The Relationship of Plant Diversity to Alberta's Range Health Assessment

by

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

Master of Science

in

Rangeland and Wildlife Resources

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Abstract

Grassland conversion to other land uses is the leading cause of decline in biodiversity in these ecosystems. In remaining grasslands, grazing can have positive or negative effects on biodiversity, and the ability to assess grazing effects is critical to apply informed conservation management decisions. One way to measure grazing impacts is with rangeland health, an operational measure of grassland function. Rangeland health is assessed by summing the scores of individual components of community integrity, structure, hydrological function and nutrient cycling, site stability and noxious weed presence. It is often assumed that if rangeland health is maintained then biodiversity is conserved; however, this has not been broadly tested. This study investigated relationships of plant diversity measures with Alberta's Rangeland Health Assessment at two different scales. In the first study, rangeland health was measured at discreet locations across a broad geographic area and plant diversity did not relate to rangeland health. However, components measuring soil erosion, nutrient cycling / hydrological function, structure and noxious weeds all related to diversity metrics, although the relationships were influenced by environmental factors such as topographical and geographical moisture gradients. In the second study, rangeland health was measured in plant communities throughout a pasture and related to community patchiness. The patchiness measure was created by comparing the number of ecosites present to the number of plant communities occurring with the addition of grazing effects. Average health scores and their range (max-min) both related to patchiness within pastures. Furthermore, the relationship between patchiness and rangeland health was influenced by abiotic differences as well as a broad climatic gradient.

Implications for grassland conservation and management are that although no direct relationship was found between plant diversity and total rangeland health score at the community level, diversity nevertheless demonstrated a sensitivity to several subcomponents of the health assessment, and therefore may provide utility in tracking ongoing diversity changes. Also, if completed in multiple locations within a pasture, rangeland health can be utilized to indicate community patchiness, which has been shown to be important to biodiversity and a variety of wildlife species. Management goals should be specified between diversity and rangeland health as there may be instances where some areas of lower levels of health can promote patchiness and overall diversity.

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List of Symbols and Abbreviations

°C - Degrees Celcius

ABMI - Alberta Biodiversity Monitoring Institute

AEP - Alberta Environment and Parks

AHM - Annual heat / moisture index

AHM_s - Annual heat / moisture ratio - rescaled

BDO - Biodiversity Convention Office

Blo - Blowout ecosite

C - Individual component score

C-1 - Component 1 - Plant community integrity of the Alberta range health assessment

C-2 - Component 2 - Canopy structure of the Alberta range health assessment

C-3 - Component 3 - Nutrient cycling / hydrological function of the Alberta range health assessment

C-4.1 - Component 4.1 - Site stability – evidence of erosion of the Alberta range health assessment

C-4.2 - Component 4.2 - Site stability – soil exposure of the Alberta range health assessment

C-5 - Component 5 - Prohibited noxious and noxious weeds of the Alberta range health assessment

Cly - Clayey ecosite

CRC - Committee on Rangeland Classification

CS - Choppy Sandhills ecosite

DCA - De-trended correspondence analyses

DMG - Dry Mixedgrass natural subregion

ECCC - Environment and Climate Change

ES - Ecosite

ESG - Ecological site groups of uplands (a combination of Sandy, Blowout, and Loamy ecosites) and lowlands (a combination of Saline Lowland and Overflow)

ESRD - Environment and Sustainable Resource Development

FF - Foothills Fescue natural subregion

GOA - Government of Alberta

Gr - Gravel ecosite

GVI - Grassland Vegetation Inventory

H - Healthy rating (score ≥ 75 %) of the Alberta range health assessment

HWP - Healthy with problems rating (score ≥ 50 % and < 75%) of the Alberta range health assessment

Li - Limy ecosite

LL - Lowland grouping of ecosites

Lo - Loamy ecosite

MAP - Mean annual precipitation

MAT - Mean annual temperature

ME - Mesic grouping of natural subregions, includes Foothills Fescue, Foothills Parkland, Northern Fescue and Central Parkland

MG - Mixedgrass natural subregion

MULTISAR - A program in southern Alberta conserving habitat for multiple species at risk.

NMDS - Non-metric multi-dimensional scaling

NSRG – Natural Subregion Groups

Ov - Overflow ecosite

PERMANOVA - permutational multivariate analysis of variance

PCT - Plant community type

PCT:ES - Plant community type by ecosite ratio

PLOS - Public Library of Science

PNAS - Proceedings of the National Academy of Sciences of the United States of America

R – R statistical computing software

r^2 - Regression goodness of fit measure

Sa - Sand ecosite

SB - Subirrigated ecosite

SE - Standard error

SL - Saline Lowland ecosite

SwG - Shallow to Gravel ecosite

Sy - Sandy ecosite

TB - Thin Break ecosite

UH - Unhealthy rating (score < 50 %) of the Alberta range health assessment

UL - Upland grouping of ecosites

XE - Xeric grouping of natural subregions, includes Dry Mixedgrass and Mixedgrass

1.0 Rangeland conservation, grazing management, rangeland health, and biological diversity

1.1 Introduction

Land managers face multiple challenges on native grasslands, where the preservation of ecological functions and overall maintenance of attributes such as biological diversity is expected (CRC 1994, West 1993). A first step to successful management is the ability to measure the current state of these functions and attributes. The rangeland health assessment was developed for this purpose. Establishing relationships between these assessments and biodiversity across a landscape are a vital component of current range management where outcomes are conservation orientated. This thesis examines linkages between rangeland health and its components to biodiversity and assesses the validity of the rangeland health tool to assess grazing effects on plant diversity.

Rangelands are generally natural systems that have developed within their landforms, climates, and disturbance regimes over the last millennia (Bailey et al. 2010, CRC 1994). The broadest definition includes all parts of the world not farmed or covered in pavement, concrete, rock, or ice (Holechek et al. 1989). For purposes of this study, rangelands are narrowed to uncultivated nor planted lands capable of providing the necessities of life to grazing animals. The primary focus is native grasslands that have included large herbivore grazing disturbance as part of their development (Teague 2013, Holechek et al. 1989, Milchunas et al. 1988).

Rangelands, including grasslands were key to the settlement of western North America through the production of livestock and conversion to cropland (Bailey et al. 2010, CRC 1994). Today, there is roughly 312 million hectares of rangelands in the United States mostly in the west, and 11.4 million hectares in Canada's Prairies (Baily et al. 2010). Since settlement it is estimated the western States lost 27% of its rangeland, with ongoing cropland conversion (CRC 1994, Greene & Stager 2001). In Canada, an estimated 80% of prairie grasslands have been lost to conversion to other uses (Bailey et al. 2010).

Similar to other areas in North America, much native grasslands in Alberta have been converted to annual crops or perennial pasture (Bailey et al. 2010). Currently only about 40% of native grasslands are left in the Dry Mixed Grass and Mixed Grass natural subregions and substantially less remaining in the Foothills Fescue (Adams et al. 2003,2013a,2013b). The parkland areas of the province have been estimated to have less than 25% native remaining and conversation is ongoing with an estimate of 1.2 to 1.3% of grasslands and parklands have been converted from native within last 10 years (Schieck et al. 2014).

Although reduced, rangelands still occur throughout many regions in Alberta from the prairies and their bordering parklands in the southeast and in the Peace River region, the foothills and Rocky Mountains to the west, and the Boreal mixed woods to the north (Downing & Pettapiece 2006, Weerstra 1986, Ehlert & Lawrence 1999). Grasslands occur extensively in the prairies and are restricted to certain slopes and soil types in the foothills, parkland, and mountain regions. These grasslands are driest in the southeast and become more mesic moving west or north (Downing & Pettapiece 2006). The drier grasslands are dominated by

needle grasses (*Sitpa spp.*) and wheat grasses (*Agropyron spp.*), but are replaced by fescues (*Festuca spp.*) as areas become more mesic.

Much loss has occurred within the last century, but there also have been many efforts for conservation and proper management of what is remaining (Bailey et al. 2010, CRC 1994, West 1993). The creation of the Forest Reserve in western Canada at the turn of the 20th century is a good example (Weerstra 1986). Although much settlement occurred during this period, the forest reserve in Canada (now Alberta) was also created that excluded the private sale of western Canada's headwaters of major prairie rivers for the conservation of water, vegetation and timber resource, and sustainable management is still at the forefront for this area today (Weerstra 1986, GOA 2004). More currently, the Land Use Framework was established in 2014, with conservation of landscapes and their connectivity are high priorities (GOA 2014). Along with conservation, there was a focus on the remaining intact grasslands to understand the effects of disturbance, particularly grazing, on ecological services such as biodiversity these remaining lands contribute (Bailey et al. 2010, CRC 1994).

1.2   Conservation and Biological Diversity

Broadly speaking biological diversity is the variety of species and ecosystems on earth (BDO 1995). The convention of biodiversity defined it as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part. This includes diversity within species, between species and of ecosystems" (Glowka et al. 1994). The last sentence refers to the different scales that biodiversity can refer to. For example, among plants, scales are

variation within a species, species diversity within a community, diversity of plant communities across an ecosystem, and lastly across landscapes (West 1993).

Biodiversity provides resilience so ecosystems can withstand changes such as climatic or physical perturbation, as well as provide structure and multiple interconnecting functions to different spatial scales (BCO 1995, West 1993, Noss 1990). For example, a diverse community commonly has only a few common species but many more rare species whose function is minimal or considered redundant with other organisms. However, many studies and statistical theories show that rare species may increase their importance to their environment after a major change and provide overall stability to the ecosystem (Peterson et al. 1998 Tilman et al. 1998, Loreau 2000, West 1993, Whittaker 1965). For plants, productivity may be maintained through diversity by increasing the probability having species that maximize production in a particular environment, or having species occupying different niches and maximizing all resources available (Tilman et al. 1998, Tilman et al. 1997, Loreau 2000).

Hence, biological diversity is often a key term utilized when referring to the conservation and sustainable management of native lands. As a country, Canada has been officially involved in the concept since the 1992 Convention of Biodiversity, a world effort to conserve biodiversity, promote sustainably use of its components, and the equitable sharing of genetic resources (Glowka et al. 1994). In 1995 and again in 2010, Canada adopted a 10 year strategic plan, endorsed federally, provincially and territorially (BCO 1995, ECCC 2016). This included a goal that agricultural landscapes provide a stable or improved level of biodiversity. Provincially, Alberta created a Land Use Framework with one of its primary outcomes as "Biodiversity and ecosystem function are sustained with shared stewardship" (GOA 2014).

Similarly the Alberta Biodiversity Monitoring Institute was established in 2007 to monitor and report on biodiversity status (Narwani & Schieck 2015).

