed Cover	2.709	0.258	1.822	0.402	0.568	0.569	0.588	0.558	1.073	0.346	0.047	0.954	1.169	0.286
Noxious Weed Density	2.765	0.429	1.723	0.632	0.288	0.834	0.594	0.621	0.576	0.632	0.034	0.992	0.990	0.325
Woody Spp Cover	0.946	0.331	2.272	0.132	**5.683**	**0.019**	1.549	0.216	1.117	0.293	3.161	0.078	0.405	0.528
Woody Spp Density	4.670	0.097	5.782	0.056	**3.148**	**0.047**	2.092	0.129	2.932	0.058	3.447	0.066	1.765	0.191
Grazing Intensity	10.466	0.063	10.663	0.058	1.298	0.271	1.664	0.151	0.981	0.434	**2.378**	**0.044**	2.371	0.068
Health	1.028	0.362	0.723	0.697	1.522	0.223	0.907	0.407	1.737	0.181	0.810	0.448	**3.783**	**0.031**

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1

Table B.4.11. Effect of significant rangeland health metric on the LS means (±SE) of various soil properties as sampled across parkland pastures of north central Alberta.

Management Factor	Treatment	C (%)	N (%)	C:N	OM (%)	pH	EC (µS/cm)	Compaction (kg/cm^2)
Rangeland Health	**Score**							
Plant Community Type	Modified-Tame	3.0 (±1.1) b	0.26 (±0.09) b		5.3 (±1.6) b		336.5 (±140.9)	1.2 (±0.4) b
	Tame	5.0 (±0.4) a	0.42 (±0.03) a		8.2 (±0.6) a		487.8 (±51.4)	2.1 (±0.1) a
Forage Cover	5						611.5 (±163.6)	1.0 (±0.3) c
	9						437.2 (±94.5)	1.8 (±0.2) ab
	12						464.2 (±60.4)	2.2 (±0.1) a
Soil Erosion	4			11.5 (±0.5)			544.5 (±120.3) ab	
	7			12.7 (±0.3)			636.1 (±72.8) a	
	10			12.2 (±0.3)			297.8 (±68.7) b	
Anthropogenic Bare Soil	0							1.2 (±0.5)
	3							2.0 (±0.3)
	5							2.0 (±0.1)
Woody Spp Cover	3			13.3 (±0.5) a			299.6 (±129.8)	
	6			12.1 (±0.2) b			497.2 (±51.8)	
Woody Spp Density	0	3.9 (±0.8)	0.30 (±0.07)	13.1 (±0.4) a		6.0 (±0.1)	287.1 (±104.7)	
	2	3.5 (±1.3)	0.28 (±0.11)	12.7 (±0.6) ab		5.9 (±0.2)	310.1 (±169.7)	
	4	5.2 (±0.4)	0.44 (±0.4)	12.0 (±0.2) b		6.2 (±0.1)	540.2 (±56.2)	
Grazing Intensity	U	3.6 (±1.9)	0.33 (±0.16)				304.7 (±238.6) ab	
	L	2.9 (±1.3)	0.23 (±0.11)				287.1 (±159.1) b	
	LM	5.3 (±0.8)	0.43 (±0.07)				575.5 (±97.4) ab	
	M	4.6 (±0.7)	0.37 (±0.06)				350.5 (±81.8) b	
	MH	5.0 (±0.8)	0.46 (±0.07)				514.3 (±99.5) ab	
	H	6.2 (±1.3)	0.48 (±0.12)				822.9 (±168.7) a	
Health	Healthy							2.07 (±0.14) a
	Problems							2.03 (±0.19) a
	Unhealthy							1.10 (±0.43) b

Table B.4.12. Summary of significant effects of rangeland health metric on various ground cover characteristics in parkland pastures of north central Alberta.

	Basal Veg Cover (%)		**Litter Cover (%)**		**Litter Depth (cm)**		**Bare Ground (%)**		**Manure Cover (%)**	
Rangeland Health	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**X^2**	**P Value**	**X^2**	**P Value**
Plant Community Type	0.676	0.413	0.447	0.505	0.037	0.847	0.780	0.377	0.162	0.688
Forage Cover	0.532	0.589	1.741	0.190	2.339	0.102	**7.763**	**0.021**	2.631	0.268
Cover of Tall Productive Forages	2.675	0.074	**11.139**	**0.001**	2.440	0.092	0.407	0.816	2.849	0.241
Weedy & Ruderal Cover	0.352	0.554	1.216	0.273	0.202	0.654	0.002	0.964	0.831	0.362
Hydraulic Function & Litter	**3.964**	**0.010**	**55.119**	**0.000**	**38.670**	**<0.001**	**24.103**	**<0.001**	**8.246**	**0.041**
Soil Erosion	1.967	0.145	0.032	0.859	**10.629**	**<0.001**	**9.176**	**0.010**	**10.492**	**0.005**
Anthropogenic Bare Soil	0.735	0.482	**17.409**	**0.000**	**13.626**	**<0.001**	**18.455**	**<0.001**	**13.883**	**0.001**
Noxious Weed Cover	**5.368**	**0.006**	**5.561**	**0.020**	0.694	0.502	0.036	0.982	2.710	0.258
Noxious Weed Density	**3.403**	**0.021**	**8.239**	**0.005**	**3.900**	**0.011**	4.027	0.259	5.557	0.135
Woody Spp. Cover	**6.793**	**0.011**	**4.408**	**0.038**	0.111	0.740	0.192	0.662	0.567	0.452
Woody Spp. Density	2.986	0.055	2.848	0.095	1.757	0.178	0.471	0.790	4.538	0.103
Grazing Intensity	0.816	0.541	**3.439**	**0.007**	**9.552**	**<0.001**	6.894	0.229	**19.942**	**0.001**
Health	1.407	0.250	**18.831**	**0.000**	**12.865**	**<0.001**	**15.036**	**0.001**	**6.740**	**0.034**

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

Note: Only trace amounts of rock, moss, and lichen were recorded.

Table B.4.13. Effect of significant rangeland health metric on the LS means (±SE) on ground cover variables sampled across parkland pastures of north central Alberta.

Rangeland Health	Treatment	Basal Veg Cover (%)	Litter Cover (%)	Litter Depth (cm)	Bare Soil (%)	Manure Cover (%)
Forage Cover	5				15.6 (±3.8) a	
	9				12.0 (±2.2) a	
	12				8.1 (±1.4) b	
Cover of Tall Productive Forages	0	55.5 (±11.2)	21.4 (±11.1) b	0.3 (±0.9)		
	7	48.2 (±3.8)	40.7 (±3.8) ab	1.2 (±0.3)		
	14	38.7 (±1.7)	50.7 (±1.7) a	1.3 (±0.1)		
Hydraulic Function & Litter	0	27.4 (±7.7) ab	27.7 (±6.6) b	0.2 (±0.5) c	43.5 (±4.3) a	1.8 (±0.7) a
	8	49.6 (±3.1) a	33.7 (±2.7) b	0.6 (±0.2) b	15.0 (±1.8) ab	1.0 (±0.3) ab
	16	40.1 (±2.9) ab	50.1 (±2.5) a	0.7 (±0.2) b	8.8 (±1.6) b	0.9 (±0.3) b
	25	37.2 (±2.3) b	57.1 (±2.0) a	2.1 (±0.1) a	4.7 (±1.3) c	0.9 (±0.2) b
Soil Erosion	4			0.9 (±0.3) b	13.9 (±2.9) a	1.0 (±0.4) ab
	7			0.9 (±0.2) b	12.8 (±1.7) a	1.2 (±0.2) a
	10			1.8 (±0.2) a	5.9 (±1.6) b	0.7 (±0.2) b
Anthropogenic Bare Soil	0		24.4 (±6.9) b	0.3 (±0.5) b	40.9 (±3.9) a	2.0 (±0.6) a
	3		42.5 (±3.7) ab	0.7 (±0.3) b	15.6 (±2.1) a	1.3 (±0.7) a
	5		51.2 (±1.7) a	1.5 (±0.1) a	6.6 (±1.0) b	0.8 (±0.2) b
Noxious Weed Cover	1	47.2 (±3.8) a	43.3 (±3.9) a			
	3	41.5 (±1.9) a	47.8 (±2.0) ab			
	5	30.4 (±3.8) b	56.4 (±3.9) b			
Noxious Weed Density	0	44.0 (±2.7) a	45.1 (±2.8) a	1.2 (±0.2) a		
	1	43.2 (±3.4) a	42.8 (±3.5) ab	0.7 (±0.3) b		
	3	40.7 (±2.8) ab	51.5 (±3.9) ab	1.6 (±0.2) a		
	5	30.4 (±3.8) b	56.4 (±3.9) b	1.6 (±0.3) a		
Woody Spp Cover	3	49.6 (±4.2) a	40.0 (±4.3) b			
	6	39.1 (±1.7) b	49.8 (±1.7) a			
Woody Spp Density	0	47.0 (±3.5)	43.3 (±3.6)			
	2	41.5 (±5.7)	47.3 (±5.8)			
	4	38.6 (±1.9)	50.1 (±1.9)			
Grazing Intensity	U		67.1 (±7.8) a	3.8 (±0.5) a		2.5 (±0.7) bc
	L		48.7 (±5.2) ab	2.0 (±0.3) a		0.9 (±0.5) abc
	LM		55.1 (±3.2) a	2.0 (±0.2) a		0.5 (±0.3) c
	M		45.8 (±2.7) ab	0.9 (±0.2) b		0.5 (±0.2) bc
	MH		46.7 (±3.2) ab	0.7 (±0.2) b		1.5 (±0.3) ab
	H		35.2 (±5.5) b	0.8 (±0.4) b		1.8 (±0.5) a
Health	Healthy		54.4 (±1.7) a	1.6 (±0.1) a	6.1 (±1.3) b	0.9 (±0.2) b
	Problems		38.5 (±2.5) b	0.8 (±0.2) b	15.7 (±1.8) a	0.9 (±0.3) ab
	Unhealthy		25.3 (±7.1) b	0.4 (±0.6) b	26.6 (±5.1) a	2.1 (±0.7) a

Table B.4.14. Significant ANOVA effects on total RHA score of north central Alberta pastures.

Rangeland Health	**F Value**	**P Value**
Plant Community Type	1.881	0.173
Forage Cover	**16.592**	**0.000**
Cover of Tall Productive Forages	**25.539**	**0.000**
Weedy & Ruderal Cover	**22.977**	**0.000**
Hydraulic Function & Litter	**70.830**	**0.000**
Soil Erosion	**11.273**	**0.000**
Anthropogenic Bare Soil	**19.314**	**0.000**
Noxious Weed Cover	**7.849**	**0.000**
Noxious Weed Density	**7.104**	**0.000**
Woody Spp. Cover	2.816	0.096
Woody Spp. Density	0.709	0.495
Grazing Intensity	**7.281**	**0.000**

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1

*Analysis includes 58 sites from the 2013 survey

Table B.4.15. Summary of LS means (±SE) for the total RHA scores for various management factors ($P < 0.05$) in north central Alberta pastures.

Rangeland Health	**Score**	RHA Score
Forage Cover	5	56.3 (±3.4) c
	9	77.5 (±2.1) b
	12	84.0 (±1.4) a
Cover of Tall Productive Forages	0	37.5 (±7.0) b
	7	65.1 (±2.4) b
	14	83.9 (±1.1) a
Weedy & Ruderal Cover	7	59.4 (±3.7) b
	14	82.1 (±1.2) a
Hydraulic Function & Litter	0	56.0 (±4.0) c
	8	65.7 (±1.6) c
	16	78.4 (±1.5) b
	25	90.4 (±1.2) a
Soil Erosion	4	72.8 (±3.2) b
	7	75.7 (±1.9) b
	10	85.8 (±1.8) a
Anthropogenic Bare Soil	0	56 (±5.0) b
	3	68.9 (±2.7) b
	5	83.7 (±1.3) a
Noxious Weed Cover	1	73.6 (±3.1) b
	3	79.2 (±1.6) b
	5	88.4 (±3.1) a
Noxious Weed Density	0	77.4 (±2.2) bc
	1	72.6 (±2.8) c
	3	82.7 (±2.3) ab
	5	88.4 (±3.1) a
Woody Spp. Cover	3	74.6 (±3.6)
	6	80.6 (±1.4)
Health	Unhealthy	43.3 (±3.1) c
	Problems	67.2 (±1.1) b
	Healthy	87.9 (±0.8) a

Table B.4.16. Coefficients of a multiple stepwise regression of total rangeland health scores against various plant community cover characteristics for north central Alberta pastures.

Predictor	**B-Estimate**	**SE**	**t**	**P Value**
Intercept	60.48	10.54	5.74	**<0.001**
Total Foliar Cover	-0.66	0.14	-4.79	**<0.001**
Total Grass Cover	0.76	0.13	5.75	**<0.001**
Legume Cover	0.67	0.11	3.02	**<0.001**
Seeded (Introduced) Grass Cover	-0.18	0.09	-2.02	**0.047**
Ruderal Grass Cover	-1.08	0.46	-2.36	**0.021**
Basal Vegetation Cover	0.22	0.09	2.53	**0.013**
Litter Cover	0.45	0.09	4.90	**<0.001**
Manure Cover	-1.18	0.60	-1.97	0.052
Woody Debris Cover	-11.52	4.09	-2.81	**0.006**
Litter Depth (cm)	2.52	0.75	3.33	**0.001**

$R^2 = 0.6759$, Adjusted $R^2 = 0.6402$, df = 91, F = 18.97, **P < 0.001**.
Terms selected using both forwards and backwards selection.

Table B.4.17. Coefficients for a multiple linear regression of total rangeland health score explained by plant community cover characteristics and soil properties for north central Alberta pastures.

Predictor	**Estimate**	**SE**	**t**	**P Value**
Intercept	90.00	15.45	5.86	**<0.001**
Total Foliar Cover	-3.58	1.65	-2.17	**0.033**
Total Grass Cover	3.52	1.63	2.16	0.337
Noxious Weed Cover	2.78	1.71	1.63	0.107
Introduced Ruderal Forb Cover	2.77	1.62	1.71	0.090
Legume Cover	3.53	1.64	2.16	**0.034**
Native Perennial Forb Cover	2.99	1.68	1.77	0.080
Ruderal Grass Cover	-0.69	0.43	-1.60	0.114
Basal Vegetation Cover	0.20	0.09	2.20	**0.031**
Litter Cover	0.42	0.10	4.38	**<0.001**
Manure Cover	-1.42	0.62	-2.31	**0.024**
Lichen Cover	15.55	10.05	1.55	0.126
Moss Cover	1.85	1.42	1.30	0.196
Wood Debris Cover	-8.09	3.83	-2.11	**0.038**
OM	-2.17	0.89	-2.43	**0.017**
pH	-4.97	1.75	-2.84	**0.006**
C	3.37	1.33	2.52	**0.014**
Litter Depth	2.84	0.76	3.74	**0.003**

$R^2 = 0.7241$, Adjusted $R^2 = 0.6642$, df = 83, F = 12.1, **P < 0.001**.
Terms selected using both forwards and backwards selection.

Appendix B.5. Soil Texture.

Table B.5.1. Results of the ANOVA analysis assessing the impact of management factors on observed levels of soil texture across 102 pastures in north central Alberta.

	Sand (%)		Silt (%)		Clay (%)	
Management	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Owned or Rented	0.002	0.965	2.531	0.115	1.052	0.308
Previous Cultivation	1.310	0.275	0.240	0.787	**3.673**	**0.029**
Grazing System	1.271	0.285	0.433	0.650	2.084	0.130
Timing of Grazing	0.404	0.751	0.348	0.791	0.491	0.689
System x Timing	0.836	0.506	0.342	0.849	1.113	0.355
Herbivore Type(s)	0.359	0.837	0.415	0.798	0.382	0.821
Herbicide	2.034	0.157	**5.194**	**0.025**	0.163	0.687
Fertilized	0.714	0.400	0.074	0.786	0.909	0.343
Manure Spreading	0.041	0.840	0.003	0.954	0.042	0.838
Harrowed	**4.727**	**0.032**	**4.398**	**0.039**	**4.644**	**0.034**
Aeration	0.332	0.566	0.068	0.795	3.127	0.080
Swathed or Mowed	0.054	0.818	0.203	0.654	0.289	0.592
*Fed Hay in Pasture Sampled	0.071	0.791	0.129	0.721	0.247	0.622
Burrowing Mammals	0.052	0.820	0.562	0.455	1.216	0.273
Fire (Survey)	0.800	0.373	0.042	0.839	2.392	0.125
Fire (Charcoal in Soil)	0.009	0.924	1.303	0.256	0.497	0.483
Rangeland Health						
Plant Community Type	**4.401**	**0.038**	3.346	0.070	**5.961**	**0.016**
Forage Cover	0.331	0.719	0.485	0.617	0.260	0.772
Cover of Tall Productive Forages	0.646	0.526	1.069	0.347	0.281	0.756
Weedy & Ruderal Cover	0.089	0.766	0.002	0.962	0.517	0.474
Hydraulic Function & Litter	0.558	0.644	0.325	0.807	1.024	0.386
Soil Erosion	2.273	0.108	2.027	0.137	0.684	0.507
Anthropogenic Bare Soil	1.026	0.362	0.013	0.987	2.467	0.090
Noxious Weed Cover	0.073	0.929	0.100	0.905	0.120	0.887
Noxious Weed Density	0.544	0.654	0.348	0.790	0.554	0.647
Woody Spp. Cover	**4.390**	**0.039**	3.508	0.064	**4.487**	**0.037**
Woody Spp. Density	0.229	0.796	0.386	0.681	0.617	0.542
Grazing Intensity	1.642	0.156	0.875	0.501	1.741	0.133
Health	0.281	0.756	0.134	0.875	0.540	0.585

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

*Includes only 58 sites from the 2013 survey

Table B.5.2. Summary of LS mean (±SE) soil texture values in relative to various management factors for north central Alberta Pastures.

Management	**Treatment**	**Sand (%)**	**Silt(%)**	**Clay(%)**
Cultivation	Cultivated			37.9 (±1.3) ab
	Never Cultivated			31.9 (±4.1) b
	Unknown			44.3 (±2.8) a
Herbicide	Sprayed in Last 3 Years	38.2 (±3.5) a		
	Not Sprayed Recently	29.5 (±1.5) b		
Harrowed	Harrowed	23.1 (±3.6) b	35.0 (±2.4) a	41.9 (±2.0) a
	Not Harrowed	34.4 (±2.6) a	28.8 (±1.7) b	36.8 (±1.4) b
Aeration	Aerated			49.2 (±5.8)
	Not Aerated			38.1 (±1.2)
Rangeland Health	**Score**			
Plant Community Type	Modified-Tame	44.4 (±6.1) a	23.9 (±4.1)	31.7 (±3.3) b
	Tame	28.8 (±2.2) b	31.8 (±1.5)	39.4 (±1.2) a
Anthropogenic Bare Soil	0			41.6 (±5.2)
	3			44.2 (±2.8)
	5			37.1 (±1.3)
Woody Spp Cover	3	42.7 (±5.7) a	24.3 (±3.4)	33.0 (±3.1) b
	6	28.7 (±2.3) b	31.9 (±1.5)	39.4 (±1.2) a

Appendix C: Chapter 5.

Appendix C.1 Seed Bank Heterogeneity

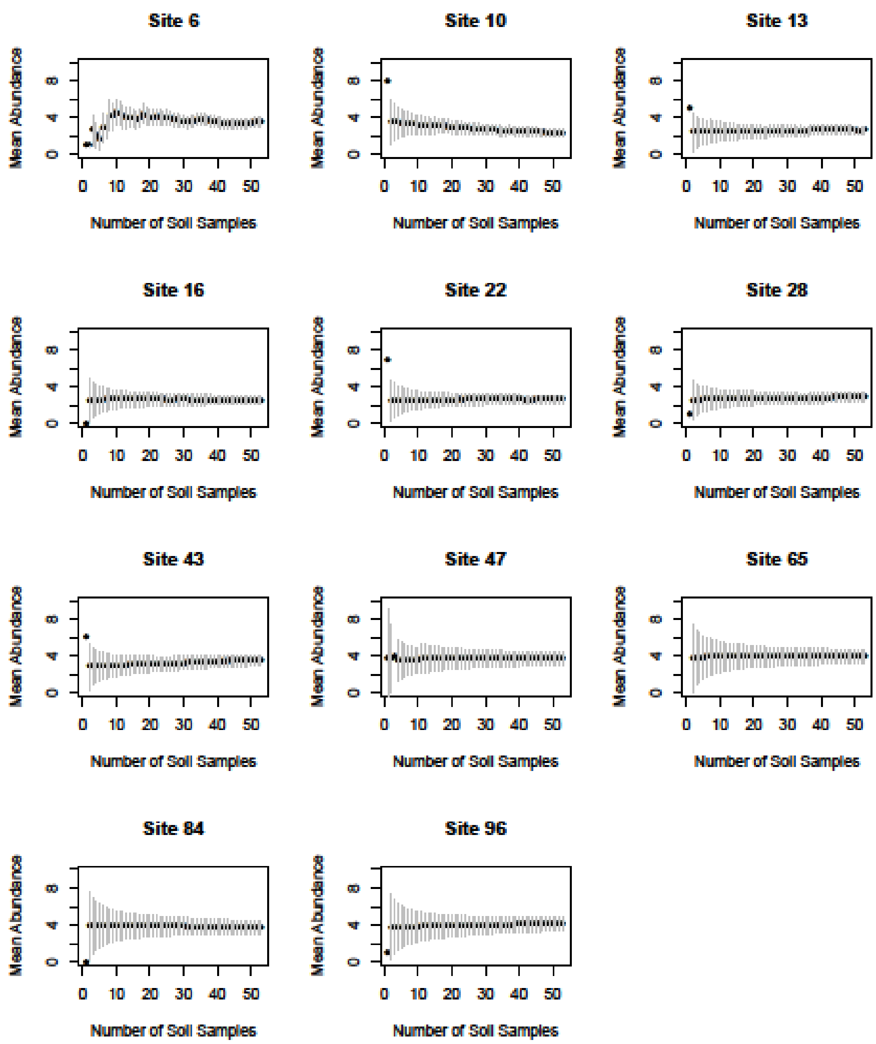


Figure C.1.1. Mean seed abundance (±SE) found within each of 11 pastures as a function of an increasing number of soil cores during subsampling. A maximum number of 53 cores were sampled across all sites.

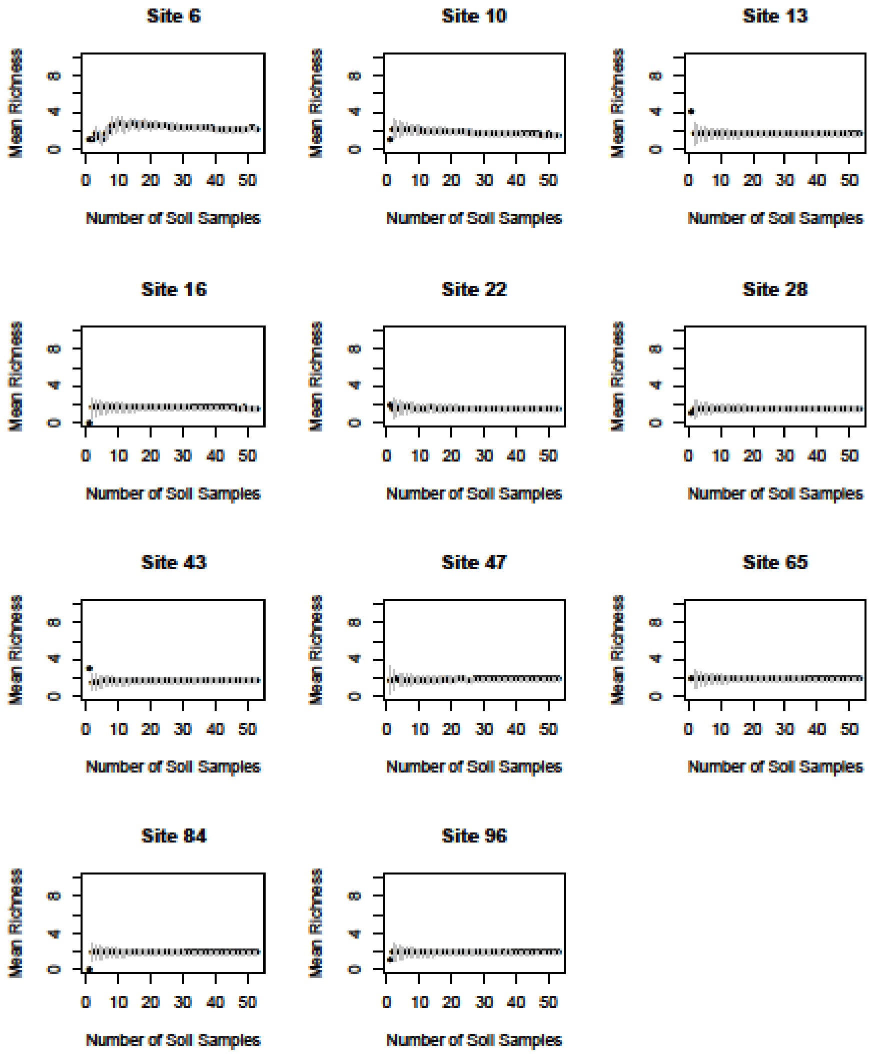


Figure C.1.2. Mean seed bank species richness (±SE) found within each of 11 pastures as a function of an increasing number of soil cores. A maximum number of 53 cores were sampled across each pasture.

The topographic position of soil cores affected seed bank composition (Table C.1.1), with cores sampled from uplands and mid-slope positions containing seed banks most similar to aboveground vegetation (Table C.1.2). Cores sampled from lowlands and mesic depressions contained seed banks unique from all other landscape positions. Mesic depressions in particular contained several unique sedges, grasses, and forbs characteristic of poorly drained soils (Table C.1.3). Aspect did not affect seed bank composition (Table C.1) and supports the idea that seeds can have limited dispersal, thus collecting numerous samples across representative areas of the pasture may be better at quantifying seed bank richness, diversity and composition.

Table C.1.1. Results of the perMANOVA examining differences in seed bank composition among topographical features.

Topographic Factor	Mean Square	F Model	R^2	P Value
Topographic Position	2.02	5.03	0.02	**0.002**
Aspect	0.90	2.26	0.03	0.319

Distance = Bray-Curtis, Permutations = 999

Bold: P < 0.05, Black: P < 0.10, Grey: P > 0.10

Table C.1.2. Results of the perMANOVA contrasts examining seed bank composition among topographic positions.

Topographic Position	Mean Square	F Model	R^2	P Value
Upland vs. Midslope	0.41	1.05	0.00	0.361
Upland vs. Lowland	2.60	6.24	0.02	**0.001**
Upland vs. Depression	2.15	5.54	0.04	**0.001**
Midslope vs. Lowland	3.06	7.39	0.02	**0.001**
Midslope vs. Depression	2.15	5.45	0.02	**0.001**
Lowland vs. Depression	1.47	3.39	0.02	**0.001**

Distance = Bray-Curtis, Permutations = 999

Bold: P < 0.05, Black: P < 0.10, Grey: P > 0.10

Table C.1.3. Indicator species associated with the seed bank among various topographic positions.

Topographic Position	Species	A	B	P value
Upland	*Chenopodium salinum*	0.92	0.06	0.017
Lowland	*Chenopodium album*	0.70	0.15	0.049
	Juncus tenuis	0.84	0.16	0.005
Depression	*Agrostis scabra*	0.80	0.13	0.008
	Agrostis stolonifera	1.00	0.13	0.003
	Alopecurus aequalis	1.00	0.07	0.028
	Carex praticola	0.79	0.13	0.013
	Chenopodium gigantospermum	0.80	0.07	0.040
	Erysimum cheiranthoides	0.87	0.07	0.045
	Gnaphalium uliginosum	0.75	0.33	0.001
	Juncus bufonius	0.86	0.40	0.001
	Plantago major	0.60	0.27	0.006
	Poa palustris	0.91	0.47	0.001
	Potentilla norvegica	0.58	0.20	0.017
	Rorippa palustris ssp. *palustris*	0.86	0.07	0.029
	Trifolium repens	0.62	0.13	0.025
Lowland + Depression	*Cerastium arvense*	0.99	0.06	0.028
	Ranunculus sceleratus ssp. *multifidus*	0.99	0.09	0.026

Permutations = 999

A = Probability of occurring, B = Fidelity

Appendix C.2 Summary Results of the NMDS of Seed Banks for All Pastures

Table C.2.1. Summary of significant management centroids from the NMDS of seed bank composition in Figure 5.3 ($P < 0.10$).

Management	R^2	P Value	Centroid	MDS 1	MDS 2
Cultivated	0.12	**0.001**	Cultivated	0.36	0.00
			Never Cultivated	0.00	0.10
			Unknown	-0.04	-0.02
Fire (Charcoal in Soil)	0.04	**0.023**	Fire	0.00	-0.10
			No Fire	0.00	0.04
Grazing System	0.04	0.100	Abandoned (None)	0.10	-0.13
			Continuous	0.05	-0.02
			Rotational	-0.05	0.03
Hay	0.05	**0.023**	Animals Fed Hay	0.01	0.10
			No Hay	0.00	0.05
			Unknown	0.00	-0.09
Herbivores	0.08	**0.030**	Cattle	-0.01	-0.01
			Horses	0.12	0.07
			Multiple Species	-0.03	-0.04
			Other	-0.34	0.16
			No Livestock (None)	0.10	-0.13
Manure	0.03	**0.048**	Manured	-0.08	0.06
			No Manure	0.03	-0.02
Swath/Mow	0.05	**0.016**	Swath/Mow	-0.23	-0.02
			No Swath/Mow	0.02	0.00

Table C.2.2. Biplot vectors associated with the final NMDS ordination of seedbank composition. Data were collected from 102 pastures across north central Alberta during 2012 and 2013 (Fig 5.3).

Biplot		**MDS 1**	**MDS 2**	**r^2**	**P Value**
Soil Properties	OM	0.23	0.97	0.10	**0.005**
	EC	0.39	0.92	0.10	**0.006**
	pH	0.98	-0.19	0.03	0.263
	N	0.24	0.97	0.09	**0.007**
	C	0.24	0.97	0.07	**0.025**
	C:N Ratio	0.71	-0.71	0.01	0.484
	Sand	0.86	0.51	0.10	**0.011**
	Clay	-0.99	-0.15	0.07	**0.031**
	Silt	-0.68	-0.73	0.08	**0.019**
	Compaction	-0.60	-0.80	0.08	0.154
Litter Depth	Depth	0.16	-0.99	0.01	0.469
Basal Cover	Vegetation	0.18	0.98	0.02	0.379
	Litter	0.23	-0.97	0.09	**0.012**
	Bare Ground	-0.55	0.84	0.06	0.058
	Manure	-0.74	0.67	0.01	0.603
	Rock	-0.99	-0.17	0.05	0.065
	Lichen	1.00	-0.09	0.06	0.060
	Moss	0.69	-0.73	0.02	0.392
	Wood	0.63	0.78	0.04	0.130
Pasture Characteristics	Years Farmed	0.55	-0.84	0.09	0.051
	Pasture Age	0.62	-0.79	0.17	**0.005**
Rangeland Health	Total RHA Score	-0.27	-0.96	0.05	0.078
	Forage Cover	-0.70	-0.71	0.04	0.144
	Cover of Tall Productive Forages	-0.23	-0.97	0.08	**0.020**
	Weedy & Ruderal Cover	0.43	-0.90	0.04	0.148
	Hydraulic Function & Litter	0.46	-0.89	0.03	0.163
	Soil Erosion	-0.98	-0.21	0.05	0.079
	Anthropogenic Bare Soil	0.36	-0.93	0.03	0.209
	Noxious Weed Cover	-0.88	-0.47	0.00	0.890
	Noxious Weed Density	-0.76	-0.65	0.01	0.579
	Woody Spp. Cover	-0.96	0.28	0.13	**0.002**
	Woody Spp. Density	-0.90	0.44	0.04	0.145
Similarity	Sorenson's	-0.38	-0.92	0.07	**0.017**
Seed Bank	Shannon's Diversity	0.41	0.91	0.17	**0.002**
	Simpson's Diversity	0.31	0.95	0.12	**0.003**
	Pielou's Evenness	-0.55	-0.83	0.10	**0.007**
	Richness	0.38	0.92	0.37	**0.001**
	Abundance	-0.13	0.99	0.22	**0.001**
	Total Graminoids	0.84	-0.54	0.02	0.302
	Total Broad Leaf	-0.22	0.97	0.33	**0.001**
	Total Native	0.58	0.81	0.24	**0.001**
	Total Introduced	-0.55	0.83	0.18	**0.001**
	Noxious Weeds	-0.84	0.55	0.00	0.836
	Legumes	-0.69	-0.72	0.01	0.548
	Woody	0.93	0.37	0.04	0.107
	Native Ruderal Forbs	0.19	0.98	0.15	**0.001**
	Native Perennial Forbs	0.98	0.21	0.27	**0.001**
	Introduced Ruderal Forbs	-0.42	0.91	0.33	**0.001**
	Seeded Graminoids	0.00	-1.00	0.08	**0.016**
	Native Grasses	0.84	0.54	0.17	**0.001**
	Ruderal Grasses	0.20	0.98	0.05	0.100
	Graminoids	0.68	0.73	0.12	**0.001**

Table C.2.3. Relationship of seed bank species' abundance to the NMDS (Fig. 5.3) axes (P < 0.10).

Species	MDS 1	MDS 2	r^2	P Value
Achillea millefloium	0.27	-0.07	0.18	**0.001**
Agrostis scabra	0.32	0.12	0.19	**0.001**
Amaranthus blitoides	0.02	1.01	0.05	0.089
Androsace septentrionalis	0.33	-0.17	0.19	**0.001**
Antennaria parvifolia	0.46	-0.22	0.10	**0.006**
Arabis Spp.	1.03	0.03	0.08	**0.014**
Artemisia frigida	0.89	-0.09	0.14	**0.001**
Campanula rotundifolia	0.57	-0.22	0.10	**0.005**
Capsella bursa-pastoris	-0.20	0.12	0.11	**0.006**
Cardamine pensylvanica	0.50	0.36	0.10	**0.004**
Carex praticola	0.51	-0.21	0.06	**0.033**
Carex rostrate	0.17	0.27	0.05	0.091
Carex Spp.	0.16	0.10	0.06	0.055
Carex sychnocephala	-0.04	0.16	0.08	**0.017**
Carum carvi	0.02	1.01	0.05	0.089
Cerastium arvense	0.69	0.05	0.11	**0.001**
Chenopodium album	-0.07	0.34	0.16	**0.001**
Chenopodium gigantospermum	0.23	0.54	0.14	**0.002**
Danthonia intermedia	0.43	-0.21	0.05	0.088
Descurainia Sophia	0.18	0.39	0.10	**0.009**
Elymus trachycaulus trachycaulus	0.41	0.15	0.06	0.054
Elytrigia repens	-0.05	-0.34	0.05	0.095
Epilobium ciliatum	-0.03	-0.24	0.06	0.059
Galeopsis tetrahit	0.21	0.08	0.06	**0.036**
Gnaphalium uliginosum	-0.32	0.27	0.17	**0.001**
Grlyceria grandis	0.17	0.25	0.08	**0.015**
Hordeum jubatum	-0.37	0.27	0.06	**0.041**
Houstonia longifolia	1.25	-0.39	0.10	**0.007**
Huechera richardsonis	1.07	0.29	0.11	**0.002**
Juncus arcticus	0.37	-0.42	0.07	**0.027**
Juncus tenuis	0.23	0.19	0.11	**0.002**
Koeleria macrantha	0.71	0.89	0.06	0.060
Lepidium densiflorum	-0.20	0.36	0.09	**0.013**
Matricaria discoidea	-0.35	0.31	0.07	**0.025**
Medicago sativa	-0.26	-0.29	0.05	0.075
Mentha arvensis	0.24	0.49	0.05	0.098
Monolepis nuttalliana	0.00	0.73	0.06	**0.047**
Penstemon procerus	0.68	0.16	0.08	**0.022**
Phleum pratense	-0.15	-0.27	0.08	**0.021**
Picea glauca	0.71	0.89	0.06	0.060
Plantago major	-0.03	0.05	0.06	**0.036**
Poa pratensis	-0.03	-0.26	0.11	**0.004**
Polygonum aviculare	-0.17	0.46	0.08	**0.023**
Polygonum convolvulus	-0.32	0.08	0.09	**0.016**
Polygonum lapathifolium	-0.27	0.12	0.06	**0.044**
Potentilla gracilis	1.07	0.49	0.08	**0.042**
Potentilla norvegica	0.16	-0.01	0.06	0.069
Potentilla pensylvanica	1.11	-0.22	0.17	**0.001**
Puccinellia nuttalliana	0.36	-0.08	0.04	0.096
Ranunculus macounii	0.12	0.36	0.05	0.088
Ranunculus sceleratus	0.21	0.17	0.07	**0.029**
Rorippa palustris	-0.28	0.26	0.09	**0.009**
Senecio vulgaris	-0.23	0.77	0.08	**0.016**
Solidago canadensis	0.64	0.20	0.09	**0.010**
Solidago missouriensis	0.91	-0.09	0.10	**0.008**
Sonchus arvensis	-0.41	0.14	0.08	**0.025**
Spergula arvensis	-0.43	0.11	0.09	**0.007**
Sporobolus cryptandrus	0.85	0.55	0.09	**0.010**
Stellaria media	-0.27	0.13	0.05	0.063
Stipa viridula	1.08	-0.04	0.07	0.054
Thermopsis rhombifolia	1.08	-0.04	0.07	0.054
Urtica dioica	0.17	0.28	0.09	**0.006**

Appendix C.3 Summary Tables for the NMDS of Seed Banks for Cultivated Pastures.

Table C.3.1. Summary of significant management centroids for the NMDS ordination (Fig. 5.5) of seed bank composition and pasture age (P < 0.10).

Management Factor	r^2	P Value	Centroid	NMDS 1	NMDS 2
Burrowing Mammals	0.03	0.085	Present	0.05	0.00
			Absent	-0.07	0.01
Feeding Hay in Pasture	0.06	0.069	Hay	0.14	0.04
			No Hay	0.03	0.04
			Unknown	-0.08	-0.05
Fire (Historical)	0.05	**0.019**	Fire (Charcoal in Soil)	-0.10	-0.05
			No Fire	0.05	0.02
Herbicide	0.04	0.078	Sprayed	-0.04	-0.18
			Not Sprayed	0.00	0.02
Herbivores	0.10	**0.048**	Cattle	-0.01	0.01
			Horses	-0.20	-0.02
			Multiple Herbivores	-0.01	-0.01
			Sheep/Alpaca	0.37	-0.04
			No Livestock	-0.12	-0.23
Manure	0.08	**0.002**	Manured	0.16	-0.03
			No Manure	-0.06	0.01

Table C.3.2. Significant biplot vectors for various pasture characteristics based on the NMDS (Fig. 5.5) of seedbank composition for cultivated pastures.

Biplot		**NMDS 1**	**NMDS 2**	**r^2**	**P Value**
Pasture Characteristics	Pasture Age	-0.71	-0.71	0.12	**0.012**
Rangeland Health	Total RHA Score	-0.56	-0.83	0.01	0.623
Similarity	Sorenson's	-0.31	-0.95	0.02	0.490
Seed Bank	Shannon's Diversity	0.19	0.98	0.23	**0.001**
	Simpson's Diversity	0.31	0.95	0.16	**0.003**
	Pielou's Evenness	0.21	-0.98	0.07	0.092
	Richness	0.12	0.99	0.35	**0.001**
	Abundance/Seed Density	0.66	0.75	0.13	**0.006**
	Total Graminoids	-1.00	-0.03	0.04	0.282
	Total Broad Leaf	0.77	0.64	0.23	**0.001**
	Total Native	-0.22	0.98	0.24	**0.001**
	Total Introduced	0.96	0.28	0.12	**0.016**
	Noxious Weeds	0.98	-0.21	0.01	0.650
	Legumes	-0.83	0.56	0.00	0.998
	Woody	-0.64	0.77	0.08	**0.045**
	Native Ruderal Forbs	-0.12	0.99	0.10	**0.030**
	Native Perennial Forbs	-0.87	0.49	0.19	**0.001**
	Introduced Ruderal Forbs	0.90	0.43	0.25	**0.001**
	Seeded Grasses	-0.75	-0.66	0.09	**0.032**
	Native Grasses	-0.55	0.84	0.11	**0.022**
	Ruderal Grasses	-0.03	1.00	0.12	**0.011**
	Graminoids	0.28	0.96	0.10	**0.032**

Appendix C.4. Summary of the CCA of the Seed Bank Constrained by Aboveground Vegetation

Table C.4.1. Axes included in reduced CCA model (Fig. 5.6) of seed bank constrained by plant community.

Axis	X^2	F Value	P Value
CCA 1	0.55	12.40	**0.001**
CCA 2	0.49	10.90	**0.001**
CCA 3	0.45	10.00	**0.001**
CCA 4	0.30	6.78	**0.001**
CCA 5	0.29	6.45	**0.001**
CCA 6	0.24	5.30	**0.001**
CCA 7	0.19	4.34	**0.001**
CCA 8	0.19	4.19	**0.001**
CCA 9	0.17	3.75	**0.001**
CCA 10	0.15	3.42	**0.003**
CCA 11	0.13	2.85	**0.001**
CCA 12	0.12	2.72	**0.001**
CCA 13	0.12	2.71	**0.001**
CCA 14	0.11	2.40	**0.007**
CCA 15	0.10	2.32	**0.002**
CCA 16	0.10	2.18	**0.003**
CCA 17	0.08	1.90	**0.037**
CCA 18	0.07	1.61	0.059
CCA 19	0.06	1.34	0.121
CCA 20	0.05	1.12	0.313
CCA 21	0.05	1.09	0.321
CCA 22	0.04	0.89	0.620
CCA 23	0.04	0.87	0.648
CCA 24	0.04	0.81	0.703
CCA 25	0.03	0.68	0.734
CCA 26	0.03	0.59	0.810
CCA 27	0.02	0.43	0.873

anova(cca)::vegan
Permutations = 999

Table C.4.2. Plant community variables permutationally selected for the CCA (Fig. 5.6) model (seed bank composition constrained by plant community cover).