1.3   *Biodiversity and Livestock Grazing*

A significant portion of the livestock industry relies on native grasslands and their inherent biological diversity for consistent primary production of animals through various landscapes, climatic differences, and both anthropogenic and natural disturbances (Baily et al. 2010, CRC 1994). A diverse landscape provides areas that can produce forage in varying annual weather conditions or other perturbations (Loreau 2000, Tilman et al. 1998). In fact early establishment livestock in western Canada relied on grassland and parkland species that evolved with grazing (Evans 2001). Historically, bison grazing and fire played a key role in shaping the grasslands communities and promoting heterogeneity (Knapp et al. 1999, Morgan 1980). These grassland mosaics withstood chinook winds, provided seasonal growth, provided winter shelter, and recovered from grazing disturbances. Today, although livestock grazing is significantly different than the historic grazing patterns of bison (Evans 2001), grazing plays a role in maintaining biodiversity within these natural systems that evolved with large wild herbivores that are no longer present (Knapp et al. 1999, Collins and Barber 1986).

Grazing intensity, frequency, and seasonality have the ability to modify plant species composition and production (Briske et al. 2008, Teague et al. 2013), therefore understanding the relationship between grazing management and its implication to diversity is important. At the plant community scale, relationships between grazing disturbance and plant diversity are often characterized by the intermediate disturbance hypothesis (Grime 1973, Connell 1978)

where natural disturbances such as grazing prevent climatically dominant species from excluding other species from the community in certain environments. Grassland communities on landscapes that evolved with grazing tend to allow a broader diversity of flora to be maintained within the community's structure under light to moderate grazing than one with grazing removed (Collins & Barber 1986, Milchunas et al. 1988, Bai et al. 2001). Overgrazing however, causes species to decline to only those tolerant of heavy grazing and lessens structural complexity (Collins & Barber 1986, Noss 1990, West 1993). This in turn reduces fauna that rely either on these species or their corresponding structural attributes (Fuhlendorf & Engle 2001, Sliwinski & Koper 2015, Milchulnas et al. 1988). Overgrazing also affects rangelands ability to maintain its ecological functions such as capturing water, cycling nutrients, development of soils, and limits productivity and the number of fauna an area can sustain (West et al. 2016, Briske et al. 2005, Adams et. al. 2011, TGUCT 1995). All these factors limit overall biodiversity, whereas a broad range of flora creates habitat for a broad variety of species such as birds with obligate structure requirements, grazing or browsing animals, insects and predators (Fuhelndorf & Engle 2001, Vavra 2005, Kostenko et al. 2017). Therefore at this scale, it seems that light to moderate grazing provides optimal biodiversity.

Other studies suggest that at a landscape scale, uneven use with overgrazed areas and others under-grazed increased biodiversity (Collins & Barber 1986, Fuhlendorf & Engle 2001, Sliwinski & Koper 2015). For example, different types of song birds rely on differing vegetation structures, therefore a more diverse landscape create habitat for more types of birds (Bock et al. 1993, Vavra 2005). A landscape with a mosaic of high to low grazing rates was shown to increase this heterogeneity in structure, and provide habitat for more types of songbirds than

one with less heterogeneity (Lwiwski et al. 2015, Swilinsky & Koper 2015).   At this scale, it seems different grazing regimes across the landscape provide more diversity than a single level of grazing.

This highlights the importance of managing grazing appropriately for maintaining biological diversity. This is especially required as criticisms arise regarding grazing on rangelands. For example, cattle grazing has been viewed as negatively affecting wildlife habitat (Fleischner 1994), where most examples are often from mismanaged areas (Vavra 2005). Also, recently cattle production has been identified as the most burdensome to the environment in a life cycle analysis (Eshel et al. 2014). This account included rangelands and a blanket statement that livestock production negatively affected ecosystem services and biodiversity.

1.4   Range Management and Health Assessments

Rangeland management, the manipulation of rangeland components through a plant / animal interface to obtain the sustainable use of goods and services they provide, has evolved over the past century (Holechek et al. 1989). Since early settlement rangelands were (and still are) utilized for primary industries such as livestock and timber production. In both Canada and the United States, rangelands went through a period of degradation particularly in the last half of the 19th century and first half of the 20th century as livestock production increased along with settlement (CRC 1994, Bailey et al. 2010, Adams et al. 2004). Much of this land was publicly owned and a desire for better conservation led to the study and practice of rangeland management. To promote informed management, inventory and assessment methodologies were created to understand current state and capability of rangelands (Weerstra 1986, CRC

1994, Adams et al. 2004, Rumbolt et al. 2011). Rangeland health assessments are one type of these methodologies that evolved from earlier methodologies.

The term rangeland health has received much attention the last few decades as range management practices developed. The need for a consistent assessment arose as disagreements of the state of rangelands occurred at a time during a time of heightened goals of increased conservation and management of native areas used for livestock production (CRC 1994, TGUCT 1995). The first type of assessment, named range condition, was developed early on in the study of range management (Dyksterhuis 1949, TGUCT 1995). It was plant community succession based and although popular, was criticized through time because it did not assess all the functions of rangelands or accurately depict alternative successional pathways and stable vegetational states (TGUCT 1995, Westoby et al. 1989, Friedel 1991). Range condition only compared the successional status of the current community to the climax community that could grow on that particular location, and did not assess other important physical parameters (TGUCT 1995), and also assumed that all declines in condition are reversible back to a single climax (Westoby et al. 1989, Friedel 1991, Laycock 1991).

Although measuring grazing effects on plant community composition is a fundamental part of assessing rangelands, it is not their sole value. Rangeland are recognized for other attributes such as wildlife habitat, water, minerals, energy, wood products, plant and animal genepools, and recreational opportunities, and require management practices that incorporate these values (CRC 1994, Fuhlendorf et al. 2012, Noss et al. 2002). Newer rangeland health measures were created from these observations, and although the assessment developed slightly different between jurisdictions, the premise is the same that the assessment captures

species composition changes, but also measures rangeland's ability to maintain ecological functions such as soil/site stability, hydrologic function, nutrient cycling and biotic integrity (TGUCT 1995, Adams et al. 2011, Pellant et al. 2005). A common function or trait that rangeland health assessments purport to address is biological diversity.

1.5   Biodiversity Measures

It is often assumed that if rangeland functions exist, biodiversity will be conserved. In fact, many consider biological diversity a function of healthy rangelands (Adams et al. 2011, CRC 1994, TGUCT 1995). Others suggest that biodiversity metrics could be used an indicator for range health (Symstad & Jonas 2011). However when studying or measuring biodiversity, choosing what to focus on, how to sample it, and what scale to use are necessary considerations that may have various impacts to the biodiversity message.

Biological diversity is seldom measured as a complete census of all the species within a geographic area. These tasks are both cost and time prohibitive (Magurran 2004). Instead, biodiversity measures commonly focus on groups of species studied within defined geographic areas (West 1993, Noss 1990). Also, biodiversity measures are most useful when used as a comparative study, where differences of biodiversity are compared between two assemblages (Magurran 2004).

There are different types of biodiversity metrics to choose from as well. The most common method and probably simplest form of capturing biodiversity is species richness, where the number of species of a defined assemblage is counted (Magurran 2004, West 1993). Species rank - abundance models are separate measure that describes the distribution of

species, or evenness; whether their abundance is evenly spread or dominated by a few. Indices have also been created that attempt to combine these two measures into one value. The Shannon and Simpson's index may be the two most common, Shannon being highly influence by richness, and Simpson's by abundance distribution (Magurran 2004).

Lastly, appropriate scale must be chosen, whether at the species, plant community, or landscape (West 1993, Noss 1990). The above measures work well for plant communities, but at the landscape, overall diversity is measured by comparing how dissimilar communities within a defined area are. The term beta diversity is used as a measure for this scale; the premise being if plant communities within a defined area are markedly different, the overall diversity should be greater (Magurran 2004). Analyzing the dissimilarity between numerous communities becomes complex with uneven sample sizes, and data is often not normally distributed. However, non-parametric, multivariate approaches have been developed to handle some of these issues (Anderson et al. 2006). All these factors lead to problems with assessing what kind of biological diversity to measure and care needs to be taken in deciding and describing what to report.

1.6   Project Description

As field measures for rangeland management evolve, understanding their relationships to factors such as biodiversity should be investigated. Many studies and theories consider variation of grazing through changes in cattle grazing intensity and time, and is usually managed at the pasture or study site scale. To accomplish either much sampling is required to split different areas by grazing effects, or study areas are small. Range health methodology is a

rapid assessment operationally used to measures at the plant community scale or rolled up to the pasture level. It measures the cumulative response of an area to all its historic management practices and disturbances (Adams et al. 2011, TGUCT 1995). Therefore it could be considered a proxy measure to historic and present grazing effects. This concept was tested with range condition and plant species diversity and showed similar trends to theorized models for some biodiversity measures (Bai et al. 2001). Range health considers more attributes than range condition including non-native species, structure, soil protection, and moisture retention. The implication of all these considerations needs to be tested similarly for its relationship to plant diversity.

This study was divided into two scales to consider the value of range health as an indicator of plant diversity at the community and pasture level. The plant community is generally the lowest scale which rangeland health is assessed. Here the direct relationship of range health and plant diversity was studied. Alberta rangeland health assessment information was collected on sites throughout the Prairie and Parkland portions of Alberta, Canada, and was tested against diversity measures derived from abiotic and species composition information collected at the same locations. This approach tested whether diversity metrics responded to rangeland health in ways such as the other theories that predict changes to diversity with disturbance (Grime 1973, Milchunas et al. 1988). Rangeland health may measure the level of disturbance, however how that describes diversity within a variable grassland setting across a broad topographic and climatic regime is unknown. Also, the components that make up the rangeland health assessment such as community structure, soil/site stability, hydrologic function, nutrient cycling and biotic integrity were tested separately for their linkages.

The study at the pasture scale used current inventory methodology that captures overall pasture health by dividing the unit into broad community types, and rating each with a rangeland health assessment. This information was utilized to investigate the influence of rangeland health to the arrangement of community type patches. Differences in community types was also determined so that these community differences could suggest an overall heterogeneity within a grazing unit. This methodology could provide a much needed tie of practical inventory information and assessments to landscape diversity measures.

Range health is often touted as a measure of ecological functions including biological diversity (Adams et al. 2011, Pellant et el. 2005, TGUCT 1995). This study evaluates that claim specifically using the Alberta Rangeland health Assessment. Furthermore, relationships between assessed rangeland health, a component of it, or wrapped up at the pasture level and plant diversity would increase the value of the rangeland health assessment. Currently its utility provides a rapid assessment to an overall understanding of certain processes, but this study may also create a means to understand the implications to biological diversity and grazing effects.