Species	X^2	**F Value**	**P Value**
Achillea millefolium	0.09	1.96	0.071
Agrostis scabra	0.18	4.12	**0.019**
Alopecurus pratensis	0.07	1.50	0.220
Artemisia figida	0.39	8.71	**0.001**
Bromus inermis ssp. *pumpellianus*	0.09	2.12	0.086
Carex bebbii	0.12	2.80	0.070
Carex filifolia	0.11	2.52	**0.038**
Carex praegracilis	0.10	2.20	**0.034**
Chenopodium album	0.10	2.34	**0.013**
Dactylis glomerata	0.07	1.63	0.178
Elytrigia repens	0.13	2.99	**0.012**
Festuca hallii	0.53	11.90	**0.001**
Galeopsis tetrahit	0.14	3.03	**0.048**
Juncus arcticus ssp. *balticus*	0.17	3.89	**0.005**
Koeleria macrantha	0.18	3.98	**0.006**
Lepidium densiflorum	0.09	1.98	0.141
Lolium perenne	0.07	1.54	0.186
Phalaris aurundinacea	0.08	1.70	0.227
Plantago major	0.15	3.43	**0.041**
Poa pratensis	0.12	2.61	**0.003**
Polygonum aviculare	0.25	5.51	**0.006**
Silene latifolia ssp. *alba*	0.15	3.48	**0.036**
Spergula arvensis	0.13	2.94	**0.044**
Symphyotrichum laeve	0.12	2.72	**0.035**
Trifolium hybridum	0.08	1.87	0.113
Trifolium repens	0.07	1.61	0.174
Urtica dioica	0.40	8.93	**0.001**

anova(cca)::vegan
Permutations = 999

Table C.4.3. Species vectors for CCA (Fig. 5.6) of seedbank constrained by plant community. Seed bank vectors included at P < 0.1, while all plant community variables used in constrained ordination were included.

	Seed Bank				Plant Community			
Scientific Name	**CCA 1**	**CCA 2**	**r^2**	**P Value**	**CCA 1**	**CCA 2**	**r^2**	**P Value**
Achillea millefolium	-0.42	-0.82	0.09	0.089	-0.46	-0.29	0.28	**0.012**
Agrostis scabra	-3.17	0.68	0.77	**0.001**				
Agrostis stolonifera	-3.17	0.68	0.05	0.096				
Alopecurus pratensis	0.70	1.56	0.08	**0.049**	0.03	-0.07	0.01	0.609
Androsace septentrionalis	-0.66	-3.45	0.34	**0.013**				
Antennaria parvifolia	-0.72	-2.39	0.14	**0.041**				
Artemisia frigida	-1.22	-6.08	0.29	**0.020**	-0.17	-0.50	0.25	**0.019**
Bromus inermis ssp. *pumpellianus*					0.01	-0.05	0.00	0.489
Campanula rotundifolia	-0.91	-2.45	0.08	0.075				
Carex atherodes	0.49	0.74	0.02	0.093				
Carex bebbii					-0.04	-0.14	0.02	0.164
Carex filifolia					-0.58	0.04	0.33	**0.013**
Carex praegracilis					-0.74	0.17	0.56	**0.007**
Carex praticola	-1.54	0.01	0.09	0.088				
Cerastium arvense	-2.55	-0.55	0.16	**0.040**				
Chenopodium album	0.95	2.13	0.66	**0.001**	0.25	0.57	0.35	**0.015**
Chenopodium gigantospermum	-0.19	0.54	0.11	0.067				
Dactylis glomerate					0.15	0.23	0.07	0.110
Danthonia intermedia	-0.75	0.43	0.07	0.083				
Deschampsia cespitosa					-0.22	-0.20	0.05	0.099
Descurainia sophia	0.35	0.41	0.09	0.094				
Eleocharis acicularis	-1.23	0.09	0.02	0.089				
Elytrigia repens					0.31	0.56	0.37	**0.001**
Festuca hallii	-3.05	0.29	0.08	**0.046**	-0.93	0.29	0.95	**0.001**
Festuca saximontana	-1.44	0.04	0.05	0.086				
Fragaria virginiana	-2.07	0.40	0.50	**0.003**				
Galeopsis tetrahit					0.16	0.47	0.22	**0.036**
Galium aparine	0.97	1.96	0.14	**0.040**				
Galium boreale	-3.54	0.29	0.08	**0.046**				
Heuchera richardsonii	-6.96	1.85	0.84	**0.001**				
Houstonia longifolia	-1.20	-7.75	0.24	**0.026**				
Juncus acticus ssp. *balticus*	-3.98	1.03	0.66	**0.002**	-0.88	0.23	0.83	**0.001**
Koeleria macrantha					-0.30	-0.02	0.09	0.053
Lepidium densiflorum					0.02	0.00	0.00	0.874
Lolium perenne	0.55	0.86	0.18	**0.042**	0.05	0.02	0.00	0.415
Penstemon procerus	-8.36	3.04	0.86	**0.001**				
Phalaris aurundinacea					0.03	0.03	0.00	0.806
Plantago major					0.01	-0.01	0.00	0.983
Poa pratensis					0.01	-0.28	0.07	0.096
Poa secunda	-3.05	0.29	0.08	**0.046**				
Polygonum aviculare					-0.07	-0.02	0.01	0.507
Potentilla gracilis	-9.02	3.27	0.86	**0.013**				
Potentilla pensylvanica	-1.07	-5.66	0.26	**0.027**				
Ranuculus rhomboideus	-0.47	-4.04	0.19	**0.040**				
Ranunculus sceleratus ssp. *multifidus*	0.18	0.61	0.07	0.067				
Senecio vulgaris	0.86	1.74	0.19	**0.047**				
Setaria viridis	0.51	0.78	0.04	0.099				
Silene latifolia ssp. alba					0.04	-0.09	0.01	0.251
Solidago canadensis	-4.87	0.42	0.70	**0.005**				
Solidago missouriensis	-1.75	-3.64	0.19	**0.038**				
Spergula arvensis					0.00	-0.02	0.00	0.965
Symphyotrichum laeve					0.01	-0.09	0.01	0.189
Trifolium hybridum					0.09	-0.02	0.01	0.703
Trifolium repens					0.09	-0.15	0.03	0.324
Urtica dioica	1.25	3.22	0.35	**0.009**	0.29	0.67	0.49	**0.017**

Table C.4.4. Summary of significant management centroids for CCA (Fig. 5.6) of seed bank constrained by plant community (P < 0.10).

Management	r^2	P Value	Centroid	CCA 1	CCA 2
Cultivation	0.22	**0.001**	Cultivated	0.14	-0.14
			Never Cultivated	-2.31	0.15
			Unknown	0.29	0.38
Feeding Hay	0.09	**0.014**	Hay	0.43	0.80
			No Hay	-0.09	-0.07
			Unknown	-0.12	-0.32
Grazing Intensity	0.17	**0.045**	U	-2.33	0.05
			L	0.04	-0.39
			LM	0.17	0.10
			M	-0.02	-0.30
			MH	0.22	0.32
			H	0.24	0.10
Grazing System	0.14	**0.033**	Continuous	0.08	-0.13
			Rotational	0.14	0.06
			Not Grazed	-0.23	0.53
Herbivores	0.17	0.068	Cattle	0.04	-0.11
			Horses	0.25	0.00
			Mult. Herbivores	0.19	-0.10
			Sheep/Alpaca	0.59	0.96
			No Livestock	-2.33	0.53
Timing of Grazing	0.20	**0.034**	All Year	0.67	1.28
			Growing Season	0.08	-0.11
			Winter	-0.08	-0.11
			Not Grazed	-2.36	0.53

Table C.4.5. Significant biplot vectors for CCA (Fig. 5.6) of seedbank composition.

Biplot		**CCA1**	**CCA 2**	**r^2**	**P Value**
Similarity	Sorenson's	-0.99	0.16	0.01	0.838
Seed Bank	Shannon's Diversity	-0.25	-0.97	0.06	0.145
	Simpson's Diversity	-0.35	-0.94	0.07	0.094
	Pielou's Evenness	-0.07	-1.00	0.15	**0.021**
	Richness	-0.14	0.99	0.01	0.727
	Abundance/Seed Density	0.09	1.00	0.19	**0.005**
	Total Graminoids	-1.00	-0.01	0.22	**0.013**
	Total Broad Leaf	0.52	0.85	0.28	**0.002**
	Total Native	-0.99	0.14	0.36	**0.003**
	Total Introduced	0.67	0.75	0.39	**0.001**
	Noxious Weeds	0.64	-0.77	0.02	0.238
	Legumes	0.99	-0.15	0.02	0.518
	Woody	-0.34	-0.94	0.00	0.824
	Native Ruderal Forbs	-0.23	-0.97	0.01	0.763
	Native Perennial Forbs	-1.00	-0.07	0.50	**0.005**
	Introduced Ruderal Forbs	0.59	0.81	0.49	**0.001**
	Seeded Grasses	-0.23	-0.97	0.04	0.191
	Native Grasses	-0.99	-0.13	0.01	0.726
	Ruderal Grasses	-0.93	0.38	0.06	0.135
	Graminoids	-0.95	0.31	0.71	**0.002**

Appendix C.5 Seed Bank of Stockpiled Manure

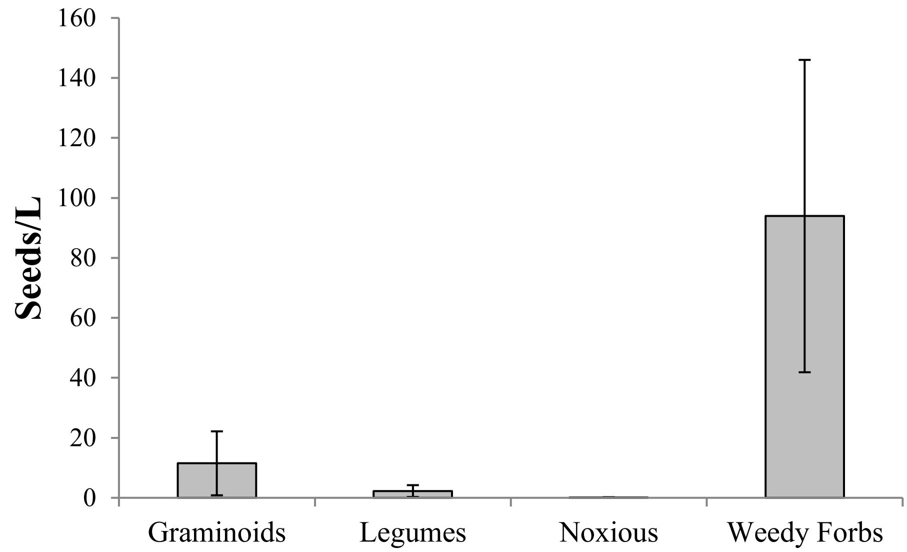


Figure C.5.1. Seed density (seeds/L ± 1 SE) of stock piled manure collected when producers confirmed that they spread manure on their pasture(s). Note that the noxious weeds category included one forb, stork's bill (*Erodium cicutarium*), and one grass, green foxtail (*Setaria viridis*).

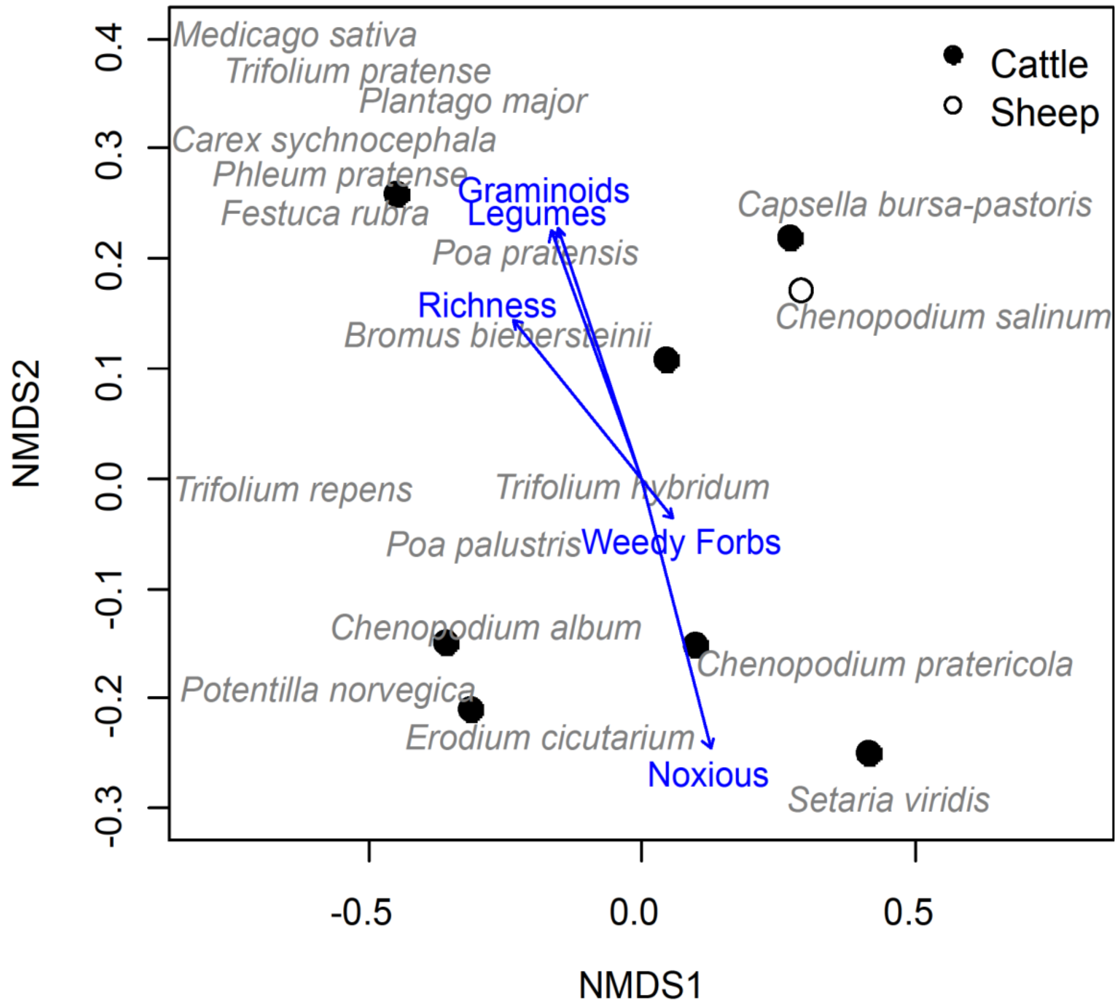


Figure C.5.2. NMDS ordination of the germinable seeds identified in stockpiled manure (distance = Bray-Curtis, dimensions = 2, stress = 0.04). The majority of manure piles were comprised of cattle manure (closed circles), while one pile consisted of sheep manure (open circle). All species and biplots vectors are displayed.

Appendix C.6 Complete ISA Tables

Table C.6.1. Indicator species analysis of seed bank's species association with management factors ($P < 0.05$) in north central Alberta's pastures.

Management	Category	Species	A	B	P value
Ownership	Rented	*Cardamine pensylvanica*	0.93	0.20	0.043
		Chenopodium capitatum	0.97	0.20	0.019
		Cirsium arvense	0.65	0.80	0.022
		Elymus trachycaulus ssp. *trachycaulus*	0.98	0.20	0.024
		Rumex salicifolius	1.00	0.20	0.012
Cultivated	Never	*Achillea millefolium*	0.73	0.75	0.001
		Antennaria Spp.	0.91	0.25	0.021
		Artemisia frigida	0.71	0.38	0.013
		Campanula rotundifolia	0.94	0.38	0.002
		Cardamine pensylvanica	0.80	0.25	0.031
		Carex praticola	0.97	0.25	0.006
		Cerastium arvense	0.96	0.50	0.001
		Corydalis aurea	0.91	0.25	0.014
		Agrostis scabra	0.90	0.63	0.002
		Festuca saximontana	0.80	0.25	0.026
		Juncus arcticus ssp. *balticus*	0.95	0.25	0.015
		Penstemon procerus	0.99	0.38	0.004
		Solidago canadensis	0.97	0.25	0.007
	Unknown	*Medicago sativa*	0.75	0.41	0.016
		Ranunculus macounii	0.84	0.29	0.012
	Unknown + Never	*Alopecurus aeqalis*	0.93	0.28	0.027
		Lepidium densiflorum	0.90	0.44	0.031
	Unknown + Cultivated	*Capsella bursa-pastoris*	1.00	0.51	0.032
		Chenopodium album	1.00	0.59	0.045
System	None (Abandoned)	*Medicago sativa*	0.75	0.75	0.008
		Danthonia intermedia	0.83	0.50	0.008
		Trifolium pratense	0.71	0.50	0.025
		Carex aenea	0.51	0.50	0.036
		Potentilla gracilis	1.00	0.25	0.032
		Penstemon procerus	0.99	0.25	0.021
		Huechera richardsonis	0.98	0.25	0.021
		Juncus tracyi	0.96	0.25	0.045
		Solidago canadensis	0.93	0.25	0.011
		Juncus arcticus	0.87	0.25	0.027
Timing	Never (Abandoned)	*Danthonia intermedia*	0.83	0.50	0.015
	Winter Grazed	*Astragalus cicer*	0.98	0.33	0.021
		Brassica napus	0.92	0.33	0.034
		Carex atherodes	1.00	0.33	0.027
		Festuca ovina var. *duriuscula*	0.91	0.33	0.048
		Festuca rubra	0.87	1.00	0.009
		Medicago lupulina	0.84	0.33	0.029
		Phleum pratense	0.61	1.00	0.037
		Polygonum lapathifolium	0.87	0.67	0.019
		Rumex crispus	0.92	0.33	0.032
		Senecio vulgaris	0.90	0.33	0.043
	Grazed Year Round + Winter	*Urtica dioica*	0.97	0.55	0.039
	Abandoned + Winter	*Medicago sativa*	0.88	0.71	0.005
		Trifolium pratense	0.94	0.43	0.048
		Penstemon procerus	1.00	0.29	0.049
System x Timing	Abandoned	*Danthonia intermedia*	0.85	0.50	0.008
	Winter Grazed	*Astragalus cicer*	0.98	0.33	0.021
		Brassica napus	0.92	0.33	0.034
		Carex atherodes	1.00	0.33	0.021
		Festuca ovina var. *duriuscula*	0.91	0.33	0.048
		Festuca rubra	0.77	1.00	0.014
		Medicago lupulina	0.84	0.33	0.029
		Phleum pratense	0.53	1.00	0.009
		Poa pratensis	0.49	1.00	0.013
		Polygonum lapathifolium	0.80	0.67	0.009
		Rumex crispus	0.92	0.33	0.032
		Senecio vulgaris	0.90	0.33	0.043
	Grazed Year Round + Winter	*Urtica dioica*	0.95	0.55	0.041
	Abandoned + Winter	*Medicago sativa*	0.83	0.67	0.016
		Trifolium pratense	0.92	0.50	0.012
Herbivores	Sheep/Alpaca	*Brassica kaber*	0.76	0.50	0.030
		Gnaphalium uliginosum	0.65	1.00	0.035
		Polygonum convolvulus	0.68	0.75	0.025
		Sonchus arvensis	0.90	0.75	0.001
	No Livestock	*Danthonia intermedia*	0.89	0.50	0.011
		Medicago sativa	0.63	0.75	0.017
Herbicide	Sprayed	*Urtica dioica*	0.92	0.38	0.049
	Not Sprayed	*Agrostis scabra*	1.00	0.29	0.040

Fertilized	Fertilized	*Elytrigia repens*	0.80	0.44	0.024
Manure	Manure Spread	*Chenopodium salinum*	0.95	0.60	0.001
		Erysimum cheiranthoides	0.80	0.28	0.024
		Schedonorus arundinaceus	0.91	0.12	0.033
		Hordeum jubatum	0.84	0.24	0.025
		Matricaria matricarioides	0.69	0.20	0.049
		Poa compressa	0.85	0.24	0.036
		Stellaria media	0.81	0.40	0.021
Harrowed	Harrowed	*Bromus inermis pumpelianus*	1.00	0.09	0.043
		Chenopodium salinum	0.80	0.44	0.003
		Polygonum lapathifolium	0.85	0.32	0.010
		Silene alba	0.83	0.29	0.045
		Stellaria media	0.78	0.38	0.040
	Not Harrowed	*Androsace septenrionalis*	0.86	0.37	0.037
		Bromus biebersteinii	0.90	0.26	0.010
Aeration	Aerated	*Brassica napus*	0.98	0.25	0.040
		Bromus ciliatus	0.99	0.25	0.046
		Carex atherodes	1.00	0.25	0.040
		Chenopodium album	0.69	1.00	0.039
		Medicago lupulina	0.96	0.25	0.041
		Medicago sativa	0.88	0.50	0.025
		Melilotus officinalis	1.00	0.25	0.046
		Poa compressa	0.82	0.50	0.036
		Silene latifolia sbsp. *alba*	0.86	0.50	0.037
		Trifolium hybridum	0.66	1.00	0.037
		Trifolium pratense	0.88	0.50	0.020
		Trifolium repens	0.78	1.00	0.014
	Not Aerated	Poa pratensis	0.79	1.00	0.042
Swathed or Mowed	Swathed or Mowed	*Astragalus cicer*	0.99	0.22	0.015
		Polygonum lapathifolium	0.84	0.44	0.022
		Spergula arvensis	0.81	0.56	0.017
		Trifolium pratense	0.84	0.33	0.026
Hay In Pasture	Animals Fed Hay	*Beckmannia syzigachne*	0.71	0.31	0.023
		Chenopodium album	0.76	0.75	0.006
		Setaria viridis	1.00	0.13	0.027
		Urtica dioica	0.90	0.50	0.007
	No Hay	*Carex* Spp.	0.68	0.63	0.003
		Cerastium vulgatum	0.76	0.21	0.042
		Poa compressa	0.72	0.26	0.033
		Stellaria media	0.73	0.37	0.044
		Typha latifolia	0.81	0.19	0.049
	Unknown	*Carex aenea*	1.00	0.23	0.004
		Festuca rubra	0.84	0.51	0.032
		Poa pratensis	0.50	1.00	0.019
Burrowing Mammals	Burrows	*Limosella aquatica*	1.00	0.15	0.013
	No Burrows	*Carex rostrata*	0.91	0.17	0.389
		Polygonum aviculare	0.87	0.14	0.015
Recent Fire	No Fire	*Juncus bufonius*	0.94	0.41	0.049
	Fire (Survey)	*Bromus anomalus*	0.94	0.27	0.002
		Cardamine pensylvanica	0.93	0.13	0.042
		Elymus trachycaulus ssp. *trachycaulus*	0.98	0.13	0.017
		Fragaria virginiana	0.82	0.20	0.048
		Galeopsis tetrahit	0.72	0.40	0.029
		Geranium bicknellii	0.89	0.33	0.010
		Sonchus arvensis	0.74	0.40	0.011
		Typha latifolia	0.93	0.33	0.002
Historical Fire	No Fire	*Chenopodium album*	0.88	0.63	0.012
		Gnaphalium uliginosum	0.96	0.63	0.001
	Fire (Charcoal in Soil)	*Bromus anomalus*	0.86	0.13	0.031
		Carex bebbii	1.00	0.19	0.001
		Dracocephalum parviflorum	0.93	0.13	0.029
		Geranium bicknellii	0.89	0.26	0.018

ISA ran in R using *indicspecies:multipatt* (Caceres and Legendre, 2009).
A = Probability of occurring, B = Fidelity

Table C.6.2. Indicator species analysis for the rangeland health assessment (RHA) categories describing shifts in seed bank composition (P < 0.05) in north central Alberta's pastures.