2.0 Rangeland health is not a predictor of plant diversity in grasslands of Alberta, Canada.

2.1 Introduction

Declines in biodiversity have been reported globally due to habitat loss (Aguiar 2005, Bennett 2003, Davis 2003). North American grassland ecosystems in particular have declined extensively due to agricultural conversion, industrial use and urban expansion (Bailey et al. 2010, CRC 1994, Greene & Stager 2001, Schieck et al. 2014). Grassland ecosystems provide critical habitat for wildlife, as well as supporting ecosystem functions and processes. The latter enable water capture and storage, biomass (forage) production for wildlife and livestock, nutrient cycling, carbon storage, site stability, and the maintenance of biological diversity (Adams et al. 2016, CRC 1994, Hewins et al. 2018). Land conversion is considered the highest cause of species loss (Aguiar 2005, Bennett 2003), but some suggest that certain land uses can reduce diversity through the degradation of ecosystems (Chillo et al. 2015, Eshel et al. 2014, Fleischner 1994). Remaining native grasslands in North America are often managed extensively for grazing livestock, particularly cattle, and it is critical to understand the effects of grazing management on important ecosystem attributes, such as biodiversity, to ensure these values are conserved. Rangeland health assessments evaluate grazing management effects on rangeland composition and function, and are assumed to correspond with plant diversity, and in turn, overall biodiversity (Adams et al. 2016, CRC 1994, Pellant et al. 2005, TGUCT 1995), although this has not been broadly tested.

A foundational theory relating grazing to plant diversity is the intermediate disturbance hypothesis, which predicts that the highest plant diversity will occur at moderate levels of disturbance (Connell 1978, Grime 1973). However, other evidence suggests that plant communities without evolutionary histories of grazing are more sensitive, while those that co-evolved with grazing exhibit responses to grazing dependent on climatic conditions (Milchunas et al. 1988, Willms et al. 2002). In areas such as the North American Great Plains, higher plant diversity can be maintained with light to moderate grazing intensity (Bai et al. 2001, Collins & Barber 1986, Knapp et al. 1999, Lwiwski et al. 2015). In contrast, when grazing is removed or is excessive, diversity often declines (Collins & Barber 1986, Lwiwski et al. 2015, Willoughby & Alexander 2005).

Rangeland health assessments were developed in response to the need for an effective and pragmatic method of measuring how grazing impacts the rangeland resource, including grassland ecosystems (CRC 1994, TGUCT 1995). Standardization of rangeland health was sought to provide a consistent overall measure of ecological function based on scientific principles that promote sustainability, are based on an ecological land capability classification, include soil conservation criteria, and are evaluated relative to a desired plant community (TGUCT 1995). Assessment protocols were created on these principles in multiple jurisdictions throughout North America (Adams et al. 2016, CRC 1994, Pellant et al. 2005). As an example, the assessment developed in Alberta, Canada (Adams et al. 2016), and used in this study, was developed to measure disturbance and grazing utilization effects on ecological attributes such as plant species composition, habitat structure, hydrological function, nutrient cycling, and soil conservation. Assessments consist of comparing the target community against an expected

reference community maintained within natural disturbance thresholds. It is Important to note that these thresholds include light grazing and implicitly assume that plant community compositional changes occur with increasing or decreasing intensity of grazing (Briske et al. 2008, Collins & Barber 1986).

The native grassland assessment is based on five unequally weighted components with categorical scoring in each (Adams et al. 2016), all of which may have a relationship with plant diversity. Two of these components relate directly to species composition, and are unlikely to be independent of each other. First, the plant community integrity component assesses how similar the dominant species of the remaining community is to that of the reference community, with scoring decreasing as species become less similar to the reference (Adams et al. 2016). In some ecosystems, compositional changes due to grazing may not immediately reduce plant species richness indicating that disturbance tolerant species are co-existing or replacing less tolerant species, until such point that overall plant richness declines with extreme levels of grazing intensity that only a few plant species can tolerate (Grime 1973, Milchunas et al. 1988). Similarly, low to nil levels of grazing may permit competitive exclusion of subordinate species, leading to reduced species richness. Collectively, this may create a decreasing slope, or humped shaped relationship, between this score and plant diversity.

Second, the noxious weed component is also related to plant species composition, but focus is only on those species designated as noxious or prohibited noxious by federal, provincial, or municipal governments (Adams et al. 2016). Invasion of communities by weeds may initially lead to increased species richness (Martin-Fores et al. 2017), although substantial increases in weed abundance are often associated with reductions in plant richness (Belcher &

Wilson 1989, Davis et al. 2005, Hejda et al. 2009, Tilman 1999). This question pertains solely to regulated weeds, however, unregulated non-native introduced species are common in many grasslands of Alberta and may effect multiple components within a rangeland health assessment (Adams et al. 2016). When present, the latter species do not reduce the weed score, but instead affect the integrity component by potentially removing or substituting plant species that otherwise positively contribute to the structure, nutrient cycling, hydrological function, as well as stability of soil (Tilman 1999).

The remaining three components of rangeland health - canopy structure, nutrient cycling / hydrological function, and site stability components - are measures of ecosystem attributes, and score reductions therein may indicate missing or impaired functional groups of plant species (Goswamai et al. 2017, Magurran 2004, Tilman et al. 1997). Canopy structure scores are downgraded if entire vegetation layers are absent relative to the reference community (Adams et. al. 2016). If structure is related to plants occupying different niche spaces, then more structure should indicate greater diversity (Whittaker 1965), and a reduction of structural layers would therefore indicate the loss of diversity. The component addressing nutrient cycling / hydrological function is a measure of residual litter retained in a plant community, which is often considered critical for preserving moisture and conserving organic matter in some ecosystems (Willms et al. 1993, Xiong & Nilsson 1999). Although litter may be reduced by previous years grazing events, even to the point where current annual production is affected, litter induced changes to plant species may take many seasons (Goswami et al. 2017, Willms et al. 1993). Overall climate greatly affects the influence litter has on plant diversity. In mesic locations, less litter tends to be associated with higher species richness and is not

strongly related to plant production (Willms et al. 1986, Xiong & Nilsson 1999). In contrast,

litter in arid regions has limited effects on richness and is strongly related to productivity

(Willms et al. 1986, Willms et al. 2002). Finally, site stability assessments grade the amount of

exposed soil and evidence of erosion (Adams et al. 2016). Soil stability is known to depend on

vegetation structure, and in some cases high plant diversity reduces erosion (Pohl et al. 2009,

Quijas et al. 2010). Thus, low levels of soil erosion and exposure are expected to be associated

with greater plant diversity.

When summed together, the five health components provide an overall rangeland

health score, but on their own these categories allow managers diagnostic insight to facilitate

the interpretation of rangeland (i.e. ecosystem) health (CRC 1994). Rangeland health

categories are used to simplify and communicate scores, and are typically categorized as

satisfactory (>75 %; healthy - H), at risk of degradation (>50 % but below 75 %; healthy with

problems - HWP), or indicate obvious degradation (<50 %; unhealthy - UH) (Adams et al. 2016,

CRC 1994). As every component is weighted differently towards the overall score, each may

uniquely relate to diversity, and may have varied responses to environmental conditions.

Additionally, the overall relationship between rangeland health and plant diversity remains

unclear.

Our research objective was to use an existing native grassland range health data set

paired with detailed vegetation composition data from widely distributed grassland field sites

across Alberta, Canada, to test the relationship between rangeland health and plant diversity.

Specific objectives were 1) to examine the relationship between rangeland health, including its

individual component scores, with plant diversity metrics (richness and diversity), 2) examine

whether these patterns were consistent across local and regional moisture gradients, and 3)

examine the relationship of introduced plant species with rangeland health and diversity.

2.2   Methods

2.2.1   Study Sites

We sampled 79 grasslands distributed across south-central Alberta, Canada (Figure 2.1).

Native grassland sites sampled were part of the Alberta Biodiversity Monitoring Institute's

(ABMI) long-term monitoring program of randomly located sites on an approximate 20 x 20 km

provincial grid (ABMI 2014). Where grid locations fell on annual cropland, ABMI added

grassland sites located near grid locations to their site matrix (known as 'off-grid' locations) to

increase the effective sample size for grasslands, an ecosystem that otherwise would have been

under-represented by the grid. Most sites occurred on flat, loamy textured soils characteristic

of the prairie region across the province (Downing & Pettapiece 2006, Appendix A).

Study sites spanned a range of climatic conditions; 30 year (1984 -2013) mean annual

temperatures (MAT) ranged from 2.2 to 5.9 °C, with a mean of 4.1 °C (SE ±0.1) and median of

4.1 °C. Mean annual precipitation (MAP) ranged from 308.5 mm to 665.6 mm, with a mean of

380.9 mm (±5.9) and median of 357.8 mm (Mbogga et al. 2010). As available moisture is known

to be the primary driver of productivity in grasslands, especially when MAP is below 500 mm

(Sims & Singh 1978), we used annual heat:moisture index (AHM), an index of aridity, to

represent the range of climatic conditions at these sites. AHM 30 year average (1984 -2013)

was calculated using Climate AB ver. 3.21, which interpolates long-term climate data from

weather stations for individual locations (http://tinyurl.com/ClimateAB) (Mbogga et al. 2010).

AHM is calculated from mean annual temperature (MAT) and precipitation (MAP) in the

following formula:

$$(MAT + 10) / (MAP / 1000)$$

AHM therefore accounts for both moisture and temperature differences as the geographic

location changes (Mbogga et al. 2010). A higher number indicates a warm, drier climate,

whereas lower values indicate a cooler and wetter climate. AHM ranged from 22.5 to 53.9

across study sites, and had a mean of 39.5 (±0.6) and median of 39.8.

Similar to climate, study sites also ranged widely in vegetation, and is best described

geographically by natural subregion delineations (Downing & Pettapiece 2006) (Figure 2.1).

Dominant native grassland vegetation ranged from needle-and-thread grass (*Stipa comata* Trin.

and Rupr.), western porcupine grass (*Stipa curtiseta* (A.S. Hitchc.) Barkworth), and blue grama

grass (*Bouteloua gracilis* (HBK.) Lag.) at sites within the Dry Mixed Grass and Mixed Grass

subregions of southeastern Alberta, to plains rough fescue (*Festuca hallii* Vasey) grasslands in

the Northern Fescue and Central Parkland subregions of north-central Alberta. To the west,

sites within the Foothills Fescue and Foothills Parkland subregions were commonly comprised

of foothills rough fescue (*Festuca campestris* Rydb.) grasslands. Sites in the west and north

were often interspersed amongst trees, mostly trembling aspen (*Populus tremuloides* Michx.),

and introduced grasses such as smooth brome (*Bromus inermis* Leyss.) and Kentucky bluegrass

(*Poa pratensis* L.) were common. Crested wheatgrass (*Agropyron pectiniforme* R. and S.) was a

common introduced grass species in the southeast.

2.2.2   Site Stratification and Data Collection

Field surveys were completed by first stratifying the ABMI sites by differences in dominant vegetation, landscape position, and management unit, and then performing a detailed assessment within each stratified area (AEP 2018). Each assessment first entailed the collection of vegetation and physical information. A 30 m linear transect with a random starting location was established in each representative vegetation type, and positioned at a constant landscape position. Plant cover was assessed in ten micro-plots located systematically at 3 m intervals along the transect. Shrub cover was measured at each micro-plot within a 1 m² quadrat, while the cover of all grasses, forbs, and ground components (litter, soil, rock, & bryophytes) were assessed in a 20 x 50 cm quadrat nested within the micro-plot. Physical site information, such as slope, aspect, soil classification, natural subregion, ecological site, and (more broadly) the nearby presence of shrubs and trees, was also recorded along each transect (see Inventory Manual (AEP 2018) for examples of data sheets). Above ground current annual plant biomass (cut approximately 1 cm above the soil surface), standing and detached litter, and shrub current annual growth were collected from three, 0.5 x 0.5 m plots, located systematically along the transect. Samples were dried for at least 48 hrs at 65° C to constant mass, then weighed.

Ecological site (hereafter ecosite) classification provides the reference community vegetation parameters of the desired stable plant community needed to assess rangeland health, as recommended by the TGUCT (1995) through use of site specific soil and landscape attributes. This classification system divides the upland landscape into categories characterized by climate, soil type, texture, and/or landform elements; categories developed represented

repeating soil and landscape attributes that a land manager could readily recognize (Adams et al. 2005, 2013a, 2013b). In Alberta's classification of grasslands a 'range-site' determination is used that once located geographically within a natural subregion, can be analogous to ecosite (Adams et al. 2013a, 2013b, 2003, TGUCT 1995). For example, the ecosite 'Loamy' is common in the Grassland and Parkland natural regions, and represents a moderate sloping to level landscape with loamy textured Chernozemic soils (Adams et al. 2013a). Other ecosites assessed were Blowout, Saline Lowland, Sandy, and Overflow areas, which all occurred at lower frequency (Appendix A).

A native grassland rangeland health assessment was completed (Adams et al. 2016) for each vegetation type at a site (N=124 assessments across 79 sites), with a 'within site' maximum of 4 vegetation types (mean = 1.6 ±0.07). To conduct the assessment, the reference plant community was determined using ecosite information coupled with existing published classifications of grassland vegetation types for each climatic region (i.e. natural subregion) (Adams et al. 2005, 2013a, 2013b, DeMaere et al. 2012, Kupsch et al. 2012, 2013). Each health assessment used a series of questions to address fundamental attributes of plant community composition, canopy structure, hydrology and nutrient cycling, site stability, and noxious weed presence, with different weightings to various questions (Adams et al. 2016). During the assessment, individual components of the rangeland health evaluation were compared to the potential biological and environmental characteristics identified for the reference community (for the known ecosite), and finally component scores were summed to provide a total score of rangeland health (%).

2.2.3   Plant Community Diversity Metrics

For each plant community evaluated, we calculated four metrics, plant species richness, species evenness (as measured by an alpha value), Simpson's diversity, and vegetation heterogeneity (measured via dissimilarity index), using the foliar cover of species observed in the ten micro-plots.

Species richness for each assessment was the total number of different plant species found among all micro-plots within a plant community (i.e., transect). Species evenness was the distribution of the relative abundance of each species that occurred in each assessment (Magurran 2004), calculated using rank abundance distribution in R software (Oksanen et al. 2017, R Core Team 2018). Evenness was calculated by first ranking species from the most to least abundant, then fitting a pre-emption model curve to the relationship (Magurran 2004). The pre-emption model was used as it consistently performed well with the data set of 10 micro-plots. The pre-emption model fitted a line to the log of the rank abundance distribution (Oksanen et al. 2017, Wilson 1991), where the slope of the line (called the alpha value) described the evenness of the plant community. Comparatively high values of alpha indicated a higher rate of decay, and in turn, a higher dominance of only a few species and a rapid reduction in species abundance, whereas lower alpha values indicated less decay and greater evenness. The Simpson's diversity index represented a combination of species richness and evenness (Magurran 2004), and was calculated as follows:

$$D = \frac{\sum n_i \, (n_i - 1)}{N(N - 1)}$$

where n_i is the cover of the ith species and N is the total cover of all individuals. Finally, heterogeneity of composition for each plant community/transect was calculated using a Bray-Curtis dissimilarity matrix created using the foliar cover data for each micro-plot at a site. The "Betadisper" function (Oksanen et al. 2017) in R software was used to provide a mean distance of the 10 micro-plots from their assessment median (Anderson et al. 2006). A greater dissimilarity index value indicated more heterogeneity among micro-plots within an assessment.

2.2.4 Data Analysis

The relationship of species richness, evenness, Simpson's diversity and heterogeneity, with total rangeland health scores, as well as each of the component scores (C-1 – C-5) of the range health assessment, were tested with linear mixed models with sampling site as a random effect. This enabled assessments within a site to be utilized independently, yet account for their relatedness (Bolker et al. 2008), both in soil/landscape properties and overarching climatic conditions. Results of the model were summarized with a type III Analysis of Variance table using the 'Anova' function in R software (Fox & Weisberg 2011).

Both ecosite type and AHM index were included as fixed factors within this analysis. To incorporate ecosite differences, two broad categories based on their landscape position were created. Sandy, Blowout, and Loamy ecosites all occurred on uplands, representing reasonably flat locations and were neither overly water receiving or shedding; and are from here on forward called "uplands" (UL, n = 92). Overflows and Saline Lowlands were not directly in riparian or wetlands zones but represented depression areas that received more moisture, and

were collectively called "lowlands" for this assessment (LL, n = 32)(refer to Appendix A for further ecosite breakdown). AHM was incorporated as a continuous climatic covariate. In cases where AHM and RH were analyzed together, the output of the model suggested AHM needed to be rescaled for balance prior to analysis. All results are reported with $P \leq 0.05$ considered significant, although any trends ($P \leq 0.10$) were also noted.

Differences in plant community composition in relation to rangeland health scores, and the identity of specific plant species that influenced these differences, were analyzed using Non-Metric Multidimensional Scaling and indicator species analysis (Oksanen et al. 2017, Oksanen 2015, De Caceres & Jansen 2016). Permutational multivariate analysis of variance (PERMANOVA) using a Bray-Curtis distance matrix were used to identify whether defined associations, such as range health categories, ecosite groups, or broad geographically defined climatic (i.e., moisture) groupings, were compositionally different (Anderson 2001). As species composition changes have a direct relationship with the range health assessment (Adams et al. 2016), pairwise permutational multivariate analysis of variance identified specific differences between health categories (Hervé 2017, Oksanen et al. 2017). Other available site variables were overlain, including: the various diversity metrics, rangeland health scores, individual component scores, vegetation productivity and litter mass.

For the further analysis of plant species composition, two broadly defined groups were identified, including a xeric group (XE; n= 80) composed of the Mixed Grass and Dry Mixed Grass, and a mesic group (ME; n=44) that included the Foothills Fescue, Northern Fescue, Central Parkland and Foothills Parkland (Downing & Pettapiece 2006). Saline lowlands were

distant outliers in plant community analysis and thus removed in these analyses based on the very specific plant communities known to tolerate high salinity (Adams et al. 2013a, 2013b).

To augment the initial analysis of the broader community data, a separate analysis was performed with introduced species to evaluate their specific relationship to the range health scores. This analysis was completed by first determining their total cover in each plant community, and then quantifying their relative influence on the Simpson's diversity index. Categorization of species (i.e. native vs introduced) was performed using the Flora of Alberta (Moss 1983). The contribution of introduced species within the Simpson's index was calculated by grouping introduced species and calculating the percentage of the total index they contributed (Moss 1983, Magurran 2004). A mixed model was again used to test whether introduced species cover and its effect on Simpson's diversity relationships varied with rangeland health scores, ecosite groupings, and AHM, with sampling site as the random factor.

2.3   Results

2.3.1   Range Health Score and Components

Across all sites range health averaged 76 % (±1.7); the lowest score was 30 % and highest was 100 %, the mode was 100% (n = 19) and the median was 75 %. Categorically this sampling of plant communities indicated 56 % of the assessments were healthy, 34 % were healthy with problems, and 10 % were unhealthy (Table 2.1). Table 2.2 lists the distributional breakdown of component scores across all categorical levels. With the exception of component 1 (C-1), plant community integrity), the most frequent score reported by a substantial margin was the highest

one for most components. Within component 1, the intermediate score (15) was most

frequent, followed closely by each of the higher scores.

The plant community metrics of diversity did not related to the total rangeland health

scores (P ≥ 0.23 for range health and its interactions; Table 2.3), although metrics did relate to a

few of the components that made up the overall health score (Table 2.4). Measures of C-1

(integrity) did not associate with any metrics, alone or in combination with other fixed effects (P

≥ 0.11; Table 2.4a). Component 2 (canopy structure) related to Simpson's diversity in a 3-way

interaction with ecosite groupings and AHM and ecosite groupings themselves (P ≤ 0.04; Table

2.4b). Lowland sites with a component score of 3 had reduced Simpson's diversity at higher

AHM, as did upland sites with a score of 10 (Figure 2.2a, Table 2.5). Lowlands themselves

exhibited reduced Simpson's diversity at the lowest score of C-2 , whereas no distinct

differences occurred in uplands within ecosite groups. However, upland had higher values than

lowland in the top score (Figure 2.2b).

Component 3 (nutrient cycling / hydrological function) scores showed a 3 way trend

with AHM and ecosite grouping (P = 0.09) as well as each were associated with C-3

independently to influence species richness (P ≤ 0.04, Table 2.4c). Although richness declined

with increasing AHM in all significant scoring categories and ecosite groups of the C-3

component, richness decreased more rapidly in the poorest scoring category (i.e., lowland 13,

upland 0) per ecosite group (Table 2.6, Figure 2.3a). This trend was clearly notable with score

categories themselves where low scores began with higher richness at low AHM but decreased

at a faster rate with increased aridity (Figure 2.3b). Species richness was only lowest with the

top score in lowland ecosite (Figure 2.3c). There were no lowlands falling in the lowest scoring

(0) category of C-3. Lastly, component 3 by itself was associated with species richness (P =

0.01, Table 2.4c), although mean comparisons showed no differences (P > 0.10, 16.0 ±1.9, 16.5

±0.80, and 15.4 ±0.60, for the scoring categories of 0, 13, and 25, respectively).