RHA Category	Score	Species	A	B	P value
Plant Community	Tame	*Chenopodium album*	0.99	0.59	0.025
		Capsella bursa-pastoris	0.96	0.52	0.031
	Modified-Tame	*Achillea millefolium*	0.78	0.67	0.001
		Agrostis scabra	0.94	0.50	0.003
		Androsace septentrionalis	0.91	0.50	0.008
		Antennaria parvifolia	0.96	0.25	0.005
		Arabis holboellii var. *retrofracta*	1.00	0.17	0.014
		Artemisia frigida	0.99	0.42	0.001
		Campanula rotundifolia	0.96	0.42	0.001
		Cardamine pensylanica	0.97	0.25	0.004
		Carex aenea	0.60	0.33	0.047
		Carex praticola	0.96	0.17	0.015
		Cerastium arvense	0.97	0.50	0.001
		Corydalis aurea	0.94	0.17	0.038
		Danthonia intermedia	0.83	0.25	0.016
		Huechera richardsonii	1.00	0.17	0.014
		Juncus balticus	0.94	0.17	0.034
		Penstemon procerus	0.99	0.25	0.007
		Potentilla pensylvanica	1.00	0.25	0.002
		Solidago canadensis	0.96	0.17	0.019
		Solidago missouriensis	0.93	0.17	0.024
		Sporobulus cryptandrus	1.00	0.17	0.013
Forage Cover	Score 9	*Artemisia frigida*	0.96	0.19	0.045
		Cerastium arvense	0.92	0.22	0.032
		Potentilla pensylvanica	1.00	0.11	0.039
	Score 5	*Glyceria grandis*	0.75	0.33	0.024
		Polygonum aviculare	0.90	0.22	0.018
	Score 5 + 9	*Danthonia intermedia*	1.00	0.14	0.042
Cover of Tall Productive Forages	Score 7	*Juncus bufonius*	0.83	0.71	0.046
	Score 0	*Amaranthus blitoidies*	1.00	0.50	0.022
		Amaranthus retroflexus	1.00	0.50	0.020
		Beckmannia syzigachne	0.90	1.00	0.004
		Capsella bursa-pastoris	0.70	1.00	0.034
		Carum carvi	1.00	0.50	0.022
		Chenopodium salinum	0.54	1.00	0.037
		Corydalis aurea	0.79	0.50	0.050
		Hordeum jubatum	0.87	1.00	0.004
		Lepidium densiflorum	0.95	1.00	0.001
		Tripleurospermum perforatum	0.74	1.00	0.005
		Monolepis nuttalliana	0.99	0.50	0.016
		Plantago elongata	1.00	0.50	0.022
Weedy & Ruderal Cover	Score 7	*Capsella bursa-pastoris*	0.56	1.00	0.019
		Lepidium densiflorum	0.89	0.40	0.037
		Limosella aquatica	0.81	0.30	0.024
		Matricaria discoidea	0.80	0.30	0.040
		Polygonum aviculare	0.94	0.20	0.025
		Schedonorus arundinaceus	0.80	0.20	0.049
		Tripleurospermum perforatum	0.68	0.30	0.045
Hydraulic Function & Litter	Score 0	*Amaranthus blitoidies*	1.00	0.25	0.043
		Amaranthus retroflexus	0.99	0.25	0.026
		Carum carvi	1.00	0.25	0.043
		Dactylis glomerata	0.87	0.25	0.015
		Lepidium densiflorum	0.76	0.75	0.011
		Monolepis nuttalliana	0.94	0.25	0.043
		Polygonum aviculare	0.80	0.50	0.009
		Polygonum lapathifolium	0.72	0.50	0.035
	Score 8+16+25	*Taraxacum officinale*	0.99	0.91	0.004
Soil Erosion	Score 10	*Bromus anomalus*	1.00	0.11	0.042
	Score 7 + 4	*Descurainia sophia*	0.87	0.25	0.027
		Galeopsis tetrahit	0.92	0.29	0.013
	Score 4	*Populus balsamifera*	0.83	0.20	0.008
		Urtica dioica	0.86	0.53	0.010
Anthropogenic Bare Soil	Score 0	*Alopercurus aequalis*	0.95	0.40	0.019
		Amaranthus blitoidies	1.00	0.20	0.046
		Amaranthus retroflexus	0.99	0.20	0.042
		Carum carvi	1.00	0.20	0.046
		Eleocharis acicularis	0.99	0.20	0.042
		Elymus trachycaulus ssp. *trachycaulus*	0.98	0.20	0.040
		Festuca rubra	0.90	0.60	0.037
		Festuca saximontana	0.97	0.40	0.005
		Lepidium densiflorum	0.77	0.60	0.022
		Monolepis nuttalliana	0.98	0.20	0.031
		Polygonum aviculare	0.83	0.40	0.018
		Spergula arvensis	0.66	0.60	0.046

		Sporobulus cryptandrus	0.98	0.20	0.043
Noxious Weed Cover	Score 5	*Agrostis stolonifera*	0.86	0.16	0.037
	Score 3	*Potentilla norvegica*	0.63	0.74	0.045
	Score 1	*Cirsium arvense*	0.73	0.53	0.017
		Rumex salicifolius	1.00	0.12	0.022
		Mentha arvensis	0.91	0.12	0.046
Noxious Weed Density	Score 5	*Arabis holboellii* var. *retrofracta*	1.00	0.12	0.021
	Score 3	*Veronica peregrina*	0.50	0.84	0.020
Woody Cover	Score 3	*Androsace septentrionalis*	0.89	0.64	0.001
		Artemisia frigida	0.98	0.21	0.006
		Bromus anomalus	0.84	0.21	0.019
		Bromus ciliatus	0.68	0.14	0.049
		Campanula rotundifolia	0.94	0.36	0.001
		Carex spp.	0.58	0.64	0.045
		Cerastium arvense	0.65	0.36	0.019
		Potentilla pensylvanica	1.00	0.21	0.002
		Puccinellia nuttalliana	0.95	0.14	0.037
		Rumex crispus	0.68	0.21	0.048
		Sporobulus cryptandrus	1.00	0.14	0.022
		Typha latifolia	0.92	0.29	0.006
Wood Density	Score 2	*Artemisia frigida*	0.89	0.25	0.031
		Danthonia intermedia	0.97	0.38	0.001
		Potentilla pensylvanica	0.98	0.25	0.005
		Tripleurospermum perforatum	0.94	0.25	0.020
	Score 0	*Bromus anomalus*	0.90	0.19	0.041
		Carex rostrata	0.82	0.24	0.044
	Score 0 + 2	*Androsace septentrionalis*	0.92	0.48	0.018
Grazing Intensity	U	*Danthonia intermedia*	0.72	0.50	0.007
		Huechera richardsonis	0.95	0.25	0.029
		Juncus tracyi	0.90	0.25	0.043
		Medicago sativa	0.47	0.75	0.011
		Potentilla gracilis	1.00	0.25	0.047
		Solidago canadensis	0.87	0.25	0.029
	L	*Festuca ovina* var. *duriuscula*	1.00	0.22	0.023
	H	*Limosella aquatica*	0.80	0.38	0.018
		Matricaria perforata	0.64	0.38	0.034
		Plantago major	0.42	1.00	0.040
		Rumex crispus	0.85	0.25	0.039
	H + L	*Medicago lupulina*	1.00	0.24	0.010
	U+L+M+MH+H	*Thlaspi arvense*	0.95	0.82	0.010
		Trifolium repens	0.97	0.69	0.001

ISA ran in R using *indicspecies:multipatt* (Caceres and Legendre, 2009).
A = Probability of occurring, B = Fidelity

Appendix C.7. Seed Bank Relationship to Rangeland Health

Questions from the rangeland health assessment (RHA) were examined for their relationship with shifts in seed bank composition. Scores were linked to significant differences seed densities for functional groups, shifts in composition, and indices of diversity. However overall health scoring of healthy, healthy with problems, and unhealthy was not significantly associated with shifts in seed bank characteristics or composition.

The first question (Appendix A.2) distinguishes modified-tame and tame pastures ($P = 0.009$; Table C.7.1). Modified-tame pasture communities were associated with higher seed densities of graminoids, native perennial forbs and native perennial grasses (Table C.7.3). Seed densities of introduced species were 48.8% lower in modified pastures compared to tame pastures ($P = 0.006$; Table C.7.4 and C.7.5), which in turn, was explained by fewer introduced ruderal forbs in the former ($P < 0.001$; Table C.7.6 and C.7.7). In contrast, the density of native perennial forbs was more than 10 times greater in modified-tame pastures (741 ±114 seeds/m^2) compared to that in tame pastures (68 ±42 seeds/m^2) ($P < 0.001$; Table C.7.6 and C.7.7). Richness and diversity within the seed bank did not differ between tame and modified-tame pastures, but each was associated with a unique suite of plant species. The introduced ruderal forbs *Chenopodium album* and *Capsella bursta-pastoris* were indicators of tame pastures, while seed from common yarrow, fringed sage and harebell were strong indicators of modified-tame pastures ($P = 0.001$; Table C.6.2). Note that modifie-tame pastures have seed bank characteristics that resemble non-cultivated fields described in Chapter 5.

Seed bank composition was not associated with the scores of cover forage classes ($P \geq 0.80$; Table C.7.1). Native perennial forb abundance was an indicator of the second highest RHA score (RHA score = 9) (Table C.7.3), which was the maximum possible score that modified pasture communities containing perennial forbs could receive. Correspondingly, the native perennial forbs *Artemisia frigida*, *Cerastium arvense*, and *Potentilla pensylanica* were all indictors of this RHA score ($P < 0.05$; Table C.6.2). The next lowest score (RHA scaore = 5) was associated with *Glyceria grandis* and *Polygonum* *aviculare*, while pastures with scores of 5 or 9 contained the native grass *Danthonia intermedia* (Table C.6.2).

A decrease in the relative canopy cover of tall productive forage species was associated with differences in seed bank composition (P = 0.045; Table C.7.1). Seed banks differed the most between pastures with the highest score (RHA score = 14), representing ≥75% (relative) cover, and those with <40% cover representing the lowest score (RHA score = 0; P = 0.035; Table C.7.2). Seed banks from pastures with aboveground forage cover of between 40% to 74% (RHA score = 7) were marginally dissimilar from pastures with < 40% cover (RHA score = 0; P = 0.084; Table C.7.2). Among individual plant species, RHA scores = 7 were associated with *Juncus bufonius* (p = 0.046), a weedy rush (Table C.6.2). When cover decreased to < 40% (RHA score = 0), native ruderal forbs and graminoids became abundant (P < 0.1; Table C.7.3), including the noxious weed *Matricaria perforata* (P = 0.005; Table C.6.2).

Evidence of soil erosion was linked to divergence in seed bank composition (P = 0.008). *Bromus anomalus* (P = 0.042) was present exclusively in pastures with stable soils (RHA score = 10), while increasingly more ruderal species (P < 0.05) were associated with intermediate erosion (RHA scores = 7 & 4), with the exception of *Populus balsamifera* (Table Table C.6.2). Anthropogenic increases in bare soil were associated with shifts in seed bank composition (P = 0.019; Table C.7.1). The lowest scores for bare soil (RHA score = 0) were associated with native ruderal forbs, native species overall, and ruderal grasses (Ps < 0.019; Table C.7.3)

Noxious weed density was not associated with significant shifts in seed bank composition (P = 0.288; Table C.7.1). However, low scores (RHA score = 0) were associated with introduced ruderal robs and total broad leaf plants in the seed bank (Ps < 0.049) while noxious weeds were associated with scores of 0 to 5 (P = 0.028; Table C.7.3). High noxios weed seed densities were associated with the lowest scores for both noxious weed criteria (Ps < 0.042; Table C.7.6 and C.7.7).

Litter quantity, a measure of hydraulic function and indicator of ecological function, was not associated with distinct shifts in seed bank communities (P = 0.125; Table C.7.1). Pastures with sparse or absent litter (RHA score = 0) were associated primarily with introduced annuals like *Amaranthus* spp., *Lepidium densiflorum, Polygonum aviculare* (Ps < 0.043) in the seed bank, and also included *Dactylis glomerata* which is a forage grass decreaser (P = 0.015; Table C.6.2). Thin, sparse litter was also associated with higher similarity between the plant community and seed bank (P = 0.016; Tables C.7.8 and C.7.9), this was likely caused by the recruitment of ruderal species from the seed bank with disturbances like heavy grazing. Common pasture weed *Taraxacum officinale* was associated with higher litter scores (scores 8 through 25) (P = 0.004; Table C.6.2).

Encroachment of woody species, scored aboveground in the RHA by their cover (P = 0.027) and density (i.e., level of infestation) (P = 0.04) were associated with shifts in seed bank composition (Table C.7.1). Where distinct differences were found between pastures with low and heavy infestations of woody vegetation (P = 0.025; Table C.7.2). Pastures with 5 % to 15% woody cover (RHA score = 3) were positively associated with native perennial forbs in the seed bank (P = 0.006), while pastures with heavy infestations of woody species (RHA score = 0) were weakly associated with woody species in the seed bank (P = 0.062, Table C.7.3; and P =0.056, Table C.7.6). Seed densities of introduced ruderal forbs significantly higher and native perennial forbs were lowered when woody cover was less than 5% (RHA score = 6) when compared to a woody cover > 5% (RHA score = 3) (Tables C.7.6 and C.7.7). Seeds from native prairie forbs like *Artemisia frigida*, native Parkland grasses like *Bromus anomalus* and *B. ciliatus*, and graminoids like *Typha latifolia* were associated with lower woody cover scores (RHA score = 3) (Ps < 0.05; Table C.6.2). Note that germination of woody species was relatively low (Figure 5.1), which could explain its weak relationships the RHA's assessment of woody encroachment.

Table C.7.1. Results of the perMANOVA tests evaluating seed bank composition responses to rangeland health assessment factors based on the assessment of 102 sample sites examined across north central Alberta during 2012 and 2013.

RHA Category	Mean Square	F Model	R2	P Value
Plant Community Type	0.656	2.163	0.021	**0.009**
Forage Cover	0.247	0.803	0.016	0.802
Cover of Tall Productive Forages	0.432	1.421	0.028	**0.045**
Weedy & Ruderal Cover	0.276	0.898	0.009	0.578
Hydraulic Function & Litter	0.376	1.234	0.036	0.125
Soil Erosion	0.485	1.601	0.031	**0.012**
Anthropogenic Bare Soil	0.454	1.494	0.029	**0.019**
Noxious Weed Cover	0.340	1.111	0.022	0.295
Noxious Weed Density	0.333	1.088	0.032	0.288
Woody spp Cover	0.538	1.767	0.017	**0.027**
Woody spp Density	0.442	1.455	0.029	**0.040**
Grazing Intensity	0.313	1.021	0.051	0.422
Health	0.307	1.002	0.020	0.466

Distance = Bray-Curtis, Permutations = 999

Table C.7.2. Results of the perMANOVA contrasts assessing the influence of rangeland health scores on pasture seed bank composition.

Rangeland Health	Scores	Mean Square	F Model	R^2	P Value
Cover of Tall Productive Forages	0 * 7	0.45	1.40	0.08	0.084
	0 * 14	0.52	1.73	0.02	**0.035**
	7 * 14	0.36	1.17	0.01	0.242
Soil Erosion	4 * 7	0.50	1.57	0.03	**0.045**
	4 * 10	0.32	1.11	0.02	0.288
	7 * 10	0.59	1.96	0.02	**0.011**
Anthropogenic Bare Soil	0 * 3	0.53	1.54	0.07	**0.048**
	0 * 5	0.76	2.54	0.03	**0.003**
	3 * 5	0.17	0.56	0.01	0.937
Woody Spp Density	0 * 2	0.28	0.96	0.03	0.485
	0 * 4	0.53	1.77	0.02	**0.025**
	2 * 4	0.39	1.27	0.02	0.162

Distance = Bray-Curtis, Permutations = 999

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$

Table C.7.3. Indicator species analysis of the seed bank's functional plant group in response to various management factors and rangeland health (P < 0.10).

Rangeland Health	Category	Functional Group	A	B	P value
Plant Community Type	Modified-Tame	Graminoids	0.80	1.00	**0.001**
		Native Perennial Forbs	0.92	0.92	**0.001**
		Native Perennial Grasses	0.70	0.58	0.061
Forage Cover	Score 9	Native Perennial Forbs	0.73	0.63	**0.041**
Cover of Tall Productive Forages	Score 0	Native Ruderal Forbs	0.63	1.00	**0.021**
	Score 7 + 14	Graminoids	1.00	0.77	0.092
Hydraulic Function & Litter	Score 0	Ruderal Grasses	0.50	1.00	**0.022**
		Native Ruderal Forbs	0.41	1.00	0.064
Erosion	Score 4	Woody Species	0.78	0.23	**0.007**
Anthropogenic Bare Soil	Score 0	Native Ruderal Forbs	0.56	1.00	**0.017**
		Native Species	0.55	1.00	**0.019**
		Ruderal Grasses	0.65	1.00	**0.005**
Noxious Weed Density	Score 0	Introduced Ruderal Forbs	0.37	1.00	**0.024**
		Introduced Species	0.33	1.00	**0.031**
		Total Broad Leaf Plants	0.34	1.00	**0.049**
	Score 0 +1 + 5	Noxious Weeds	0.93	0.75	**0.028**
Woody Cover	Score 3	Native Perennial Forbs	0.75	0.79	**0.006**
Wood Density	Score 0	Woody Species	0.74	0.24	0.062
Grazing Intensity	U	Graminoids	0.54	0.75	0.085
		Total Grasses + Graminoids	0.28	1.00	0.083

A = Probability of occurring, B = Fidelity

Table C.7.4. Significant ANOVA effects of rangeland health criteria on the total seed density (seeds/m^2) of various major plant groups.

	Graminoids		Broad Leaf		Native		Introduced		Total	
Rangeland Health	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Plant Community Type	0.271	0.604	0.098	0.755	3.601	0.061	**7.856**	**0.006**	0.332	0.566
Forage Cover	0.371	0.691	0.068	0.934	0.295	0.745	0.759	0.471	0.086	0.918
Cover of Tall Productive Forages	1.302	0.277	2.474	0.089	3.013	0.054	0.335	0.717	1.166	0.316
Weedy & Ruderal Cover	0.785	0.378	1.129	0.291	0.090	0.765	0.401	0.528	0.157	0.693
Hydraulic Function & Litter	0.994	0.399	0.933	0.428	0.882	0.453	0.059	0.981	0.217	0.884
Soil Erosion	0.788	0.458	2.165	0.120	**6.115**	**0.003**	0.627	0.537	0.844	0.433
Anthropogenic Bare Soil	2.054	0.134	0.944	0.393	3.065	0.051	0.598	0.552	1.448	0.240
Noxious Weed Cover	0.432	0.650	0.969	0.383	0.661	0.519	1.746	0.180	1.214	0.301
Noxious Weed Density	0.401	0.753	1.184	0.320	0.482	0.696	2.690	0.051	1.788	0.155
Woody Spp Cover	0.035	0.526	0.309	0.580	1.228	0.271	**5.771**	**0.018**	1.389	0.241
Woody Spp Density	0.967	0.384	0.092	0.912	0.015	0.986	0.604	0.548	0.395	0.674
Grazing Intensity	1.009	0.417	1.254	0.290	0.184	0.968	0.926	0.468	1.039	0.400
Health	1.611	0.205	0.845	0.433	0.471	0.626	0.024	0.976	0.034	0.966

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$
*Includes only 58 sites from the 2013 survey

Table C.7.5. Mean (±SE) responses in total seed density (seeds/m^2) of the major plant groupings in relation to various rangeland health criteria.