Component 4.1 (site stability – evidence of erosion) did not associate with richness in

this study, but did relate to species evenness, Simpson's index and heterogeneity measures (P ≤

0.04, Table 2.4d) within 3-way and less complex interactions involving AHM and the ecosite

groupings. Alpha values increased (indicating less evenness) with AHM in both the lowland and

upland sites for those sites with a maximum soil erosion score of 10, with larger increases

evident in lowlands compared to uplands (Table 2.7a, Figure 2.4a). In additional 2-way

interactions, evenness decreased (alpha values increased) with increasing AHM for sites

receiving a C-4.1 score of 3 or 10, but not 7 (Table 2.7a, Figure 2.4b). Lower evenness (higher

alpha values) occurred in the bottom scores (i.e., 3 and 7) of lowland ecosite groups, especially

compared to the upland ecosites, although the lower scores in both ecosite groups had low

sample numbers (Figure 2.4c). Although, C-4.1 itself associated with evenness (Table 2.4d),

inspection revealed no mean differences (P > 0.10, 0.32 ±0.06, 0.34 ±0.04, and 0.27 ±0.01, for 3,

7, and 10 scores, respectively). The significant 3-way interaction indicated the Simpson's index

decreased with increasing AHM, but only in lowlands having the highest score (10) for C-4.1

(Table 2.4d, 2.7b, Figure 2.5a). Simpson's index values also decreased in general with

increasing AHM, with a steeper decline occurring for plant communities receiving the lowest

soil erosion score compared to those having the highest (Table 2.7b, Figure 2.5b). The

component 4.1 by ecosite group interaction indicated uplands had higher Simpson's index value

in the lower scores (3, 7) compared to lowlands of the same scores, although sample numbers

were low (Figure 2.5c). The Simpson's index also responded overall to soil erosion by itself although no mean differences were noted (P > 0.10, 0.75 ±0.10, 0.74 ±0.05 and 0.78 ±0.01, for 3, 7 and 10 scores respectively). Lastly, heterogeneity responded to the component 4.1 score as well, but further interacted with AHM and ecosite groupings (Table 2.4d). The 3-way and component 4.1 by AHM interactions did not yield significant regressions (Table 2.7c). However, uplands generally had higher heterogeneity, particularly in areas with greater AHM (Figure 2.6a). The component 4.1 scores by ecosite group interaction indicated higher dissimilarity between upland and lowlands only in the middle score (7) (Figure 2.6b), although the trend (P=0.08) of heterogeneity and C-4 .1 itself indicated differences in the top and bottom scores (P < 0.10, 0.31 ±0.02, 0.32 ±0.02, and 0.34 ±0.01, for categories 3, 7, and 10, respectively, Table 2.4d).   Species diversity measures did not related to component 4.2 (site stability – bare soil) (P ≥ 0.42), nor did the component interact with either ecosite groupings or AHM (P ≥ 0.14, Table 2.4e).

Component 5 (prohibited noxious / noxious weeds) was not associated with species richness or evenness, but did alter vegetation heterogeneity through more complex interactions, as well as a 3-way interaction on Simpson's diversity (P ≤ 0.05, Table 2.4f). These results should be interpreted with caution, as many interactions in both Simpson's and heterogeneity measures were influenced by varying and small sample sizes for this component. The 3-way interaction between ecosite groupings, AHM and C-5 values generally showed decreasing Simpson's diversity values as AHM increased for select upland and lowland scores (Table 2.8a, Figure 2.7a), as did a 2-way interaction between C-5 and AHM (Figure 2.7b). A trend also occurred between C-5 scores and ecosite groupings (P = 0.07, Table 2.4f) where the

bottom three scores in lowlands typically had higher Simpson's values than the top three, with no distinct differences in uplands among scoring categories (Figure 2.7c). Although there was a main-effect between the Simpson's index and the weed component score (P = 0.02, Table 2.4f), there were no differences detected between score means (P > 0.10, 0.79 , 0.77 ±0.07, 0.78 ±0.03, 0.69 ±0.08, 0.69 ±0.08, 0.79 ±0.01, for the 0, 1, 2, 4, 6, & 10 scores, respectively). A significant interaction between all fixed effects and vegetation heterogeneity indicated similar trends of either no relationship, or varying decreases in heterogeneity with increasing AHM, with the exception of the category 2 score in uplands, wherein heterogeneity increased (Table 2.8b, Figure 2.8a). AHM by weed score showed that heterogeneity decreased with increasing AHM within each of the two highest scoring categories for C-5, with the largest decline occurring in category 6 (Figure 2.8b). Heterogeneity indicated an effect between scores, as well as an ecosite grouping by score interaction; where high dissimilarity was noted in the 4 score in upland ecosites, and the 0 score in lowland, and lowest occurred in the 6 score of lowland (Figure 2.8c). Individual scores indicated the 0 and 1 score different from one another (P < 0.10), while all others were in between. Dissimilarity index values were 0.39, 0.29 ±0.03, 0.31 ±0.02, 0.33 ±0.04, 0.29 ±0.04, and 0.35 ±0.01, for the 0, 1, 2, 4, 6 and 10 C-5 component scores, respectively.

2.3.2   Plant Species Composition

Species compositional differences were detected between plant communities sampled based on the resultant range health categories, ecosite groupings and regional (geographic) moisture groups (Table 2.9). Pairwise permutation multivariate analysis of variance for the range health categories further indicated the H, HWP, and UH groupings were all significantly

different in vegetation composition from one another (P ≤ 0.02). Additionally, there was an interaction between range health categories by regional moisture groups that revealed better separation of range health classes in mesic areas (ME) than xeric areas (XE) (Figure 2.9). Productivity, litter, introduced species cover, diversity metrics, range health and its component scores with a significant correlation to the overall ordination (Table 2.10) were added to the ordination plots (Figure 2.9). Visually, range health components C-1 (Integrity), C-2 (structure) and C-5 (weeds, where higher values mean less weeds) were all closely aligned, and associated with the sites that had healthy assessments (Figure 2.9). Component scores for C-3, C-4.1 and C-4.2 were not correlated with the ordination. Total production and litter biomass were visually associated with both the H and HWP groupings, while being negatively associated with the UH grouping, and introduced species cover was associated with HWP and UH. Finally, plant species richness and evenness were correlated to the ordination plot, with richness related to the H and HWP groupings, similar to that of biomass, and alpha to UH (increasing alpha means less evenness, and more likely to be unhealthy). Heterogeneity and Simpson's index did not correlate with the ordination (Table 2.10).

Indicator species analysis highlighted those plant species significantly associated with each range health category group in both the regional XE and ME moisture groupings (Table 2.11). In the XE region, no species were associated with healthy (H) plant communities, and only crested wheatgrass was positively associated with unhealthy (UH) communities. Two species were associated with sites that were healthy with problems (HWP), a non-native grass, Kentucky bluegrass and a single forb - low goldenrod *(Solidago missouriensis* Nutt.*)*. In contrast, within the ME region, several native grasses, including june grass *(Koeleria macrantha* (Ledeb.)

J.A. Schultes*)*, western porcupine grass and plains rough fescue, together with little club-moss

(*Selaginella densa* Rydb.), Canada goldenrod (*Solidago Canadensis* L.), and prairie crocus

(*Anenome patens* L.) were associated with H communities. Also in the ME region, the

disturbance adapted species Sandberg bluegrass (*Poa sandbergii* Vasey), common dandelion

(*Taraxacum officinale* Weber) and gumweed (*Grindelia squarrosa* (Pursh) Dunal) were

indicative of sites that were UH. No species were associated with indicating HWP.

2.3.3   Introduced Species Response

The cover of introduced plant species responded to the interaction between range

health scores and ecosite groupings, as well as with AHM and these main effects individually

(Table 2.12a). The interaction indicated that the cover of introduced species increased with

declining RH scores, and that this increase was generally much greater in lowlands than uplands

(Figure 2.10a). In general, the cover of introduced species decreased as AHM increased (Figure

2.10b), and lowlands had greater cover of introduced species than uplands (Figure 2.10c).

The relative contribution of introduced plant species cover to Simpson's diversity in this

study was influenced by main effects of range health scores, ecosite groupings (trend, P = 0.06)

and AHM (Table 2.12b). Introduced species comprised a greater proportion of Simpson's

diversity at lower range health scores, following a unimodal pattern and peaking in plant

communities with less than 60 % health (Figure 2.11a). Notably, the lowest contribution of

introduced species occurred in communities with health scores above 80%. Overall, the

contribution of introduced species decreased linearly with increasing AHM (Figure 2.11b). The

ecosite group trend means indicated a similar tendency as did the cover of introduced species, however these were not significantly different (P > 0.10, Figure 2.11c).

2.4   Discussion

In this study, plant diversity metrics were not predicted by total rangeland health scores. If score reductions indicate increased disturbance or degrading range condition (i.e. in response to heavier livestock grazing), these findings did not follow the pattern noted in other studies that compare biodiversity and grazing disturbance (Bai et al. 2001, Collins & Barber 1986, Collins 1987, Lwiwski et al. 2015, Willms et al. 2002). However, it is important to note that this comparison may not be appropriate as these previous studies compared specific vascular species composition changes to reductions in diversity, whereas the rangeland health metric used here was comprised of a broader suite of measures. In general, rangeland health assessments have a broader focus on maintaining ecosystem function, which includes soil conservation, nutrient cycling, and the maintenance of vegetation functional structure, all which had varying relationships with diversity in the current study based on the component scores (Adams et al. 2016, CRC 1994, TGUCT 1995).

One reason that overall range health did not relate to diversity metrics may be that C-1 (integrity) did not reflect diversity responses, and this component made up 40 % of the overall score. C-1 (integrity) directly evaluates plant community composition and scores decrease as the former shifts between categories away from the reference community. Hence, a relationship may have been expected given that this question most closely reflects floristic

composition. This result may have arose because as defined in the health assessment protocol, the reference community is not solely comprised of plants that dominate due to an advantage in the undisturbed state for a given climate, but instead includes those plants tolerant of and adapted to grazing. Therefore, directly relating these findings to theories such as those associated with the intermediate disturbance hypothesis may be problematic; instead, these results suggest the upper range of C-1 (integrity) scores are not representative of undisturbed communities, but also include disturbed species and therefore potentially higher levels of inherent diversity (Connell 1978, Grime 1973, Milchunas et al. 1988, Sasaki et al. 2009). Given this, an obvious flaw in Alberta's range health system and its ability to be linked to diversity may be the upper integrity score category, which includes completely undisturbed communities (Willoughby & Alexander 2005), resulting in a blend of non-disturbed and minimally disturbed vegetation within this coarse category of 'healthy' communities.

Another reason for the lack of a relationship of plant diversity to overall rangeland health and C-1 (integrity) scores may have been that although there was evidence that species composition shifted as scores declined this may not have readily decreased diversity. The lack of a response may be that this component is too coarse of a measure as it only notes changes in dominant plant species, whereas diversity measures are often driven by intermediate and less prominent species (Whittaker 1965). For example, non-native introduced species rapidly increased their cover and contribution to diversity as the scores declined, especially below 80%. Reference communities in native grasslands are expected to have very low amounts of non-native species, so in all cases the presence of introduced species would have prompted substantial scoring reductions to component 1 (Adams et al. 2016). These compositional

changes however, may not necessarily affect other components so readily as introduced

species may have replaced some native species by occupying similar niches (Tilman 1999),

thereby stabilizing other attributes of health (i.e., structure, nutrients cycling / hydrological

function, site stability). This was evident in assessments where significant amounts of

introduced species occurred, yet overall health remained near 70 %. Only when entire

vegetation layers, litter, or site stability were affected did assessments produce differences in

diversity. At the lowest scores, even introduced species began to decline as well. There is

significant debate whether competitive species (rather than predation such as herbivory)

causes the removal of native species (Davis 2003, Gurevitch & Padilla 2004). In this study, it

appeared that especially in more mesic environmental conditions (i.e., lowlands and/or

reduced aridity) and those with moderate health, introduced species outcompeted some

dominant native plants, but did not overtake the entire community, thereby maintaining (on

balance) most function and associated diversity. This coincides with some theories that view

invasibility of communities a factor of their environmental conditions, although overall

susceptibility to invasive plants may be reduced due to limiting resources available in a diverse

functioning community (Davis et al. 2005, Tilman 1999).

Despite the low sensitivity of vegetation diversity metrics to overall range health and C-

1 (integrity), several individual components of the health assessment did relate to plant

diversity. In this evaluation, the range health components that related to diversity (i.e., C-2, C-

3, C-4.1, and C-5) compared expected functional processes of those exhibited by the reference

community, and were more so based on their perceived losses or reductions (Adams et al.

2016). For example, Simpson's index was only reduced significantly once two or more

structural layers were removed from the plant community, a pattern that was only prevalent in lowlands. This component (C-2) may not follow a linear relationship with conventional diversity metrics, as the loss of only a few species with certain functional traits will reduce this score, such as one or two major species that occupy a given structural layer (i.e., a shrub layer composed of one or two species). Nevertheless, as diversity metrics are regulated by all species including those considered rare, the overall diversity is still likely to be high, especially in mesic, species-rich environments (Whittaker 1965). However once this layer is removed and no other species occupy this strata, overall functions of the community may be impaired (Goswami et al. 2017, Magurran 2004, Tilman et al. 1997, Tilman 1999).

Unlike the components evaluating vegetation characteristics directly, those assessing soil conditions such as site stability (i.e. C-4.1 evidence of erosion), appeared to be more sensitive to detecting diversity changes, although small sample sizes due to only a few low scoring assessments may have influenced these results. Observed responses in C-4.1 were influenced by interactions with regional and local growing conditions (i.e. AHM and ecosite, respectively), however, a reduced score for evidence of erosion was associated with notable decreases in vegetation evenness and Simpson's diversity, particularly in drier climates. Conservation of soils and stability of plant communities are considered benefits of high plant diversity through greater soil cohesion from increased above and below ground biomass (Pohl et al. 2009, Quijas et al. 2010, Tilman et al. 1997). Should this be the case, evidence of soil instability may also be an indication that plant diversity has been reduced, and was supported here.

Changes in nutrient cycling / hydrological function were less clear as an indicator of changes in vegetation diversity. As this component (C-3) essentially quantified litter abundance, this metric alone may not be suitable for characterizing the mechanistic impact of changes in abiotic conditions (i.e., soil moisture and temperature through litter modification) on observed plant diversity. Our richness measures followed previous studies showing that in drier areas plant diversity and productivity decreases with lower litter, but this pattern reverses in more moist areas, with richness increasing with decreased litter (Willms et al. 1986, Xiong & Nilsson 1999).   As productivity and moisture retention are directly related in arid areas (Willms et al. 1986), and productivity is also related with diversity (Dorji et al. 2014, Tilman 1999), reducing productivity through the removal of litter could reduce overall plant diversity in dry areas. In moister areas however, reduced diversity is more likely to occur when high moisture leads to excessively high litter levels, which after an extended accumulation period, reduces light availability, alters germination, and limits subsequent opportunities for plant growth, particularly for early and mid-seral plant species (Lamb 2008, Marty 2015, Willoughby & Alexander 2005).

Although there were relationships between the noxious weed component (C-5) score and plant diversity, their relationship remained unclear in this study. Weeds in this assessment were defined by very specific plant species within the community as they were limited to prohibited noxious and noxious weeds (i.e. nuisance/agronomic weeds were excluded). Hence, lower scores for this component solely reflected an increase in noxious weeds (Adams et al. 2016). While this change can effect species richness once noxious weeds dominate a site (Hejda et al. 2009), the scoring of this component is based on early detection of serious weeds

(i.e. 1 %), and our results seemed to indicate that even the lowest scores were attained before distinct diversity changes were evident. In other words, similar to the pattern observed for introduced species, the presence of noxious weeds themselves alone appeared to have limited impact on the diversity metrics assessed, and instead the diversity of these native grasslands appeared to be resistant to small influences of noxious weeds per-se. It is important to note that the sites assessed in this study were not overly infested with noxious weeds, as there were only one case in this study where noxious weed cover exceeded 15% and led to the lowest score for component C-5.

Some studies have shown differences in plant community heterogeneity induced by grazing (Fuhlendorf & Smeins 1999, Fuhlendorf & Engle 2004, Lwiwksi et al. 2015, Virk & Mitchel 2014). In this study however, heterogeneity was only related to ecosite groupings where more dissimilarity occurred in upland locations. In contrast, lowlands were relatively homogeneous in composition, which may be explained by these areas having consistently fewer species that generally had higher dominance, and were more adept at utilizing the favorable and uniform resources (water and nutrients) at these locations. The lack of overall response in heterogeneity to grazing may also be due to sampling design and the makeup of the heterogeneity metric. Heterogeneity was assessed as the dissimilarity between adjacent micro-plots sampled on each transect, yet rangeland inventory methods require that transects be situated in areas with relatively consistent vegetation (i.e. uniform 'plant communities') *a-priori* to clearly separate different ecosites for sampling (AEP 2018). These stratified locations may have reduced the opportunity to measure heterogeneity, with ten micro-plots possibly being insufficient to assess plant community heterogeneity.

In this study the relationship of diversity metrics with range health components were subject to multiple interactions with both the regional and local growing conditions that constrained where and how well individual range health components accounted for differences in plant diversity. For example, contrasting moisture regimes had different effects on diversity. Geographically, an increase in regional moisture supply (as evidenced by lower AHM) showed generalized increases in diversity metrics. This response however, differed from that at the local topographic level, where (more mesic) lowlands were generally lower in most diversity measures. Moisture effects at both levels were ultimately found to be influential on plant richness, diversity, and evenness, as well as account for many interactions with range health components throughout this study. These results are consistent with previous studies and theories, where induced effects to diversity are caused by immediate moisture affecting the response of plants to disturbance , and broad climatic changes strongly influencing the type of vegetation and its response to disturbance (Dorji et al. 2014, Moeslund et al. 2013, Milchunas et al. 1988, Sasaki et al. 2009, Willms et al. 2002).

A limitation of this project was the lack of spread in range health scores between assessments across the regional and local growing conditions, causing low to none in sample sizes for many of the low component scores. Although this random sampling of native grassland sites provided good news that Alberta's grasslands are typically healthy or scored in the upper portion of healthy with problems, this did not provide a great representation for diversity measures with lower range health. However, for interpreting plant diversity relationships, this study did show that rather than assess plant species composition alone, it

seems prudent to look at other factors when trying to assess diversity responses, especially the structure of vegetation, litter mass, and site stability.

2.5 Conclusion

Overall rangeland health scores were relatively poorly associated with plant diversity across the 124 grasslands distributed among 79 sites in Alberta grasslands of this study. Unexpectedly, the integrity component that compares current plant species composition to a potential reference community did not reveal any impact on diversity metrics, possibly because Alberta's rangeland health assessment protocol considers light disturbance impacts to be intrinsic to the reference community (Adams et al. 2016), or potentially due to the insensitivity of the utilized diversity metrics to species composition changes.

Other components of the rangeland health assessment that dealt with non-vegetation characteristics of the grassland (i.e. soil and plant residue) were found to relate to observed plant diversity measures in this study, indicating the broader criteria used in the existing rangeland health assessment (i.e. compared to traditional plant ecology practices such as those used in Clementsian successional evaluation) may provide a metric to assess plant diversity. Site stability, canopy structure, hydrological function / nutrient cycling, and noxious weeds each interacted with environmental conditions, namely regional and local growing conditions, to influence observed metrics of plant diversity. This study suggests that in order to understand grassland diversity relationships with rangeland health, the overall assessment score should not

be the focus, but rather the individual component scores and the influence of environmental

factors that make up the assessment.

To accommodate diversity specifically into rangeland health, a more detailed criteria

may be required in the health assessment protocol that incorporates and promotes vegetation

characteristics reflective of plant diversity changes. Also, better descriptions of the diversity

effects from non-vegetative components should be considered and incorporated into the

assessment guide.

Table 2.1. Frequency of range health occurrences per category for all assessments completed. Categories were split in half to further highlight variation in health scores.

	Range Health Score	n
Healthy	87.5 - 100	35
	75 - 87.4	35
Healthy with Problems	62.5 - 74.9	26
	50-62.4	16
Unhealthy	25-50	12
	0-24.9	0
Total		**124**

Table 2.2. Frequency counts (n) for each range health component categorical score for all the assessments completed. The components evaluated are C-1 = plant community integrity, C-2 = canopy structure, C-3= nutrient cycling / hydrological function, C-4.1 = site stability – evidence of erosion, C-4.2 = site stability – soil exposure, and C-5 = prohibited noxious / noxious weeds. The maximum score in each category is underlined.

	Component 1		Component 2		Component 3		Component 4.1		Component 4.2		Component 5	
	Scores	Stats	Scores	Stats	Scores	Stats	Scores	Stats	Scores	Stats	Scores	Stats
N	**0**	0	**0**	0	**0**	14	**0**	0	**0**	0	**0**	1
	8	5	**3**	17	**13**	37	**3**	3	**3**	8	**1**	5
	15	39	**7**	37	**$\underline{25}$**	73	**7**	14	**$\underline{5}$**	116	**2**	8
	20	4	**$\underline{10}$**	70	-	-	**$\underline{10}$**	107	-	-	**4**	6
	27	38	-	-	-	-	-	-	-	-	**6**	7
	$\underline{40}$	38	-	-	-	-	-	-	-	-	**$\underline{10}$**	97
Total		124		124		124		124		124		124

Table 2.3. Results of mixed model testing for the fixed effects of range health score (RH), annual:heat moisture (AHM) index, and ecosite groupings (ESG), on plant species richness, evenness, Simpson's index, and vegetation heterogeneity. No relationships were significant (P > 0.05). AHM was rescaled (AHM_s) to more closely match range health.