Rangeland Health	**Treatment**	**Native**	**Introduced**
Plant Community Type	Modified-Tame	2770.4 (±491.8)	2363.3 (±875.4) b
	Tame	1468.8 (±179.6)	4615.3 (±319.7) a
Weedy & Ruderal Cover	0	3074.2 (±1220.1)	
	7	2259.8 (±418.5)	
	14	1456.3 (±189.4)	
Soil Erosion	4	1024.7 (±449.1) b	
	7	1902.4 (±271.6) a	
	10	1566.6 (±256.4) a	
Anthropogenic Bare Soil	0	3612.8 (±761.6)	
	3	1418.7 (±413.0)	
	5	1540.7 (±190.4)	
Noxious Weed Density	0		5570.7 (±527.4)
	1		3832.3 (±661.1)
	3		3853.0 (±544.1)
	5		3528.4 (±734.7)
Woody spp Cover	3		2779.7 (±816.5) b
	6		4600.2 (±325.7) a

Table C.7.6. Significant relationships between seed density (seeds/m^2) of specified plant functional groups and rangeland health scores.

	Native & Introduced				Introduced						Native									
	Legumes		Ruderal Grasses		Noxious Weeds		Ruderal Forbs		Seeded Graminoids		Ruderal Forbs		Perennial Forbs		Perennial Grasses		Graminoids		Woody Spp.	
Rangeland Health	X^2	P Value	F Value	P Value	X^2	P Value	F Value	P Value	F Value	P Value	F Value	P Value	X^2	P Value	X^2	P Value	X^2	P Value	X^2	P Value
Plant Community Type	0.414	0.520	0.183	0.670	0.979	0.322	**13.198**	**0.0004**	0.392	0.533	0.089	0.766	**16.127**	**0.0001**	3.504	0.061	3.643	0.056	0.054	0.817
Forage Cover	0.487	0.784	0.020	0.980	0.957	0.620	1.365	0.260	1.045	0.356	0.163	0.849	3.112	0.211	0.612	0.736	2.519	0.284	0.057	0.972
Cover of Tall Productive Forages	0.949	0.622	0.117	0.890	1.366	0.505	1.213	0.302	2.248	0.111	2.726	0.070	2.463	0.292	2.814	0.245	**7.157**	**0.028**	0.663	0.718
Weedy & Ruderal Cover	0.039	0.843	0.026	0.873	0.190	0.663	2.397	0.125	0.755	0.387	0.132	0.717	0.124	0.725	0.372	0.542	0.001	0.973	0.001	0.974
Hydraulic Function & Litter	0.066	0.996	1.013	0.390	1.409	0.703	0.709	0.549	1.534	0.211	2.086	0.107	4.703	0.195	4.474	0.215	0.401	0.493	1.711	0.634
Soil Erosion	**7.095**	**0.029**	**3.694**	**0.028**	0.539	0.764	0.159	0.853	1.965	0.146	**5.328**	**0.006**	1.842	0.398	1.911	0.385	1.311	0.519	**6.132**	**0.047**
Anthropogenic Bare Soil	1.018	0.601	**4.110**	**0.019**	0.688	0.709	0.965	0.384	2.936	0.058	1.529	0.222	3.360	0.186	5.853	0.054	2.010	0.366	2.158	0.340
Noxious Weed Cover	0.025	0.988	1.521	0.224	**6.345**	**0.042**	0.443	0.643	0.770	0.466	2.351	0.098	1.490	0.475	0.001	0.999	0.233	0.890	3.092	0.213
Noxious Weed Density	0.200	0.978	0.756	0.521	**10.038**	**0.018**	1.511	0.216	0.131	0.942	1.411	0.244	2.287	0.515	1.841	0.606	3.327	0.344	0.850	0.837
Woody spp Cover	0.274	0.600	1.427	0.235	0.461	0.497	**5.596**	**0.020**	0.407	0.525	0.213	0.645	**6.765**	**0.009**	1.408	0.235	0.594	0.441	0.259	0.611
Woody spp Density	1.312	0.519	2.062	0.133	0.262	0.877	1.401	0.251	0.751	0.475	0.569	0.568	1.669	0.434	0.399	0.819	1.386	0.500	5.768	0.056
Grazing Intensity	**11.626**	**0.040**	0.193	0.965	9.715	0.084	1.322	0.261	0.646	0.666	0.616	0.688	3.104	0.684	5.794	0.327	2.122	0.832	2.976	0.704
Health	3.695	0.158	0.461	0.632	1.877	0.391	0.559	0.574	3.004	0.054	0.896	0.412	0.382	0.826	1.969	0.374	0.833	0.659	1.393	0.498

Bold: $p < 0.05$, Black: $p < 0.1$, Grey: $p > 0.1$
*Includes only 58 sites from the 2013 survey
Note noxious weeds includes 1 graminoid species

Table C.7.7. Mean (±SE) seed density (seeds/m^2) of various plant functional groupings in relation to rangeland health scores.

		Native & Introduced		Introduced			Native				
Rangeland Health	**Score**	**Legumes**	**Ruderal Grasses**	**Noxious Weeds**	**Ruderal Forbs**	**Seeded Grasses**	**Ruderal Forbs**	**Perennial Forbs**	**Perennial Grasses**	**Graminoids**	**Woody Spp.**
Plant Community Type	Modified-Tame				1227.3 (±821.8) b			740.8 (±114.1) a	87.4 (±26.7)	834.1 (±196.7)	
	Tame				2940.2 (±300.1) a			68.3 (±41.7) b	36.8 (±9.8)	212.6 (±71.8)	
Cover of Tall Productive Forages	0						2824.0 (±665.7)			0.0 (±489.3) b	
	7						857.9 (±228.3)			670.1 (±167.8) a	
	14						835.0 (±103.3)			213.9 (±76.0) ab	
Soil Erosion	4	165.2 (±52.1) ab					565.6 (±249.7) b				17.5 (±4.3) a
	7	131.4 (±31.5) b					1056.1 (±151.0) a				2.9 (±2.6) b
	10	178.2 (±29.8) a					820.6 (±142.6) ab				2.1 (±2.4) b
Anthropogenic Bare Soil	0		929.4 (±206.4) a			962.8 (±639.5)			109.6 (±41.4)		
	3		284.6 (±111.9) b			807.5 (±346.8)			15.4 (±22.4)		
	5		342.3 (±51.6) b			1387.0 (±159.9)			44.4 (±10.3)		
Noxious Weed Cover	1			536.9 (±107.5) a			667.3 (±234.7)				
	3			111.1 (±53.7) b			999.1 (±119.1)				
	5			46.3 (±107.5) b			644.7 (±222.0)				
Noxious Weed Density	0			272.3 (±80.3) a							
	1			293.9 (±100.7) a							
	3			49.2 (±82.8) b							
	5			46.3 (±111.9) b							
Woody Spp Cover	3				1421.4 (±762.2) b			345.6 (±118.8) a			
	6				2948.3 (±304.0) a			115.9 (±47.4) b			
Woody Spp Density	0										10.2 (±3.7)
	2										0.0 (±6.0)
	4										3.6 (±2.0)
Grazing Intensity	U	268.1 (±98.3) a		6.0 (±237.3)							
	L	225.1 (±65.5) a		113.9 (±158.2)							
	LM	70.5 (±40.1) b		66.5 (±96.9)							
	M	142.3 (±33.7) ab		230.6 (±81.4)							
	MH	192.7 (±41.0) ab		219.7 (±99.0)							
	H	250.2 (±69.5) ab		241.3 (±167.8)							
Health	Healthy					1421.7 (±174.3)					
	Healthy with Problems					1058.6 (±256.2)					
	Unhealthy					357.5 (±713.2)					

Table C.7.8. Significant effects of various rangeland health criteria on similarity, as well as seed bank richness, diversity and evenness.

	Sørenson's Similarity		Seed Bank: Richness		Seed Bank: Shannon's Diversity		Seed Bank: Simpson's Diveristy		Seed Bank: Pielou's Evenness	
Rangeland Health	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**	**F Value**	**P Value**
Plant Community Type	0.233	0.630	0.007	0.931	0.286	0.594	0.206	0.651	0.000	0.985
Forage Cover	1.772	0.175	0.192	0.826	0.498	0.609	0.429	0.653	0.297	0.744
Cover of Tall Productive Forages	1.637	0.200	0.555	0.576	0.288	0.751	0.365	0.695	0.028	0.973
Weedy & Ruderal Cover	1.761	0.188	0.827	0.365	0.882	0.350	0.683	0.411	0.009	0.925
Hydraulic Function & Litter	**3.602**	**0.016**	0.355	0.785	0.851	0.469	0.795	0.499	0.443	0.723
Soil Erosion	1.872	0.159	1.506	0.227	2.671	0.074	2.195	0.117	0.444	0.643
Anthropogenic Bare Soil	2.276	0.108	0.112	0.894	0.267	0.766	0.379	0.686	0.077	0.926
Noxious Weed Cover	0.535	0.587	0.755	0.473	1.148	0.322	0.742	0.479	1.483	0.232
Noxious Weed Density	0.292	0.831	0.268	0.848	0.464	0.708	0.314	0.815	1.151	0.332
Woody Spp Cover	0.149	0.700	1.415	0.237	3.169	0.078	2.333	0.130	0.032	0.859
Woody Spp Density	0.684	0.507	1.143	0.323	0.540	0.585	0.515	0.599	0.879	0.418
Grazing Intensity	1.108	0.631	1.312	0.265	1.727	0.136	1.821	0.116	0.653	0.660
Health	1.655	0.196	0.000	1.000	0.532	0.589	0.737	0.481	0.443	0.644

Bold: P < 0.05, Black: P < 0.1, Grey: P > 0.1
*58 sites from the 2013 survey

Table C.7.9. Effect of management on the mean (±SE) richness, similarity, diversity, and evenness of the seed bank.

Rangeland Health	**Score**	**Sørenson's Similarity**	**Shannon's Diversity**
Hydraulic Function & Litter	0	0.485 (±0.047) a	
	8	0.375 (±0.019) ab	
	16	0.390 (±0.017) ab	
	25	0.344 (±0.014) b	
Woody Spp Cover	3		2.3 (±0.1)
	6		2.0 (±0.1)

Appendix D. Chapter 6.

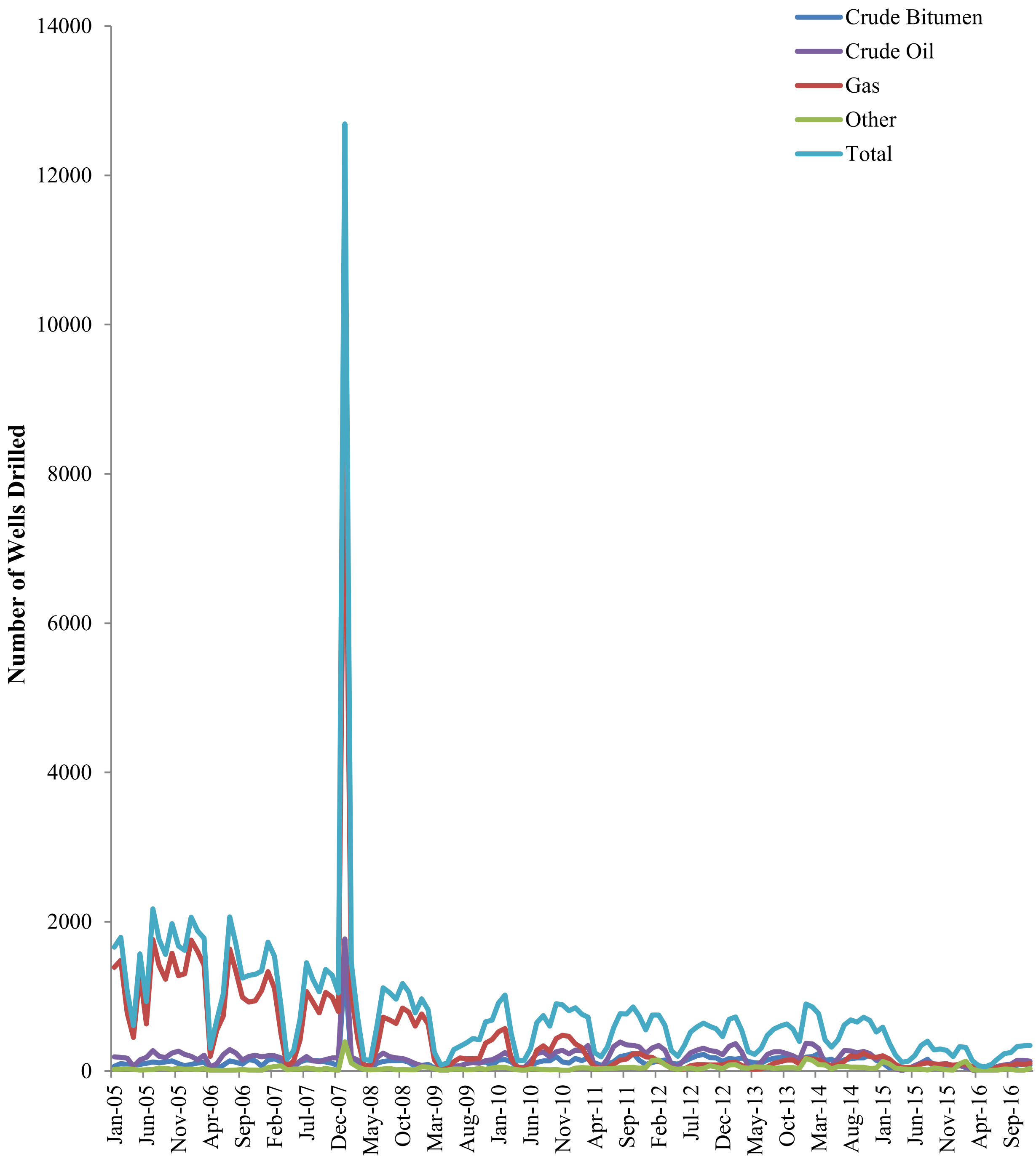


Figure D.1. Records of the number of successfully drilled wells in Alberta since January 2005 through to December 2016. These data are available to the public through the Alberta Energy Regulator (AER). Provincial drilling activity declined after autumn of 2014.

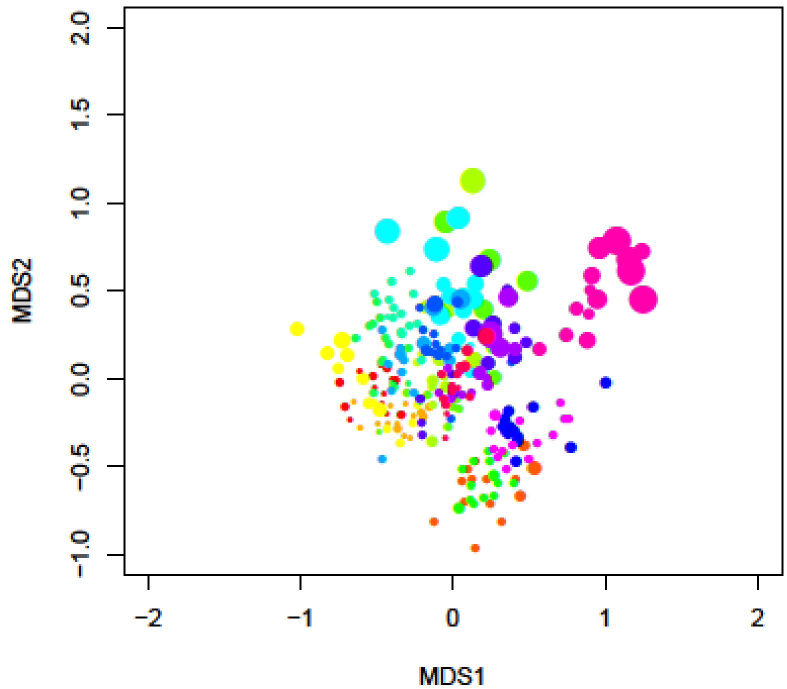


Figure D.2. NMDS ordination of aboveground plant community composition, colour coded by site (stress = 0.23, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater introduced species cover.

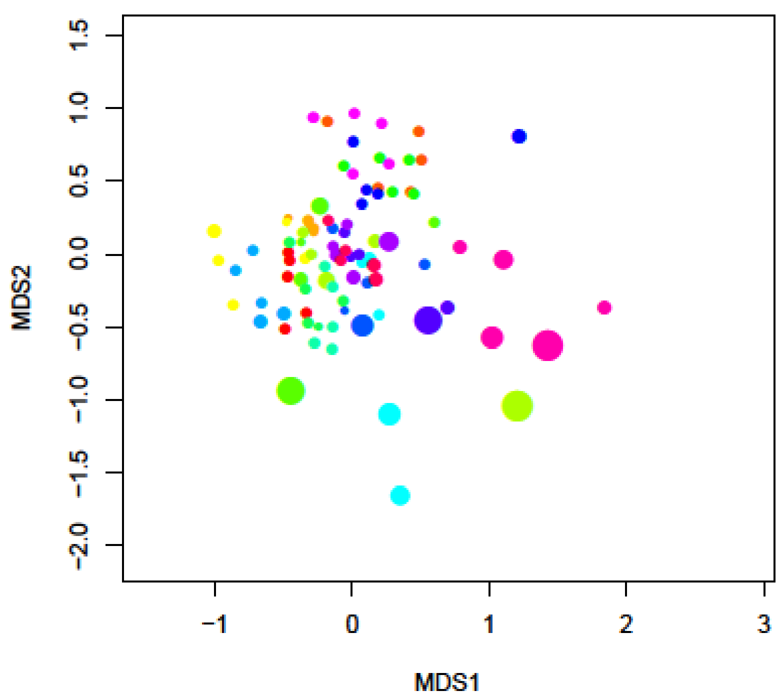


Figure D.3. NMDS ordination of aboveground plant community biomass clipped by individual species, colour coded by site (stress = 0.22, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater introduced species biomass.

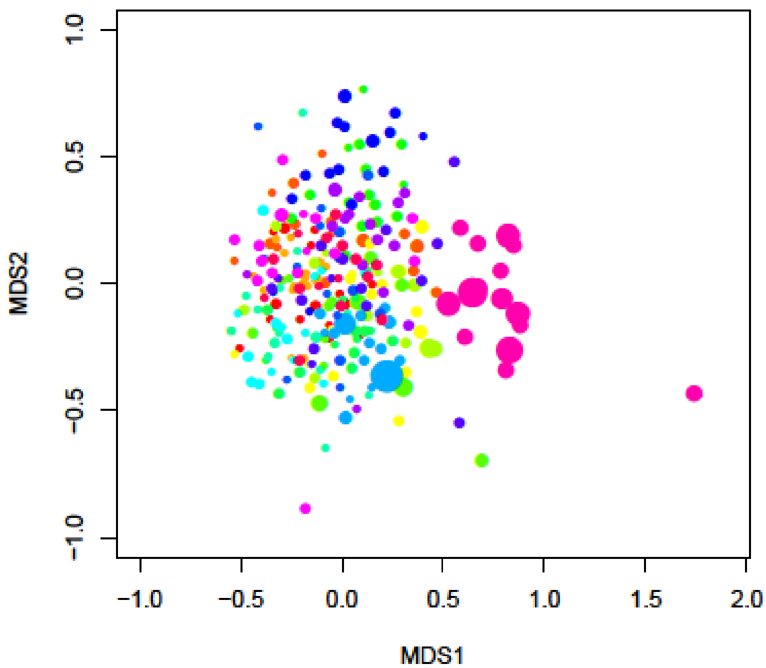


Figure D.4. NMDS ordination of seed bank composition, colour coded by site (stress = 0.28, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater introduced species seed density.

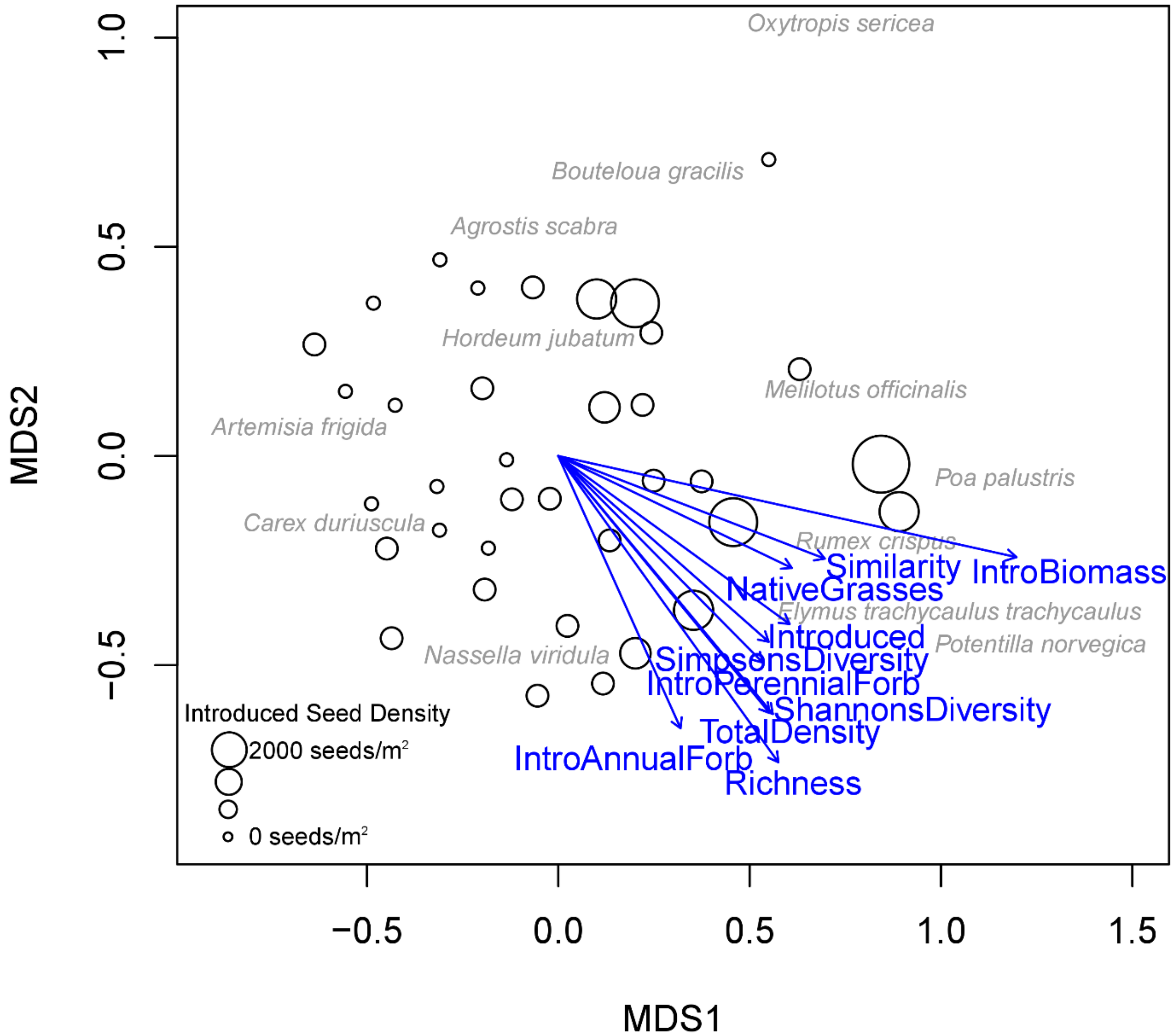


Figure D.5. NMDS ordination of seed bank composition along pipeline trenches (soil from pipeline center and edge) (stress = 0.24, dimensions = 2, distance = Bray-Curtis). Overlaid vectors represent significant seed bank characteristics and pipeline attributes (diameter and age) ($P < 0.05$). Note, no pipeline attributes were significant. Larger symbols indicate greater introduced species seed density.