		Species richness		Evenness		Simpson's		Heterogeneity	
Effect(s)	Df	Wald F	Pr(>F)	Wald F	Pr(>F)	Wald F	Pr(>F)	Wald F	Pr(>F)
RH	1	0.91	0.34	0.96	0.33	0.53	0.47	0.11	0.74
AHM_s	1	2.77	0.10	0.85	0.36	0.13	0.72	0.09	0.77
ESG	1	0.42	0.52	0.14	0.71	2.01	0.16	0.00	0.96
RH:AHM_s	1	0.25	0.62	0.00	0.99	0.04	0.83	0.24	0.63
RH:ESG	1	1.46	0.23	0.22	0.64	0.55	0.46	0.86	0.35
AHM_s:ESG	1	0.05	0.82	0.13	0.72	1.91	0.17	0.89	0.35
RH:AHM_s:ESG	1	0.09	0.76	0.01	0.92	1.00	0.32	0.86	0.36

Table 2.4. Results of mixed model testing for the fixed effects of individual component score (C), annual:heat moisture (AHM) index, and ecosite grouping (ESG), on each of plant species richness, evenness , Simpson's index and heterogeneity. The component scores of range health are C-1 = plant community integrity, C-2 = canopy structure, C-3= nutrient cycling / hydrological function, C-4.1 = site stability – evidence of erosion, C-4.2 = site stability – soil exposure ,and C-5 = prohibited noxious / noxious weeds. Bold values indicate a significant relationship (P < 0.05).

a) Component 1 (Vegetation Integrity - based on composition)							
Response:		C	AHM	ESG	C_:AHM	C_:ESG	AHM:ESG	C_:AHM:ESG
Species Richness	F	1.70	0.61	2.95	1.44	0.93	3.73	1.06
	Df	4	1	1	4	3	1	2
	Df.res	87.39	80.36	106.35	86.07	77.98	105.53	78.62
	Pr(>F)	0.16	0.44	0.09	0.23	0.43	0.06	0.35
Evenness	F	0.32	0.95	0.90	0.32	0.04	1.26	0.00
	Df	4	1	1	4	3	1	2
	Df.res	105.31	106.89	106.81	105.51	104.61	106.77	103.69
	Pr(>F)	0.87	0.33	0.34	0.86	0.99	0.27	1.00
Simpson's Index	F	0.53	0.03	0.07	0.42	0.64	0.11	0.62
	Df	4	1	1	4	3	1	2
	Df.res	104.38	106.97	106.70	104.55	105.66	106.54	105.16
	Pr(>F)	0.72	0.86	0.79	0.79	0.59	0.74	0.54
Heterogeneity	F	1.91	0.85	0.00	1.79	1.35	0.02	0.15
	Df	4	1	1	4	3	1	2
	Df.res	104.81	103.57	106.97	104.70	99.23	107.00	97.65
	Pr(>F)	0.11	0.36	0.96	0.14	0.26	0.90	0.86
b) Component 2 (Canopy Structure)							
Species Richness	F	0.69	21.09	0.49	0.32	0.06	1.12	0.08
	Df	2	1	1	2	2	1	2
	Df.res	108.13	97.06	99.10	108.75	91.15	104.07	91.66
	Pr(>F)	0.50	**<0.001**	0.48	0.73	0.94	0.29	0.92
Alpha	F	0.02	20.52	1.93	0.24	0.57	4.36	0.71
	Df	2	1	1	2	2	1	2
	Df.res	106.63	78.82	111.85	106.56	107.20	111.52	107.22
	Pr(>F)	0.98	**<0.001**	0.17	0.79	0.56	**0.04**	0.49
Simpson's Index	F	0.32	10.71	4.80	0.48	3.38	8.33	4.44
	Df	2	1	1	2	2	1	2
	Df.res	103.69	74.58	111.97	103.05	108.26	110.20	108.15
	Pr(>F)	0.73	**0.002**	**0.03**	0.62	**0.04**	**0.005**	**0.01**
Dissimilarity	F	0.15	0.92	0.01	0.09	0.62	0.50	0.63
	Df	2	1	1	2	2	1	2
	Df.res	105.20	76.65	111.99	104.86	107.81	110.94	107.77
	Pr(>F)	0.86	0.34	0.93	0.91	0.54	0.48	0.53

Table 2.4 (continued). Results of mixed model testing for the fixed effects of individual component score (C), annual:heat moisture (AHM) index, and ecosite grouping (ESG), on each of plant species richness, evenness , Simpson's index and heterogeneity. The component scores of range health are C-1 = plant community integrity, C-2 = canopy structure, C-3= nutrient cycling / hydrological function, C-4.1 = site stability – evidence of erosion, C-4.2 = site stability – soil exposure ,and C-5 = prohibited noxious / noxious weeds. Bold values indicate a significant relationship (P < 0.05).

c)	Component 3 (Hydrological Function / Nutrient Cycling)						
Response:		C	AHM	ESG	C:AHM	C_:ESG	AHM:ESG	C_:AHM:ESG
Species Richness	F	4.75	24.65	5.11	4.56	4.12	5.34	2.88
	Df	2	1	1	2	1	1	1
	Df.res	113.40	96.16	112.23	112.32	113.99	111.64	113.84
	Pr(>F)	**0.01**	**<0.001**	**0.03**	**0.01**	**0.04**	**0.02**	0.09
Evenness	F	1.51	7.99	1.03	1.64	0.32	1.20	0.01
	Df	2	1	1	2	1	1	1
	Df.res	98.56	51.12	89.70	87.86	100.30	76.16	88.30
	Pr(>F)	0.23	**0.01**	0.31	0.20	0.58	0.28	0.91
Simpson's Index	F	1.03	1.60	0.99	1.27	0.07	1.15	0.00
	Df	2	1	1	2	1	1	1
	Df.res	98.56	51.12	89.70	87.86	100.30	76.16	88.30
	Pr(>F)	0.36	0.21	0.32	0.29	0.79	0.29	0.99
Heterogeneity	F	0.38	0.93	0.44	0.31	1.05	0.59	0.47
	Df	2	1	1	2	1	1	1
	Df.res	99.99	53.64	91.44	90.05	101.68	78.75	90.48
	Pr(>F)	0.69	0.34	0.51	0.74	0.31	0.45	0.49
d)	Component 4.1 (Site Stability – Evidence of erosion)						
Species Richness	F	2.19	0.06	1.22	2.04	1.49	2.34	2.55
	Df	2	1	1	2	2	1	1
	Df.res	110.42	112.99	110.68	110.62	109.96	106.60	106.53
	Pr(>F)	0.12	0.81	0.27	0.13	0.23	0.13	0.11
Evenness	F	11.38	0.70	14.03	10.78	10.14	18.41	19.65
	Df	2	1	1	2	2	1	1
	Df.res	112.69	112.95	112.66	112.71	112.81	112.99	112.99
	Pr(>F)	**<0.001**	0.40	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Simpson's Index	F	5.21	0.16	5.45	**5.07**	**4.40**	7.45	**8.23**
	Df	2	1	1	**2**	**2**	1	**1**
	Df.res	112.68	112.94	112.60	**112.69**	**112.81**	112.97	**112.98**
	Pr(>F)	**0.007**	0.69	**0.02**	**0.008**	**0.01**	**0.007**	**0.005**
Dissimilarity	F	2.55	0.14	4.52	2.35	3.37	6.26	6.39
	Df	2	1	1	2	2	1	1
	Df.res	112.68	112.94	112.60	112.69	112.81	112.97	112.98
	Pr(>F)	0.08	0.71	**0.04**	0.10	**0.04**	**0.01**	**0.01**

Table 2.4 (continued). Results of mixed model testing for the fixed effects of individual component score (C), annual:heat moisture (AHM) index, and ecosite grouping (ESG), on each of plant species richness, evenness , Simpson's index and heterogeneity. The component scores of range health are C-1 = plant community integrity, C-2 = canopy structure, C-3= nutrient cycling / hydrological function, C-4.1 = site stability – evidence of erosion, C-4.2 = site stability – soil exposure ,and C-5 = prohibited noxious / noxious weeds. Bold values indicate a significant relationship ($P < 0.05$).

Response:		C	AHM	ESG	C:AHM	C_:ESG	AHM:ESG	C_:AHM:ESG
e)	**Component 4.2 (Site Stability - Soil exposure)**							
Species Richness	F	0.44	3.29	0.19	0.11	0.04	1.12	
	Df	1	1	1	1	1	1	0
	Df.res	116.76	115.47	79.26	116.97	53.56	88.15	0.00
	Pr(>F)	0.51	0.07	0.67	0.74	0.85	0.29	
Evenness	F	0.05	2.56	0.46	0.00	0.00	2.61	
	Df	1	1	1	1	1	1	0
	Df.res	116.56	101.85	93.58	111.75	63.14	107.04	0.00
	Pr(>F)	0.82	0.11	0.50	0.99	1.00	0.11	
Simpson's Index	F	0.69	0.15	0.14	0.48	0.91	2.17	
	Df	1	1	1	1	1	1	0
	Df.res	116.23	98.02	95.57	109.42	65.16	109.31	0.00
	Pr(>F)	0.41	0.70	0.70	0.49	0.34	0.14	
Heterogeneity	F	0.05	0.27	0.01	0.07	0.00	0.26	
	Df	1	1	1	1	1	1	0
	Df.res	116.23	98.02	95.57	109.42	65.16	109.31	0.00
	Pr(>F)	0.82	0.60	0.92	0.78	0.95	0.61	
f)	**Component 5 (Prohibited Noxious / Noxious Weeds)**							
Species Richness	F	0.78	0.36	1.64	0.54	1.70	1.42	1.62
	Df	5	1	1	4	4	1	4
	Df.res	85.82	75.63	73.24	92.57	90.77	73.11	92.20
	Pr(>F)	0.57	0.55	0.20	0.70	0.16	0.24	0.18
Evenness	F	1.57	0.09	1.45	1.12	0.96	1.06	0.99
	Df	5	1	1	4	4	1	4
	Df.res	95.83	102.99	102.99	97.48	100.97	102.99	100.24
	Pr(>F)	0.17	0.76	0.23	0.35	0.44	0.30	0.42
Simpson's Index	F	2.71	1.27	2.14	2.61	2.28	2.34	2.54
	Df	5	1	1	4	4	1	4
	Df.res	95.83	102.99	102.99	97.48	100.97	102.99	100.24
	Pr(>F)	**0.02**	0.26	0.15	**0.04**	0.07	0.13	**0.04**
Heterogeneity	F	2.91	0.02	4.98	2.50	2.82	4.94	3.04
	Df	5	1	1	4	4	1	4
	Df.res	93.02	95.13	94.06	97.72	97.80	93.97	98.19
	Pr(>F)	**0.02**	0.90	**0.03**	**0.05**	**0.03**	**0.03**	**0.02**

Table 2.5. Modelled linear relationships between Simpson's diversity and annual:heat moisture (AHM) index for each of the Component 2 (canopy structure) scores, further separated by the two ecosite groupings (ESG); lowlands (LL) and uplands (UL). Maximum scoring for C-2 is 10.