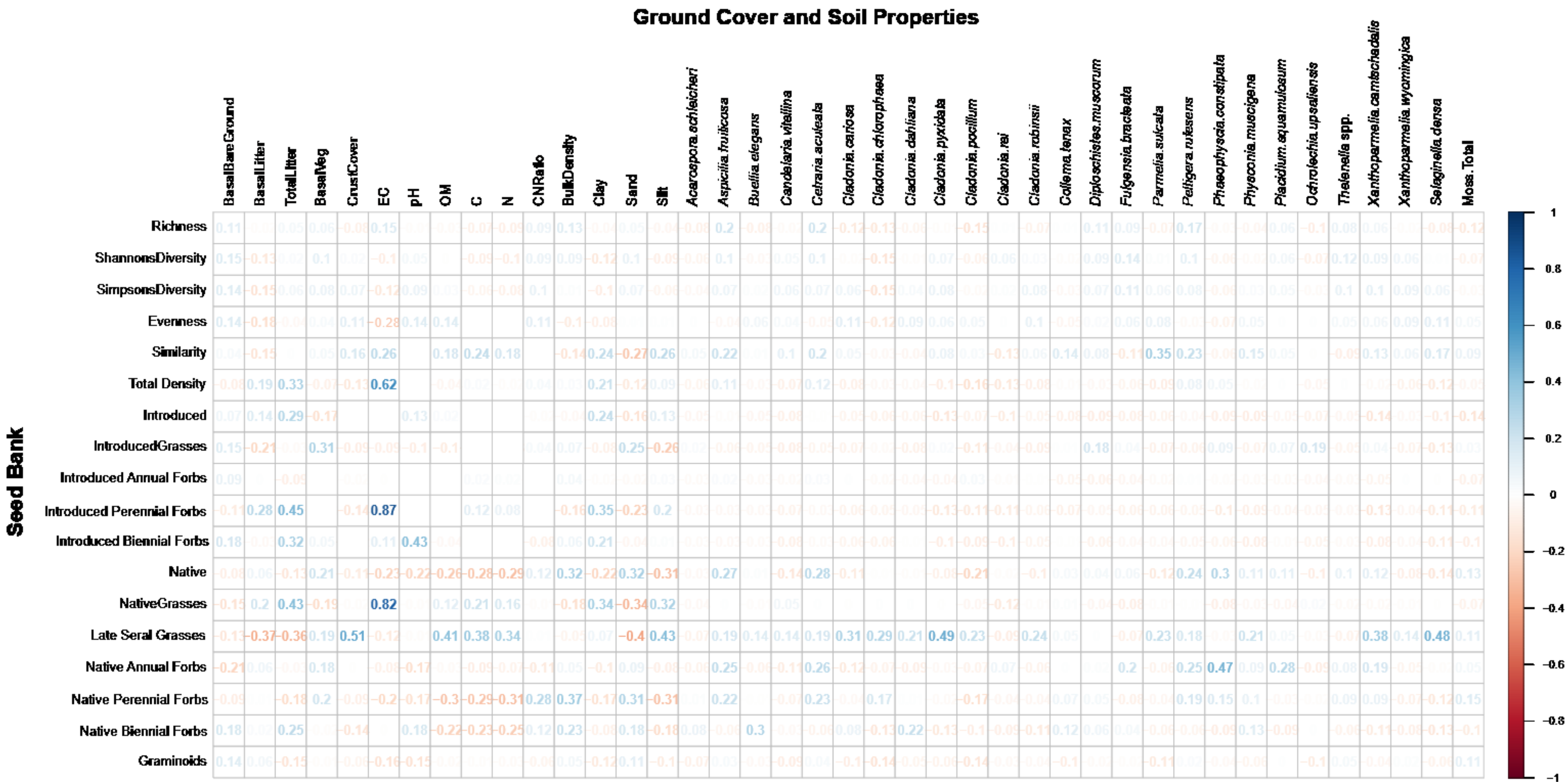


Figure D.6. Correlation coefficients (r) for the relationship between seed bank characteristics and both soil properties and ground cover including lichen species.

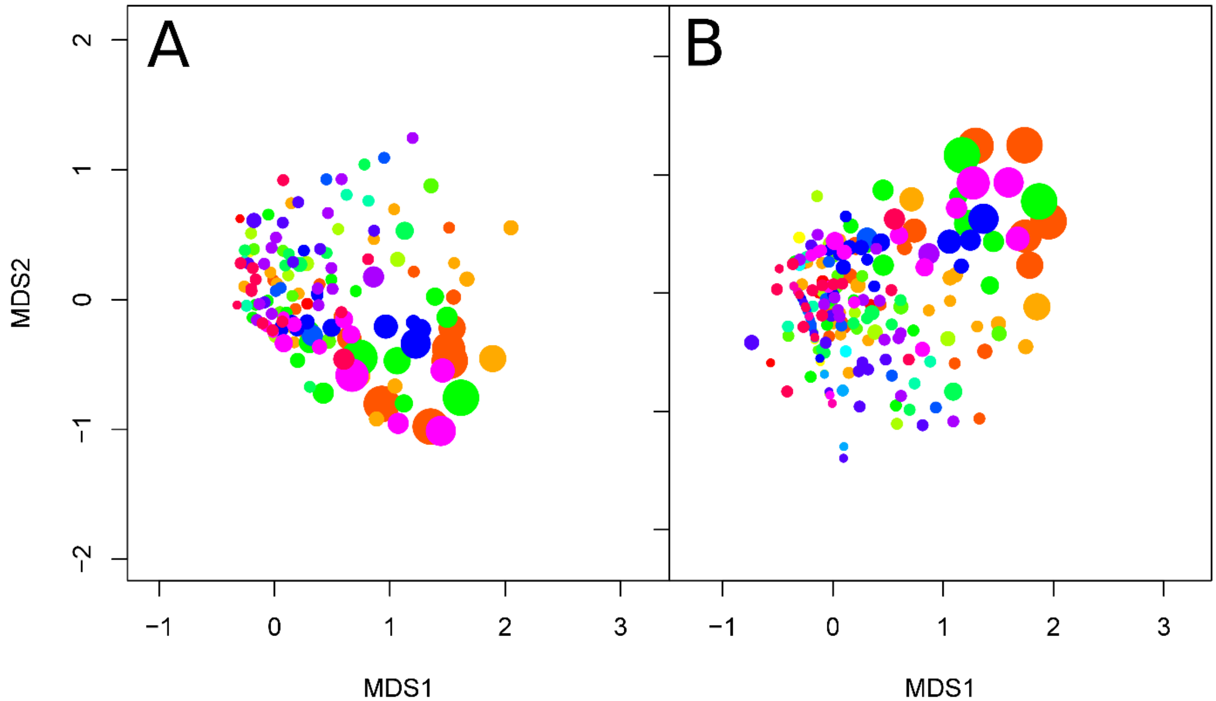


Figure D.7. NMDS ordination of biological soil crust composition, colour coded by site. Panel A is biological soil crust composition (stress = 0.13, dimensions = 2, distance = Bray-Curtis), including lichens, mosses, and spike-mosses. Panel B describes dynamics in ground cover where biological crust composition was analysed with the inclusion of the proportion of bare ground and litter cover (stress = 0.14, dimensions = 2, distance = Bray-Curtis). Larger symbols indicate greater biological soil crust cover.

Table D.1. Descriptions of pipelines surveyed at the Mattheis Research Ranch between 2013 and 2015.

Site	Pipeline Diameter (mm)	Pipeline Permit Date	Pipeline License Date	Connecting Well ID	Well Final Drill Date	Construction Date used in Analysis	Licensee
1	60.3	Apr 1995	-	159935	Sept 1993	1995	Cenovus Energy Inc.
2	60.3	Sept 1976	April 1989	58280	June 1976	1976	Cenovus Energy Inc.
3	60.3	Sept 1976	Aug 1977	237057	June 2000	1977	Cenovus Energy Inc.
4	1067	-	-	-	-	1982*	Foothills Pipe Lines Ltd.
5	60.3	Jan 2003	Jan 2004	274478	Dec 2002	2003	Cenovus Energy Inc.
6	168.3	-	-	-	-	1971*	Nova Gas Transmission Ltd.
7	88.9	Sept 1976	Aug 1977	52472	Oct 1975	1977	Cenovus Energy Inc.
8	88.9	-	-	-	-	1977*	Nova Gas Transmission Ltd.
9	88.9	March 1994	March 1995	-	-	1994	Canadian Natural Resources Limited
10	219.1	-	-	-	-	1960*	Nova Gas Transmission Ltd.
11	60.3	Jan 2003	Jan 2004	275322	Dec 2002	2003	Cenovus Energy Inc.
12	60.3	Sept 1976	Aug 1977	365478	Feb 2007	1977	Cenovus Energy Inc.
13	60.3	Dec 2006	Dec 2007	365982	Feb 2007	2007	Cenovus Energy Inc.
14	60.3	Nov 1998	Jan 2008	215578	1998	1998	Enerplus Corporation
15	88.9	-	Jan 2008	261601	Nov 2001	2001	Canadian Natural Resources Limited
16	60.3	Jan 2003	Jan 2004	274971	Dec 2002	2003	Cenovus Energy Inc.
17	88.9	-	Oct 1981	79778	Feb 1980	1981	Cenovus Energy Inc.
18	60.3	Nov 1998	Jan 2008	317636	June 1998	1998	Enerplus Corporation

*Inferred from the registration of permitted encumbrances

Table D.2. Biplot vector scores obtained from the NMDS ordination of plant community composition, for various pipeline, plant community, ground cover, and soil properties. Factors significant at $P < 0.05$ were included in the ordination.

Characteristics	**Factor**	**MDS1**	**MDS2**	**R^2**	**P value**
Pipeline	Age	-0.70	0.71	0.01	0.402
	Distance	-0.28	-0.96	0.03	**0.018**
	Diameter	-1.00	0.09	0.10	**0.001**
Plant Community	Introduced Cover	0.52	0.86	0.56	**0.001**
	Native Cover	-0.54	-0.84	0.28	**0.001**
	Richness	0.74	0.67	0.20	**0.001**
	Shannon's Diversity	0.86	0.50	0.37	**0.001**
	Simpson's Diversity	0.94	0.35	0.33	**0.001**
	Sorenson's Similarity	0.23	-0.97	0.04	**0.004**
	Evenness	0.92	0.39	0.24	**0.001**
	Total Biomass	-0.03	1.00	0.49	**0.001**
	Native Biomass	-0.40	0.92	0.24	**0.001**
	Introduced Biomass	0.38	0.92	0.50	**0.001**
Ground Cover	Bare Ground	-0.47	0.89	0.05	0.098
	Biological Soil Crust	0.28	-0.96	0.28	**0.001**
	Litter Biomass	0.25	0.97	0.48	**0.001**
	Litter Cover	-0.16	0.99	0.21	**0.001**
	Manure	0.92	-0.39	0.00	0.835
	Rocks	0.57	-0.82	0.16	**0.001**
	Stems	0.02	-1.00	0.14	**0.004**
Soil Properties	Bulk Density	-0.97	0.23	0.47	**0.001**
	C	0.76	-0.65	0.67	**0.001**
	C:N Ratio	-0.95	0.31	0.25	**0.001**
	EC	0.88	0.48	0.41	**0.001**
	N	0.76	-0.64	0.67	**0.001**
	OM	0.71	-0.70	0.59	**0.001**
	pH	0.30	0.95	0.05	0.118
Texture	Clay	0.97	-0.25	0.17	**0.043**
	Sand	-0.77	0.63	0.62	**0.001**
	Silt	0.75	-0.66	0.65	**0.001**

Table D.3. Biplot vector scores of significant plant species associated with the NMDS ordination of plant community composition. Only species significant at P < 0.001 were plotted.

Species	MDS1	MDS2	R^2	P value
Achillea millefoliium	0.98	0.18	0.05	**0.002**
Agropyron cristatum	-0.49	0.87	0.11	**0.001**
Agrostis scabra	0.99	0.16	0.21	**0.001**
Allium textile	0.99	0.11	0.00	0.798
Androsace septentrionals	0.90	-0.45	0.03	**0.034**
Antennaria parvifolia	0.97	-0.24	0.03	**0.044**
Arabis holboellii ssp. *retrofracta*	-0.38	-0.92	0.01	0.475
Arnica	0.84	0.54	0.04	**0.009**
Artemisia campestris	-0.83	0.56	0.04	**0.007**
Artemisia frigida	-0.24	-0.97	0.11	**0.001**
Artemisia ludoviciana	-0.63	0.77	0.16	**0.001**
Astragalus agrestis	0.82	-0.58	0.10	**0.001**
Astragalus cicer	-0.16	-0.99	0.03	**0.015**
Astragalus pectinatus	-0.94	0.34	0.01	0.307
Astragalus striatus	0.00	0.00	0.00	1.000
Atriplex subspicata	-0.01	1.00	0.01	0.270
Botrychium campestre	0.88	0.47	0.01	0.417
Bouteloua gracilis	0.04	-1.00	0.44	**0.001**
Bromus inermis ssp. *inermis*	0.07	1.00	0.08	**0.002**
Bromus inermis ssp. *pumpellianus*	0.00	0.00	0.00	1.000
Calamovilfa longifolia	-0.88	0.47	0.46	**0.001**
Campanula rotundifolia	-0.29	0.96	0.04	**0.008**
Carex duriuscula	-1.00	-0.09	0.12	**0.001**
Carex pensylvanica	-0.67	0.74	0.10	**0.001**
Carex praegracilis	0.00	0.00	0.00	1.000
Cerastium arvense	-0.48	0.88	0.01	0.193
Chamerhodos erecta	0.74	-0.67	0.07	**0.001**
Chenopodium album	-1.00	0.06	0.06	**0.003**
Chenopodium pratericola	0.18	0.98	0.04	**0.007**
Cirsium flodmanii	0.51	0.86	0.06	**0.001**
Cirsiumarvense	0.79	0.61	0.10	**0.001**
Cleome serrulata	-1.00	-0.04	0.01	0.198
Comandra umbellata	0.74	0.68	0.00	0.878
Conyza Canadensis	-0.01	-1.00	0.00	0.880
Crepis tectorum	0.93	0.37	0.08	**0.001**
Dalea purpurea	-0.86	0.50	0.01	0.152
Descurainia Sophia	-0.87	0.50	0.00	0.843
Distichlis stricta	0.86	0.50	0.39	**0.001**
Draba nemorosa	0.85	-0.53	0.02	0.059
Eleocharis palustris	1.00	0.04	0.07	**0.002**
Elymus junceus	0.19	-0.98	0.03	**0.035**
Elymus lanceolatus	0.41	-0.91	0.25	**0.001**
Elymus trachycaulus ssp. *subsecundus*	0.89	-0.45	0.00	0.620
Elymus trachycaulus ssp. *trachycaulus*	0.88	-0.47	0.07	**0.001**
Elytrigia repens	0.12	0.99	0.19	**0.001**
Equisetum laevigatum	-0.89	0.46	0.29	**0.001**
Erysimum cheiranthoidies	0.73	0.69	0.00	0.586
Erysimum inconspicuum	-0.67	-0.74	0.00	0.922
Escobaria viviparia	0.64	-0.77	0.00	0.788
Festuca ovina	0.56	0.83	0.03	**0.014**
Fragaria virginiana	0.00	0.00	0.00	1.000
Gaillardia aristata	0.43	-0.90	0.06	**0.001**
Gaura coccinea	0.63	-0.78	0.22	**0.001**
Geum triflorum	0.00	0.00	0.00	1.000
Glycyrrhiza lepidota	-0.71	0.71	0.12	**0.001**
Grindella squarrosa	0.60	-0.80	0.07	**0.002**
Guterrhiza sarothrae	-0.12	-0.99	0.02	0.064
Haplopappus spinulosus	0.35	-0.94	0.13	**0.001**
Hedeoma hispida	0.87	0.49	0.01	0.445
Helictotrichon hookerii	-0.34	0.94	0.01	0.178
Hesperostipa comata	-0.64	-0.77	0.01	0.210
Heterotheca villosa	-1.00	0.08	0.16	**0.001**
Hordeum jubatum	0.84	0.54	0.38	**0.001**
Juncus balticus	-0.65	0.76	0.18	**0.001**

Kochia scoparia	0.04	-1.00	0.02	0.094
Koeleria macrantha	0.28	-0.96	0.22	**0.001**
Lactuca pulchella	0.81	0.58	0.09	**0.001**
Lappula squarrosa	0.43	0.90	0.04	**0.007**
Lepidium densiflorum	0.09	1.00	0.02	0.115
Liatris punctata	-0.99	0.12	0.07	**0.002**
Linium rigidum	-0.04	-1.00	0.03	**0.022**
Lithospermum insisum	-0.97	0.23	0.21	**0.001**
Lygodesmia juncea	-0.21	0.98	0.02	0.075
Medicago sativa	0.12	-0.99	0.03	**0.035**
Melilotus alba	0.55	0.83	0.14	**0.001**
Melilotus officinalis	0.45	0.89	0.09	**0.001**
Mirabilis hirsuta	0.83	-0.55	0.00	0.953
Muhlenbergia cuspidata	0.26	-0.97	0.02	0.099
Nassella viridula	0.93	-0.38	0.12	**0.001**
Oenethera nuttallii	-0.89	0.45	0.02	**0.038**
Opuntia fagilis	0.96	0.28	0.00	0.831
Opuntia polycantha	-0.69	-0.73	0.01	0.279
Orobanche fasciculata	-0.85	-0.53	0.03	**0.025**
Orthocarpus luteus	0.29	-0.96	0.02	0.079
Oxytropis sericea	0.21	-0.98	0.03	**0.016**
Paronychia sessiliflora	0.75	-0.66	0.02	0.054
Pascopyrum smithii	0.95	-0.31	0.20	**0.001**
Penstemon gracilis	0.54	0.84	0.02	0.072
Phlox hoodii	0.07	-1.00	0.07	**0.001**
Plantago major	0.80	0.60	0.05	**0.004**
Plantago pataonica	0.44	-0.90	0.03	**0.022**
Poa palustris	0.85	0.53	0.21	**0.001**
Poa pratensis	0.40	0.92	0.39	**0.001**
Poa secunda	-0.24	-0.97	0.02	0.075
Polygonum aviculaire	0.49	-0.87	0.01	0.304
Polygonum convolvulus	0.48	0.88	0.00	0.961
Potentilla arguta	0.72	-0.70	0.00	0.636
Potentilla pensylvanica	0.71	-0.70	0.03	**0.020**
Psoralea lanceolata	-0.59	0.81	0.06	**0.002**
Pulsatilla patens	0.00	0.00	0.00	1.000
Ratibida columnifera	0.33	-0.94	0.12	**0.001**
Rosa arkansana	-0.71	0.71	0.23	**0.001**
Rumex crispus	0.79	0.62	0.07	**0.001**
Salsola pestifer	-0.95	0.31	0.03	**0.021**
Schedonnardus paniculatus	0.40	-0.92	0.05	**0.003**
Selaginella densa	0.36	-0.93	0.20	**0.001**
Shepherdia argentea	0.09	1.00	0.04	**0.021**
Silene drumondii	0.43	0.90	0.01	0.442
Sisyrinchium montanum	0.76	0.65	0.01	0.495
Solidago missouriensis	-0.52	-0.85	0.03	**0.019**
Sonchus arvensis	0.82	0.57	0.41	**0.001**
Spartina gracilis	0.80	0.60	0.00	0.681
Sphaeralcea coccinea	0.41	-0.91	0.20	**0.001**
Sporobolus cryptandrus	-0.95	0.32	0.03	**0.045**
Symphoricarpos occidentalis	-0.05	1.00	0.08	**0.001**
Symphyotrichum ericoidies	-0.69	-0.72	0.00	0.586
Symphyotrichum laevis	0.00	0.00	0.00	1.000
Taraxacum officinale	0.95	-0.32	0.29	**0.001**
Thermopsis rhombifolia	-0.06	1.00	0.07	**0.001**
Thinopyrum intermedium	0.77	0.64	0.01	0.237
Tragopogon dubius	1.00	0.07	0.10	**0.001**
Vicia americana	0.36	0.93	0.03	**0.015**
Vulpia octiflora	0.96	0.29	0.00	0.672

Table D.4. Biplot vector scores obtained from the NMDS ordination of individual plant species biomass (clipped by species), in relation to pipeline, plant community, ground cover, and soil properties, across 18 pipeline study sites. Factors significant at P < 0.05 were included in the ordination.

Characteristic	Factor	MDS1	MDS2	R^2	P value
Pipeline	Age	0.44	-0.90	0.05	0.105
	Distance	-0.46	0.89	0.02	0.400
	Diameter	-0.98	-0.22	0.11	**0.014**
Plant Community	Richness	0.91	-0.40	0.29	**0.001**
	Shannon's Diversity	0.92	-0.38	0.15	**0.001**
	Simpson's Diversity	1.00	-0.03	0.15	**0.001**
	Evenness	0.97	-0.23	0.23	0.376
	Native	-0.03	-1.00	0.23	**0.001**
	Introduced	0.65	-0.76	0.49	**0.001**
Ground Cover	Bare Ground	-0.39	-0.92	0.02	0.386
	Biological Soil Crust	0.16	0.99	0.23	**0.001**
	Litter Biomass	0.49	-0.87	0.58	**0.001**
	Litter Cover	0.00	-1.00	0.21	**0.001**
	Manure	-0.31	0.95	0.00	0.972
	Rocks	0.33	0.94	0.13	**0.003**
	Stems	-0.23	0.97	0.15	**0.001**
Soil Properties	Bulk Density	-0.90	-0.44	0.36	**0.001**
	C	0.60	0.80	0.58	**0.001**
	C:N Ratio	-0.94	-0.34	0.15	**0.003**
	EC	0.97	-0.23	0.42	**0.001**
	N	0.59	0.81	0.59	**0.001**
	OM	0.53	0.85	0.54	**0.001**
	pH	0.72	-0.69	0.05	0.078
Texture	Clay	0.87	0.48	0.11	0.145
	Sand	-0.62	-0.79	0.58	**0.001**
	Silt	0.59	0.81	0.63	**0.001**

Table D.5. Biplot vector scores associated with the NMDS ordination of individual plant biomass (clipped by species) along 18 pipeline study sites. Only species significant at $P < 0.05$ were plotted.

Species	MDS1	MDS2	R^2	P value
Achillea millefolium	0.92	-0.39	0.11	**0.025**
Achnatherum hymenoides	-0.58	0.82	0.00	0.844
Agropyron cristatum	0.23	-0.97	0.18	**0.005**
Agrostis scabra	0.91	0.41	0.08	**0.025**
Allium textile	-0.20	-0.98	0.02	0.321
Amaranthus retroflexus	-0.99	0.12	0.04	0.178
Androsace septentrionalis	0.97	0.23	0.04	0.175
Antennaria parvifolia	0.98	-0.18	0.16	**0.005**
Arabis holbellii ssp. *retrofracta*	0.97	-0.23	0.00	0.907
Artemisia campestris	-1.00	0.01	0.03	0.290
Artemisia ludoviciana	-0.21	-0.98	0.07	0.052
Astragalus agrestis	0.90	0.43	0.05	0.104
Astragalus cicer	0.70	-0.72	0.13	**0.028**
Astragalus crasiocarpus	0.88	-0.48	0.00	0.906
Astragalus striatus	0.94	0.33	0.01	0.651
Bouteloua gracilis	-0.17	0.98	0.38	**0.001**
Calamovilfa longifolia	-0.73	-0.68	0.39	**0.001**
Campanula rotundifolia	-0.39	-0.92	0.02	0.310
Carex douglasii	-0.12	-0.99	0.00	0.854
Carex duriuscula	-0.99	0.14	0.23	**0.001**
Carex filifolia	-0.03	1.00	0.12	**0.010**
Carex pensylvanica	-0.55	-0.84	0.11	**0.007**
Carex praegracilis	-0.31	-0.95	0.03	0.253
Cerastium arvense	-0.29	-0.96	0.03	0.265
Chamaerhodos erecta	0.42	0.91	0.05	0.089
Chenopodium album	0.96	-0.27	0.15	**0.016**
Chenopodium pratericola	0.56	-0.83	0.04	0.183
Cirsium arvense	0.95	-0.32	0.17	**0.005**
Cirsium flodmanii	0.23	-0.97	0.18	**0.002**
Conyza canadensis	0.92	-0.39	0.05	**0.090**
Crepis tectorum	0.96	-0.29	0.06	0.061
Dalea purpurea	-0.93	-0.36	0.04	0.215
Descurainia sophia	0.59	-0.81	0.03	0.263
Distichlis stricta	0.96	-0.27	0.25	**0.001**
Elymus junceus	-0.42	-0.91	0.03	0.216
Elymus lanceolatus	0.09	1.00	0.18	**0.002**
Elymus trachycaulus ssp. *subsecundus*	0.16	-0.99	0.00	0.916
Elymus trachycaulus ssp. *trachycaulus*	0.83	0.56	0.12	**0.019**
Elytrigia repens	0.04	-1.00	0.08	**0.045**
Equisetum laevigatum	-0.03	-1.00	0.17	**0.004**
Erigeron glabellus ssp. *pubescens*	-0.99	-0.16	0.03	0.223
Erysimum capitatum	-0.13	-0.99	0.01	0.684
Erysimum inconspicuum	-0.99	0.13	0.05	0.120
Escobaria vivipara	-0.42	-0.91	0.01	0.615
Euphorbia serpyllifolia	0.45	0.89	0.05	0.132
Fesctuca ovina	0.79	0.61	0.10	0.510
Gaura coccinea	0.99	-0.10	0.12	**0.010**
Glycyrrhiza lepidota	0.39	-0.92	0.11	**0.015**
Grindelia squarrosa	0.30	0.95	0.11	**0.016**
Haplopappus spinulosus	0.34	0.94	0.04	0.167
Hedeoma hispida	0.82	0.58	0.10	**0.030**
Hesperostipa comata	-0.60	-0.80	0.05	0.136
Heterotheca villosa	-1.00	0.06	0.13	**0.006**
Hordeum jubatum	0.87	-0.49	0.16	**0.002**
Juncus balticus	-0.57	-0.82	0.14	**0.002**
Koeleria macrantha	0.22	0.98	0.20	**0.001**
Lactuca pulchella	0.90	-0.44	0.14	**0.005**
Lactuca serriola	0.77	-0.64	0.10	**0.027**
Lepidium densiflorum	0.96	-0.27	0.02	0.405
Liatris punctata	-0.88	-0.48	0.11	**0.011**
Linum rigidum	-0.17	0.99	0.04	0.130
Lithospermum incisum	-0.56	-0.83	0.05	0.120
Lygodesmia juncea	-0.61	0.79	0.00	0.822

Medicago sativa	-0.74	-0.68	0.02	0.312
Melilotus alba	0.36	-0.93	0.07	0.065
Melilotus officinalis	-0.34	-0.94	0.04	0.175
Mirabilis hirsuta	0.21	-0.98	0.06	0.061
Nassella viridula	0.99	-0.17	0.10	**0.029**
Oenothera nuttallii	-0.84	-0.55	0.02	0.414
Opuntia fragilis	-0.29	-0.96	0.08	**0.038**
Pascopyrum smithii	0.96	0.30	0.29	**0.001**
Penstemon gracilis	0.93	0.37	0.02	0.377
Phlox hoodii	-0.05	1.00	0.13	**0.009**
Plantago major	-0.90	-0.44	0.00	0.852
Plantago patagonica	0.99	-0.16	0.01	0.519
Poa compressa	1.00	-0.03	0.05	0.085
Poa palustris	0.97	-0.23	0.16	**0.015**
Poa pratensis	0.71	-0.71	0.31	**0.001**
Poa secunda	0.97	0.25	0.04	0.189
Polygonum aviculare	0.90	-0.45	0.11	**0.044**
Potentilla arguta	0.56	0.83	0.03	0.208
Potentilla norvegica	0.97	-0.23	0.16	**0.015**
Potentilla pensylvanica	0.86	0.51	.0323.	0.217
Psoralidium lanceolatum	-0.30	-0.95	0.03	0.252
Ratibida columnifera	0.64	0.77	0.23	**0.002**
Rosa acicularis	-0.44	-0.90	0.12	**0.008**
Rosa arkansana	-0.76	-0.65	0.03	0.237
Schedonnardus paniculatus	0.47	0.88	0.11	**0.012**
Silene drumondii	0.67	0.75	0.02	0.430
Solidago canadensis	0.99	-0.15	0.19	**0.004**
Solidago missouriensis	-0.87	0.50	0.04	0.175
Sonchus arvensis	0.94	-0.33	0.41	**0.001**
Sphaeralcea coccinea	0.49	0.87	0.13	**0.005**
Sporobolus cryptandrus	-0.99	0.10	0.09	**0.023**
Symphiotrichum ericoides	0.78	-0.62	0.22	**0.001**
Symphoricarpos occidentalis	0.20	-0.98	0.21	**0.001**
Taraxacum officinale	1.00	0.07	0.23	**0.002**
Thermopsis rhombifolia	0.54	-0.84	0.04	0.157
Thinopyrum intermedium	0.79	0.61	0.10	**0.045**
Tragopogon dubius	0.98	0.18	0.05	0.121
Vicia americana	-0.17	-0.99	0.00	0.886

Table D.6. Biplot vector scores associated with the NMDS ordination of seed bank composition, in relation to pipeline, seed bank, plant community, ground cover, and soil properties, for 18 pipeline study sites. Factors significant at P < 0.05 were included in the ordination.

Characteristic	Factor	MDS1	MDS2	R^2	P value
Pipeline	Age	0.00	-1.00	0.02	**0.038**
	Distance	-0.69	0.73	0.02	0.122
	Diameter	0.37	-0.93	0.03	**0.030**
Seed Bank	Total Density	0.99	-0.17	0.32	**0.001**
	Native	0.00	-1.00	0.01	0.166
	Introduced	0.99	-0.14	0.32	**0.001**
	Introduced Grass	0.45	-0.89	0.08	**0.001**
	Native Grass	0.99	0.10	0.33	**0.001**
	Introduced Annual Forb	0.98	0.18	0.01	0.198
	Introduced Perennial Forb	0.99	-0.10	0.33	**0.001**
	Introduced Biennial Forb	0.84	-0.55	0.10	**0.001**
	Native Annual Forb	0.75	0.67	0.01	0.341
	Native Perennial Forb	-0.59	-0.81	0.15	**0.001**
	Native Biennial Forb	0.76	-0.65	0.10	**0.001**
	Graminoids	0.80	0.61	0.02	0.117
	Richness	1.00	-0.04	0.32	**0.001**
	Shannon's Diversity	0.97	0.25	0.17	**0.001**
	Simpson's Diversity	0.92	0.40	0.13	**0.001**
	Sorenson's Similarity	0.74	0.68	0.03	**0.019**
	Evenness	0.24	0.97	0.02	0.095
Plant Community	Total Biomass	0.50	-0.87	0.19	**0.001**
	Native Biomass	0.05	-1.00	0.07	**0.042**
	Introduced Biomass	0.74	-0.67	0.23	**0.001**
Ground Cover	Bare Ground	0.54	-0.84	0.07	**0.039**
	Biological Soil Crust	-0.53	0.85	0.10	**0.015**
	Litter Biomass	0.90	-0.44	0.23	**0.001**
	Litter Cover	0.76	-0.65	0.03	0.249
	Manure	-0.97	0.25	0.02	0.341
	Rocks	-0.46	0.89	0.02	0.360
	Stems	-0.94	0.34	0.04	0.156
Soil Properties	Bulk Density	-0.47	-0.88	0.20	**0.001**
	C	0.23	0.97	0.31	**0.001**
	C:N Ratio	-0.19	0.98	0.04	0.149
	EC	0.98	-0.18	0.42	**0.001**
	N	0.19	0.98	0.29	**0.001**
	OM	0.19	0.98	0.34	**0.001**
	pH	0.80	-0.60	0.08	**0.027**
Texture	Clay	0.80	0.60	0.04	0.470
	Sand	-0.25	-0.97	0.26	**0.008**
	Silt	0.21	0.98	0.29	**0.004**

Table D.7. Biplot vector scores associated with the NMDS ordination of seed bank composition, from 18 pipeline study sites. Only species significant at P < 0.05 were plotted.

Species	MDS1	MDS2	R^2	P value
Agropyron cristatum	0.44	-0.90	0.11	0.210
Agrostis scabra	0.44	0.90	0.02	**0.049**
Amaranthus blitoidies	-0.50	0.87	0.01	0.285
Amaranthus retroflexus	-0.78	0.62	0.02	0.078
Androsace septentrionalis	-0.17	0.99	0.01	0.423
Antennaria parvifolia	-0.42	0.91	0.00	0.649
Arabis holboellii ssp. *retrofracta*	0.49	0.87	0.00	0.802
Artemisia campestris	0.23	-0.97	0.03	**0.035**
Artemisia frigida	-0.78	-0.63	0.23	**0.001**
Artemisia ludoviciana	0.63	-0.77	0.00	0.516
Astragalus agrestis	0.58	-0.82	0.01	0.368
Atriplex subspicata	0.98	-0.18	0.03	**0.036**
Bouteloua gracilis	-0.28	0.96	0.16	**0.001**
Bromus inermis ssp. *inermis*	-0.82	-0.57	0.01	0.137
Calamagrostis montanensis	0.20	0.58	0.02	0.091
Calamovilfa longifolia	0.37	-0.93	0.01	0.482
Campanula rotundifolia	-0.25	-0.97	0.02	0.094
Capsella bursa-pastoris	0.96	0.26	0.01	0.310
Carex duriuscula	0.42	0.91	0.06	**0.001**
Carex pensylvanica	-0.25	0.97	0.02	0.088
Cerastium arvense	-0.92	-0.39	0.00	0.852
Chaenorhinum minus	0.96	0.29	0.03	**0.041**
Chamaerhodos erecta	0.63	0.77	0.01	0.299
Chenopodium album	0.42	-0.91	0.01	0.183
Chenopodium capitatum	0.95	-0.33	0.00	0.858
Chenopodium gigantospermum	-0.81	0.59	0.01	0.470
Chenopodium pratericola	0.55	-0.84	0.01	0.250
Cirsium flodmanii	0.71	-0.70	0.01	0.392
Conyza canadensis	1.00	-0.04	0.14	**0.001**
Crepis tectorum	0.35	0.94	0.17	**0.001**
Descurainia sophia	0.04	-1.00	0.02	**0.040**
Distichlis stricta	-1.00	-0.05	0.00	0.640
Draba nemorosa	0.53	-0.85	0.01	0.459
Elymus lanceolatus	0.84	-0.55	0.01	0.258
Elymus trachycaulus ssp. *subsecundus*	-0.20	0.98	0.01	0.242
Elymus trachycaulus ssp. *trachycaulus*	0.84	-0.54	0.03	**0.029**
Elytrigia repens	0.66	-0.75	0.01	0.353
Epilobium ciliatum	-0.16	-0.99	0.01	0.358
Erucastrum gallicum	0.41	-0.91	0.03	**0.014**
Erysimum capitatum	-0.88	-0.47	0.01	0.480
Erysimum inconspicuum	0.96	0.28	0.00	0.801
Escobaria vivipara	0.73	-0.68	0.00	0.844
Euphorbia serpyllifolia	0.02	1.00	0.01	0.353
Lycopus spp.	0.19	0.98	0.01	0.418
Festuca ovina	0.31	-0.95	0.00	0.777
Gaura coccinea	-0.51	-0.86	0.00	0.874
Hedeoma hispida	0.98	0.18	0.01	0.366
Hesperostipa comata	0.89	-0.46	0.00	0.588
Heterotheca villosa	-0.02	-1.00	0.00	0.922
Hordeum jubatum	1.00	-0.06	0.33	**0.001**
Juncus balticus	0.20	-0.98	0.09	**0.001**
Juncus tenuis	-0.80	-0.60	0.00	0.970
Kochia scoparia	-0.32	0.95	0.01	0.330
Koeleria macrantha	-0.44	0.90	0.09	**0.001**
Lactuca scariola	0.74	-0.68	0.01	0.300
Lepidium densiflorum	0.30	-0.95	0.05	**0.002**
Lepidium ramosissimum	0.19	-0.98	0.01	0.381
Liatris punctata	0.97	0.26	0.01	0.211
Linum rigidum	0.88	0.47	0.01	0.397
Lithospermum incisum	0.99	-0.11	0.01	0.430
Medicago lupulina	0.87	-0.49	0.03	**0.021**

Melilotus alba	0.77	-0.64	0.11	**0.001**
Melilotus officinalis	0.88	-0.47	0.09	**0.001**
Monolepis nuttalliana	-0.92	0.39	0.01	0.180
Nassela viridula	0.99	0.13	0.01	0.276
Oenothera nuttallii	0.42	-0.91	0.01	0.281
Oxytropis sericea	0.10	0.99	0.03	**0.026**
Pascopyrum smithii	-0.15	0.99	0.01	0.201
Plantago major	0.25	-0.97	0.00	0.814
Plantago patagonica	0.98	0.20	0.01	0.336
Poa compressa	0.06	-1.00	0.00	0.811
Poa palustris	0.96	-0.27	0.24	**0.001**
Poa pratensis	0.52	-0.85	0.08	**0.001**
Poa secunda	-0.76	0.65	0.02	0.059
Potentilla gracilis	0.99	-0.11	0.02	0.059
Potentilla norvegica	1.00	-0.06	0.18	**0.001**
Potentilla pensylvanica	0.59	0.81	0.03	**0.036**
Puccinellia nuttalliana	0.96	-0.28	0.02	0.066
Ratibida columnifera	0.98	-0.18	0.03	**0.041**
Rumex crispus	0.99	-0.13	0.32	**0.001**
Rumex maritimus	0.99	-0.16	0.14	**0.001**
Salsola pestifer	-0.09	1.00	0.00	0.664
Schedonnardus paniculatus	0.96	0.27	0.01	0.181
Silene drumondii	-0.42	0.91	0.01	0.238
Sisymbrium altissimum	0.45	-0.89	0.03	**0.032**
Solidago missouriensis	0.37	-0.93	0.01	0.283
Sonchus arvensis	0.88	-0.48	0.07	**0.001**
Sonchus asper	0.95	0.31	0.00	0.628
Sporobolus cryptandrus	0.09	-1.00	0.02	0.083
Symphyotrichum ciliatum	0.40	-0.92	0.02	0.071
Symphyotrichum ericoides	1.00	-0.02	0.05	**0.006**
Symphytrichum laeve	0.25	-0.97	0.01	0.485
Taraxacum officinale	0.41	0.91	0.05	**0.003**
Thlaspi arvense	0.03	-1.00	0.01	0.165
Tragopogon dubius	0.98	0.18	0.01	0.238
Typha latifolia	-0.11	-0.99	0.01	0.194

Table D.8. Summary of all biological soil crust species (lichens, mosses, and others) recorded during the 2015 survey, ranked by average relative cover (% ± standard deviation).

Organism	Specific Epithet	Common Name	Growth Form	Vagrant	Rank	Average Cover (%)	Std. Dev
Lichen	*Acarospora schleicheri* (Ach.) A. Massal.	Soil paint lichen	Squamulose			0.0003	0.0053
	Buellia elegans Poelt	Elegant disc lichen	Crustose			0.0003	0.0053
	Candelaria vitellina (Ehrh) A. Massal.		Crustose			0.0031	0.0172
	Cetraria aculeata (Schreber) Fr.	Spiny shield lichen	Fruticose	+		0.0035	0.0543
	Circinaria fruiticulosa (Eversm.) Sohrabi	Vagrant Aspicillia	Fruticose	+		0.0031	0.0530
	Cladonia cariosa (Ach.) Sprengel	Split-peg soldiers	Squamulose/Fruticose			0.0264	0.2300
	Cladonia chlorophaea (Florke ex Sommerf.) Sprengel	Mealy pixie-cup	Squamulose/Fruticose			0.0051	0.0476
	Cladonia dahliana Kristinsson	Peg pixie lichen	Squamulose			0.0049	0.0603
	Cladonia pocillum (Ach.) O. J. Rich	Rosette pixie-cup	Squamulose/Fruticose		**7**	0.0718	0.3700
	Cladonia pyxidata (L.) Hoffm.	Pebbled pixie-cup	Squamulose/Fruticose		**2**	1.2232	4.2445
	Cladonia rei Schaerer	Wand lichen	Squamulose/Fruticose		**6**	0.1413	0.5518
	Cladonia robbinsii A. Evans	Yellow tongue Cladonia	Squamulose			0.0021	0.0228
	Collema tenax (Sw.) Ach.	Jelly Lichen	Foliose			0.0024	0.0183
	Diploschistes muscorum (Scop.) R. Sant.	Cow pie lichen	Crustose		**9**	0.0372	0.3212
	Fulgensia bracteata var. *bracteata* (Hoffm.) Räsänen	Bracted sulphur lichen	Crustose			0.0011	0.0105
	Ochrolechia upsaliensis (L.) A. Massal.	Tundra saucer lichen	Crustose			0.0163	0.1696
	Parmelia sulcata Taylor	Hammered shield lichen	Foliose			0.0033	0.0330
	Peltigera rufesens (Weiss) Humb.	Field dog-lichen	Foliose			0.0169	0.1783
	Phaeophyscia constipata (Norrlin & Nyl.) Moberg	Upstanding shadow lichen	Foliose			0.0222	0.1129
	Physconia muscigena (Ach.) Poelt	Frosted lichen	Foliose		**8**	0.0393	0.2258
	Placidium squamulosum (Ach.) BreuÂ¯		Squamulose			0.0006	0.0074
	Thelenella spp.		Crustose			0.0003	0.0053
	Xanthoparmelia camtschadalis (Ach.) Hale	Vagabond rockfrog	Foliose	+	**3**	0.2161	0.8303
	Xanthoparmelia wyomingica (Gyelnik) Hale	Wyoming rock-shield	Foliose	+	**10**	0.0347	0.3725
Moss	*Bryum caespiticum* Hedw.	Dry calcareous Bryum moss				0.0044	0.0591
	Polytrichum piliferum Hedw.	Bristly haircap			**4**	0.1806	1.4865
	Tortella fragilis (Hook. & Wilson) Limpr.	Fragile Tortella moss				0.0336	0.3864
	Tortula ruralis (Hedw.) G. Gaertn., B. Mey. & Scherb.	Star moss			**5**	0.1414	1.1234
	Unknown Moss					0.0160	0.2640
Spike-moss	*Selaginella densa* Rydb.	Prairie club-moss			**1**	3.9333	11.6114
Cyanobacteria	NA	Nostoc				0.0039	0.0305

Table D.9. Linear regressions describing the chrono-sequence of biological crust recovery along pipelines stratified by sampling distance.

Distance from Pipeline	T Value	P Value	R^2	$y = mx + b$
0 m (Trench)	0.29	0.770	0.001	y = 0.005 (±0.02) x + 0.58 (±0.47)
1 m	-0.26	0.793	0.001	y = -0.01 (±0.03) x + 1.65 (±0.84)
5 m	-0.89	0.379	0.011	y = -0.11 (±0.13) x + 10.76 (±3.90)
20 m	0.91	0.364	0.012	y = 0.17 (±0.19) x + 5.16 (±5.76)
55 m	0.52	0.603	0.004	y = 0.08 (±0.16) x + 8.98 (±4.87)

Table D.10. Biplot vector scores associated with the NMDS ordination of biological soil crust composition, including pipeline, biological soil crust, plant community, seed bank, ground cover, and soil properties, across 18 pipeline study sites. Factors significant at $P < 0.05$ were included in the ordination.

Characteristics	Factor	MDS1	MDS2	R^2	P value
Pipeline	Age	0.92	-0.38	0.00	0.825
	Distance	0.91	-0.40	0.06	**0.001**
	Diameter	-1.00	-0.06	0.03	**0.006**
Biological Soil Crust	Biological Crust Cover	0.70	-0.71	0.69	**0.001**
	Pielou’s Evenness	0.88	0.48	0.47	**0.001**
	Richness	0.99	0.12	0.94	**0.001**
	Shannon's Diversity	0.91	0.40	0.78	**0.001**
	Simpson's Diversity	-0.99	-0.17	0.14	**0.001**
Plant Community	Total Biomass	-0.98	0.20	0.02	**0.025**
Seed Bank	Total Density	-0.99	0.14	0.02	**0.031**
	Graminoids	-0.30	0.95	0.03	**0.014**
	Introduced	-0.97	-0.26	0.03	**0.002**
	Introduced Annual Forbs	-0.84	-0.54	0.00	0.455
	Introduced Biennial Forbs	-1.00	-0.08	0.02	**0.044**
	Introduced Grasses	-0.39	0.92	0.00	0.753
	Introduced Perennial Forbs	-0.96	-0.28	0.03	**0.010**
	Native	-0.79	0.62	0.01	0.242
	Native Annual Forbs	0.28	0.96	0.02	**0.036**
	Native Biennial Forbs	-0.91	-0.41	0.01	0.095
	Native Grasses	-0.56	-0.83	0.01	0.270
	Native Perennial Forbs	-0.51	0.86	0.00	0.816
	Richness	-0.71	0.71	0.01	0.083
	Sorenson's Similarity	0.58	-0.81	0.02	**0.022**
	Shannon's Diversity	-0.32	0.95	0.00	0.508
	Simpson's Diversity	0.09	1.00	0.00	0.932
	Pielou's Evenness	0.79	-0.61	0.00	0.584
Ground Cover	Bare Ground	-0.13	0.99	0.02	**0.028**
	Litter Biomass	-0.96	0.27	0.04	**0.002**
	Litter Cover	-0.85	0.52	0.29	**0.001**
	Manure	-0.85	-0.52	0.00	0.867
	Rocks	0.94	0.33	0.02	0.054
	Stems	0.92	0.39	0.02	**0.032**
Soil Properties	Bulk Density	-0.65	0.76	0.03	**0.003**
	C	0.54	-0.84	0.17	**0.001**
	C:N Ratio	-0.68	0.73	0.04	**0.006**
	EC	-0.92	-0.39	0.03	**0.014**
	N	0.56	-0.83	0.18	**0.001**
	OM	0.54	-0.84	0.14	**0.001**
	pH	-1.00	0.05	0.02	**0.050**
Texture	Clay	-0.35	-0.94	0.00	0.912
	Sand	-0.62	0.78	0.15	**0.001**
	Silt	0.64	-0.77	0.18	**0.001**

Table D.11. Biplot vector scores for significant species associated with the NMDS ordination of biological soil crust composition, based on 18 pipeline study sites. Only species significant at P < 0.05 were plotted.

Species	**MDS1**	**MDS2**	**R^2**	**P value**
Acarospora schleicheri	0.75	0.66	0.06	**0.003**
Bryum caespiticum	0.40	0.91	0.05	**0.009**
Buellia elegans	0.95	0.31	0.03	**0.044**
Candelaria vitellina	1.00	0.02	0.22	**0.001**
Cetraria aculeata	0.40	-0.91	0.08	**0.002**
Circinaria fruiticulosa	0.41	-0.91	0.06	**0.004**
Cladonia cariosa	0.60	-0.80	0.10	**0.001**
Cladonia chlorophaea	1.00	-0.05	0.08	**0.001**
Cladonia dahliana	0.63	-0.78	0.02	0.071
Cladonia pocillum	0.95	0.32	0.17	**0.001**
Cladonia pyxidata	0.86	-0.51	0.30	**0.001**
Cladonia rei	0.46	0.89	0.20	**0.001**
Cladonia robbinsii	0.73	-0.68	0.07	**0.002**
Collema tenax	0.88	0.47	0.18	**0.001**
Diploschistes muscorum	0.55	0.83	0.06	**0.008**
Fulgensia bracteata	0.20	0.98	0.17	**0.001**
Ochrolechia upsaliensis	0.62	0.78	0.04	**0.027**
Parmelia sulcata	0.87	0.49	0.03	**0.032**
Peltigera rufesens	0.61	-0.79	0.10	**0.001**
Phaeophyscia constipata	0.46	0.89	0.26	**0.001**
Physconia muscigena	0.68	-0.73	0.24	**0.001**
Placidium squamulosum	0.43	0.90	0.10	**0.001**
Polytrichum piliferum	0.30	0.96	0.05	**0.012**
Selaginella densa	0.55	-0.83	0.59	**0.001**
Thelenella spp.	-0.15	0.99	0.01	0.127
Tortella fragilis	0.33	0.94	0.07	**0.004**
Tortula ruralis	0.96	-0.26	0.03	**0.025**
Xanthoparmelia camtschadalis	0.97	-0.24	0.29	**0.001**
Xanthoparmelia wyomingica	0.45	-0.89	0.12	**0.001**
Nostoc	0.86	0.50	0.06	**0.003**
Unknown Moss	0.88	-0.48	0.05	**0.012**
Dummy Variable	0.00	0.00	0.00	1.000

Table D.12. Biplot vector scores associated with the NMDS ordination of biological soil crust composition, including the proportion of soil exposure and litter cover. Factors significant at $P < 0.05$ were included in the ordination.

Characteristics	Factor	MDS1	MDS2	R^2	P value
Pipeline	Age	0.79	0.62	0.00	0.871
	Distance	0.55	0.83	0.09	**0.001**
	Diameter	-0.85	-0.52	0.02	**0.017**
Biological Soil Crust	Biological Crust Cover	0.77	0.64	0.77	**0.001**
	Evenness	0.96	-0.26	0.41	**0.001**
	Richness	1.00	0.04	0.85	**0.001**
	Shannon's Diversity	0.97	-0.23	0.67	**0.001**
	Simpson's Diversity	-0.89	-0.47	0.16	**0.001**
Plant Community	Total Biomass	-1.00	0.06	0.03	**0.016**
Seed Bank	Total Density	-0.98	0.22	0.03	**0.009**
	Graminoids	-0.34	-0.94	0.04	**0.003**
	Introduced	-0.99	-0.16	0.03	**0.004**
	Introduced Annual Forbs	-0.85	-0.53	0.00	0.752
	Introduced Biennial Forbs	-0.64	-0.77	0.02	**0.015**
	Introduced Grasses	-0.03	-1.00	0.01	0.291
	Introduced Perennial Forbs	-0.91	0.42	0.05	**0.003**
	Native	-0.84	0.54	0.02	0.059
	Native Annual Forbs	0.60	0.80	0.00	0.441
	Native Biennial Forbs	-0.68	-0.73	0.02	0.042
	Native Grasses	-0.55	0.83	0.02	**0.020**
	Native Perennial Forbs	-0.25	-0.97	0.00	0.878
	Richness	-0.65	-0.76	0.01	0.093
	Sorenson's Similarity	0.84	0.54	0.01	0.071
	Shannon's Diversity	-0.13	-0.99	0.01	0.282
	Simpson's Diversity	0.20	-0.98	0.00	0.438
	Pielou's Evenness	0.70	-0.71	0.01	0.212
Ground Cover	Bare Ground	0.11	-0.99	0.55	**0.001**
	Litter Biomass	-0.99	-0.11	0.05	**0.001**
	Litter Cover	-0.91	0.41	0.46	**0.001**
	Manure	-0.33	0.94	0.00	0.659
	Rocks	0.52	-0.86	0.04	**0.008**
	Stems	1.00	-0.06	0.03	**0.008**
Soil Properties	Bulk Density	-0.44	-0.90	0.04	**0.001**
	C	0.57	0.82	0.17	**0.001**
	C:N Ratio	-0.49	-0.87	0.05	**0.001**
	EC	-0.92	0.38	0.04	**0.005**
	N	0.59	0.81	0.19	**0.001**
	OM	0.54	0.84	0.14	**0.001**
	pH	-0.18	-0.98	0.07	**0.001**
Texture	Clay	-0.97	-0.25	0.00	0.877
	Sand	-0.69	-0.72	0.15	**0.001**
	Silt	0.70	0.72	0.19	**0.001**

Table D.13. Biplot vector scores for significant species related to the axes of the NMDS ordination for biological soil crust composition, including the proportion of soil exposure and litter cover along 18 pipeline study sites. Only species significant at P < 0.05 were plotted.

Species	**MDS1**	**MDS2**	**R^2**	**P value**
Acarospora schleicheri	0.90	-0.44	0.03	**0.027**
Bryum caespiticum	0.61	-0.80	0.03	**0.030**
Buellia elegans	0.95	-0.30	0.02	0.066
Candelaria vitellina	0.99	0.16	0.21	**0.001**
Cetraria aculeata	0.64	0.77	0.05	**0.006**
Circinaria fruiticulosa	0.62	0.78	0.04	**0.016**
Cladonia cariosa	0.76	0.65	0.09	**0.001**
Cladonia chlorophaea	0.90	0.44	0.11	**0.001**
Cladonia dahliana	0.94	-0.34	0.02	0.071
Cladonia pocillum	0.98	0.19	0.14	**0.001**
Cladonia pyxidata	0.89	0.46	0.33	**0.001**
Cladonia rei	0.93	-0.36	0.09	**0.001**
Cladonia robbinsii	0.89	0.45	0.06	**0.001**
Collema tenax	0.92	-0.39	0.12	**0.001**
Diploschistes muscorum	0.96	-0.29	0.03	**0.018**
Fulgensia bracteata	0.29	-0.96	0.07	**0.001**
Ochrolechia upsaliensis	0.57	-0.82	0.04	**0.005**
Parmelia sulcata	0.75	-0.66	0.02	**0.034**
Peltigera rufesens	0.82	0.58	0.07	**0.001**
Phaeophyscia constipata	0.70	-0.72	0.14	**0.001**
Physconia muscigena	0.95	0.31	0.18	**0.001**
Placidium squamulosum	0.50	-0.86	0.07	**0.001**
Polytrichum piliferum	0.31	-0.95	0.03	**0.018**
Selaginella densa	0.66	0.75	0.66	**0.001**
Thelenella spp.	-0.45	-0.89	0.01	0.155
Tortella fragilis	0.67	-0.74	0.03	**0.014**
Tortula ruralis	0.93	0.37	0.03	**0.025**
Xanthoparmelia camtschadalis	1.00	0.57	0.30	**0.001**
Xanthoparmelia wyomingica	0.59	0.80	0.14	**0.001**
Nostoc	0.76	-0.65	0.05	**0.004**
Unknown Moss	0.98	-0.18	0.04	**0.018**
Bare Ground	0.11	-0.99	0.55	**0.001**
Litter Cover	-0.91	0.41	0.46	**0.001**

Appendix E. Chapter 7.

1	Dalea D+L	25	Dalea Litter	49	Medicago D+L	73	Melilotus Defoliation
2	Trifolium D+L	26	Trifolium Litter	50	Astragalus D+L	74	Trifolium Defoliation
3	Vicia D+L	27	Medicago Litter	51	Vicia D+L	75	Medicago Defoliation
4	Melilotus D+L	28	Vicia Litter	52	Melilotus D+L	76	Dalea Defoliation
5	Astragalus D+L	29	Melilotus Litter	53	Dalea D+L	77	Vicia Defoliation
6	Medicago D+L	30	Astragalus Litter	54	Trifolium D+L	78	Astragalus Defoliation
7	Dalea Litter	31	Dalea	55	Medicago Defoliation	79	Trifolium Litter
8	Melilotus Litter	32	Vicia	56	Dalea Defoliation	80	Vicia Litter
9	Medicago Litter	33	Melilotus	57	Melilotus Defoliation	81	Dalea Litter
10	Trifolium Litter	34	Trifolium	58	Vicia Defoliation	82	Medicago Litter
11	Astragalus Litter	35	Medicago	59	Trifolium Defoliation	83	Astragalus Litter
12	Vicia Litter	36	Astragalus	60	Astragalus Defoliation	84	Melilotus Litter
13	Melilotus	37	Melilotus D+L	61	Melilotus	85	Medicago D+L
14	Dalea	38	Astragalus D+L	62	Medicago	86	Vicia D+L
15	Astragalus	39	Vicia D+L	63	Vicia	87	Dalea D+L
16	Trifolium	40	Dalea D+L	64	Astragalus	88	Melilotus D+L
17	Vicia	41	Trifolium D+L	65	Dalea	89	Trifolium D+L
18	Medicago	42	Medicago D+L	66	Trifolium	90	Astragalus D+L
19	Vicia Defoliation	43	Trifolium Defoliation	67	Trifolium Litter	91	Trifolium
20	Dalea Defoliation	44	Astragalus Defoliation	68	Vicia Litter	92	Astragalus
21	Trifolium Defoliation	45	Melilotus Defoliation	69	Medicago Litter	93	Medicago
22	Melilotus Defoliation	46	Medicago Defoliation	70	Melilotus Litter	94	Dalea
23	Medicago Defoliation	47	Vicia Defoliation	71	Astragalus Litter	95	Melilotus
24	Astragalus Defoliation	48	Dalea Defoliation	72	Dalea Litter	96	Vicia

Figure E.1. Sample of experimental design and plot plan from the native site within the Central Parkland, where treatments of defoliation (D) and litter removal (L) were applied in a two-way factorial. Within treatment plots, each of 6 legume species were randomly seeded in subplots (split-plot).

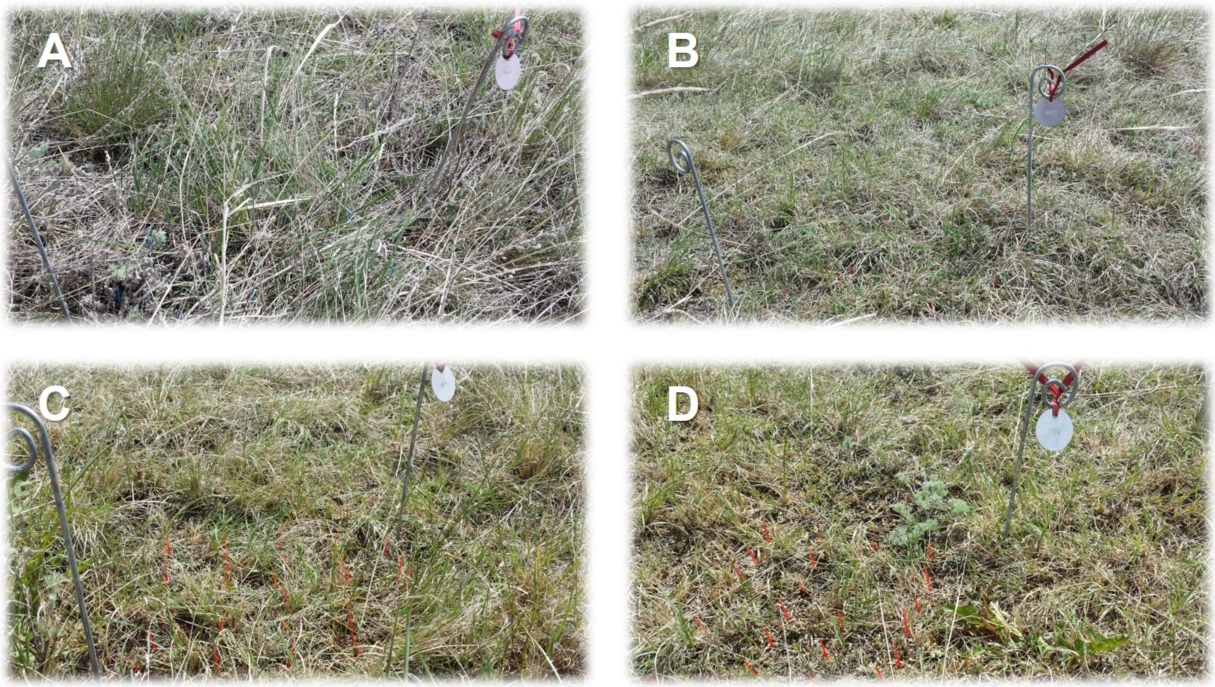


Figure E.2. Example of defoliation (D) and litter (L) removal treatments from the native Dry Mixedgrass (DMG) prairie site. The treatments were applied as follows: A = control (+L -D) (i.e. no defoliation and no removal of litter); B = defoliation (+D), which occurred every three weeks during the growing season; C = raked to remove standing and fallen litter (-L); D = defoliated and raked to remove litter (-L +D). Treatments were applied to a 1 m x 6 m strip, each subplot for a species was 1 m x 1 m, and the seeds were seeded within a 50 cm x 50 cm area in the subplot's center.

Table E.1. Summary of the ecosite and soil characteristics for each of the Dry Mixedgrass (DMG) prairie and Central Parkland (CP) native (N) and tame (T) grasslands used to study legume seedling demographics.

Property		**DMG-N**	**DMG-T**	**CP-N**	**CP-T**
Soil Properties	Carbon (%)	1.6	1.2	4.7	2.6
	Nitrogen (%)	0.2	0.1	0.4	0.2
	C:N Ratio	10.0	11.0	11.6	11.8
	Organic Matter (%)	3.1	2.4	8.3	4.7
	pH	6.5	6.9	5.7	6.2
	Electrical Conductivity	256.0	328.5	238.5	311.0
Soil Texture	Sand	67.6	72.5	67.8	71.4
	Clay	28.0	27.1	28.8	26.4
	Silt	4.4	0.4	3.4	2.2
	Texture Class	Sandy Clay Loam	Sandy Clay Loam	Sandy Clay Loam	Sandy Clay Loam
Soil Type	Sub Group	O. BRC	O. BRC	O. BLC	O. BLC
	Soil Series	Pemukan	Cavendish	Elnora	Elnora
Ecosite		Loamy-Gravely	Loamy	Loamy	Loamy

O. BLC = Orthic Black Chernozem, O. BRC = Orthic Brown Chernozem

Table E.2. Legume stages used to describe plant growth and development. Based on stages described for alfalfa by Fick and Mueller (1989).

Stage	Description
0	Cotyledons only.
1	Emergence of first true leaf bearing a single leaflet.
2	Complete emergence of first true leaf with multiple leaflets present.
3	Seedling with at least 2 or more leaves. Less than 5 cm tall.
4	Early vegetative stage. 5 to 15 cm tall.
5	Mid vegetative stage. > 15 -30 cm tall.
6	Late vegetative. > 30 cm tall.
7	Early bud development.
8	Late bud development.
9	Early flowering.
10	Late flowering. Flowers senescing.
11	Early seed pod. Fruit development beginning. Small ovules.
12	Late seed pod. Fruit developed, ovules large. Fruit and ovules still green.
13	Ripe seed pod. Fruits dry, seeds mature.

Stage is equal to the latest developmental stage present.