Simpson's Diversity	Grouping	n	Estimate	Std. Error	t value	Pr(>|t|)	r²
C2 – AHM	LL 3	5	-0.027	0.009	-3.1	**0.053**	0.763
(by ESG x Component	LL 7	11	-0.007	0.008	-0.9	0.390	0.083
2 scores)	LL 10	16	-0.006	0.006	-1.0	0.336	0.066
	UL 3	12	0.006	0.006	1.0	0.348	0.089
	UL 7	26	-0.003	0.004	-0.8	0.415	0.028
	UL 10	54	-0.005	0.001	-4.0	**<0.001**	0.230

Table 2.6. Within the Component 3 (nutrient cycling / hydrological function), modelled linear relationships with species richness with annual heat:moisture index (AHM), further stratified by component scores (0, 13 and 25). Analysis is also shown separately for the various ecosite groupings (ESG) x Component score combinations due to a 3-way interaction trend (P = 0.09, Table 2.4c). Maximum scoring for C-3 is 25.

Species Richness	Scores	N	Estimate	Std. Error	t value	Pr(>|t|)	r^2
C3 – AHM	0	14	-0.774	0.230	-3.4	**0.006**	0.485
(by Component 3 scores)	13	37	-0.250	0.113	-2.2	**0.033**	0.123
	25	73	-0.353	0.076	-4.6	**<0.001**	0.232
C3 – ESG - AHM	LL 0	0	NA				
(by ESG x Component 3	LL 13	9	-0.526	0.142	-3.7	**0.008**	0.663
scores)	LL 25	23	-0.391	0.106	-3.7	**0.001**	0.394
	UL 0	14	-0.774	0.230	-3.4	**0.006**	0.485
	UL 13	28	-0.156	0.143	-1.1	0.285	0.044
	UL 25	50	-0.356	0.085	-4.2	**<0.001**	0.266

Table 2.7. Within the Component 4.1 data (Site stability – evidence of erosion), modelled linear relationships between (a) evenness, (b) Simpson's diversity, (c) and heterogeneity with annual heat:moisture index (AHM), further stratified by component scores (3, 7 and 10). Analysis is also shown separately for the various ecosite groupings (ESG) x Component score combinations due to the 3-way interactions being significant (P < 0.01, Table 2.4d). Maximum scoring for C-4.1 is 10.

a) Evenness	Scores	N	Estimate	Std. Error	t value	Pr(>|t|)	r²
C4.1 – AHM	3	3	0.013	0.001	11.1	0.057	0.992
(by Component 4.1 scores)	7	14	0.003	0.008	0.5	0.655	0.017
	10	107	0.004	0.001	3.9	**<0.001**	0.128
C4.1 – ESG - AHM	LL 3	1	NA				
(by ESG x Component 4.1	LL 7	4	-0.097	0.030	-3.3	0.081	0.845
scores)	LL 10	27	0.007	0.003	2.8	**0.011**	0.234
	UL 3	2	0.035	NA	NA	NA	1.000
	UL 7	10	0.001	0.006	0.2	0.876	0.003
	UL 10	80	0.003	0.001	3.3	**0.002**	0.121
b) Simpson's Diversity							
C4.1 – AHM	3	3	-0.021	0.002	-8.3	0.077	0.985
(by Component 4.1 scores)	7	14	-0.002	0.008	-0.2	0.840	0.004
	10	107	-0.004	0.002	-2.1	**0.037**	0.041
C4.1 - ESG – AHM	LL 3	1	NA				
(by ESG x Component 4.1	LL 7	4	0.099	0.054	1.8	0.208	0.627
scores)	LL 10	27	-0.008	0.004	-1.9	0.067	0.128
	UL 3	2	-0.066	NA	NA	NA	1.000
	UL 7	10	0.003	0.005	0.6	0.593	0.037
	UL 10	80	-0.002	0.002	-1.3	0.192	0.022
c) Heterogeneity							
C4.1 – AHM	3	3	-0.004	0.001	-3.1	0.201	0.904
(by Component 4.1 scores)	7	14	-0.006	0.004	-1.5	0.171	0.150
	10	107	0.000	0.001	-0.2	0.817	0.001
C4.1 - ESG – AHM	LL 3	1	NA				
(by ESG x Component 4.1	LL 7	4	0.047	0.021	2.3	0.149	0.724
scores)	LL 10	27	-0.001	0.002	-0.5	0.634	0.009
	UL 3	2	-0.030	NA	NA	NA	1.000
	UL 7	10	-0.004	0.002	-1.6	0.148	0.243
	UL 10	80	0.000	0.001	0.0	0.991	<0.001

Table 2.8. Within the Component 5 data (Prohibited noxious / noxious weeds), modelled linear relationships between (a) Simpson's diversity, and (b) heterogeneity, with annual heat:moisture index (AHM), further stratified by component scores (0,1,2,4,6, and 10). Analysis is also shown separately for the various ecosite Groupings (ESG) x component score combinations due to the 3-way interactions being significant (P < 0.04, Table 2.4f).

a) Simpson's Diversity	Scores	n	Estimate	Std. Error	t value	Pr(>|t|)	r²
C5 – AHM	0	1	NA				
(by Component 5 scores)	1	5	0.005	0.010	0.46	0.677	0.066
	2	8	0.000	0.004	0.12	0.912	0.002
	4	6	-0.016	0.013	-1.26	0.275	0.285
	6	7	-0.024	0.005	-4.89	**0.005**	0.827
	10	97	-0.004	0.002	-2.20	**0.030**	0.048
C5 - ESG – AHM	LL 0	1	NA				
(by ESG x Component 5 scores)	LL 1	3	0.004	0.001	3.99	0.156	1.000
	LL 2	3	-0.003	0.000	-6.51	0.097	0.977
	LL 4	3	-0.034	0.012	-2.82	0.217	0.889
	LL 6	2	0.005	NA	NA	NA	1.000
	LL 10	20	-0.011	0.007	-1.48	0.157	0.108
	UL 0	0	NA				
	UL 1	2	-1.655	NA	NA	NA	NA
	UL 2	5	0.002	0.008	0.24	0.824	0.019
	UL 4	3	-0.009	0.014	-0.68	0.620	0.316
	UL 6	5	-0.031	0.004	-7.81	**0.004**	0.953
	UL 10	77	-0.003	0.002	-1.50	0.139	0.029
b) Heterogeneity							
C5 – AHM	0	1	NA				
(by Component 5 scores)	1	5	0.004	0.004	0.91	0.428	0.218
	2	8	0.003	0.002	1.68	0.144	0.320
	4	6	-0.007	0.007	-0.99	0.381	0.195
	6	7	-0.010	0.004	-2.75	**0.040**	0.603
	10	97	-0.002	0.001	-1.84	0.069	0.034
C5 - ESG – AHM	LL 0	1	NA				
(by ESG x Component 5 scores)	LL 1	3	0.003	0.002	2.13	0.280	0.819
	LL 2	3	0.001	0.003	0.27	0.833	0.068
	LL 4	3	-0.017	0.000	-42.68	**0.015**	1.000
	LL 6	2	0.014	NA	NA	NA	1.000
	LL 10	20	-0.003	0.003	-0.84	0.414	0.037
	UL 0	0	NA				
	UL 1	2	-0.655	NA	NA	NA	1.000
	UL 2	5	0.006	0.002	2.91	0.062	0.738
	UL 4	3	-0.003	0.012	-0.22	0.863	0.045
	UL 6	5	-0.013	0.003	-4.68	**0.018**	0.880
	UL 10	77	-0.002	0.001	-1.37	0.176	0.024

Table 2.9. Permutational multivariate analysis of variance (PERMANOVA) results evaluating plant species composition responses in relation to range health categories (RHC), ecosite groupings (ESG), and geographic moisture gradient groups created by amalgamating natural subregions (NSRG). Bold values indicate a significant relationship (P < 0.01).

	Df	SSq	MSq	F.Model	r^2	Pr(>F)
RHC	2	5.14	2.57	9.58	0.13	**0.001**
ESG	1	1.56	1.56	5.81	0.04	**0.001**
NSRG	1	2.45	2.45	9.12	0.06	**0.001**
RHC:NSRG	1	0.30	0.30	1.11	0.01	0.33
RHC:NSRG	2	1.21	0.60	2.26	0.03	**0.003**
ESG:NSRG	1	0.37	0.37	1.38	0.01	0.17
RHC:ESG:NSRG	1	0.30	0.30	1.11	0.01	0.36
Residuals	108	28.94	0.27		0.72	
Total	117	40.25			1.00	

Table 2.10. Relationship of aboveground peak productivity, litter, range health, component scores, introduced species cover and diversity measures to the plant community ordination (Figure 2.9). Components are: 1 = plant community integrity, 2 = structure, 3 = nutrient cycling / hydrological function, 4.1 = site stability – evidence of erosion, 4.2 = site stability – soil exposure, and 5 = prohibited noxious / noxious weeds. Litter values are the mean litter mass collected per assessment, production is the mean total amount of biomass collected (grass, forb, and shrub current annual growth). Bold values indicate a significant relationship (P < 0.01).

	r^2	Pr(>r)
RH (range health)	0.31	**0.001**
Component 1	0.41	**0.001**
Component 2	0.32	**0.001**
Component 3	0.04	0.15
Component 4.1	0.01	0.73
Component 4.2	0.02	0.34
Component 5	0.21	**0.001**
Litter	0.22	**0.001**
Production	0.29	**0.001**
Introduced Species Cover	0.61	**0.001**
Species Richness	0.12	**0.002**
Evenness (alpha)	0.05	0.06
Simpson's Index	0.01	0.58
Heterogeneity	0.02	0.32

Table 2.11. Indicator species analysis (R: Indespecies) results for each of the dry (XE) and mesic (ME) geographic moisture gradient groups created by amalgamating natural subregions. Only species P ≤ 0.05 are shown.

XE			ME		
Healthy (n=51)	**stat**	**p**	**(n=17)**	**stat**	**p**
			June grass		
<None>			(*Koeleria macrantha)*	0.82	0.003
			western porcupine grass		
			(*Stipa curtiseta*)	0.76	0.01
			Plains rough fescue		
			(*Festuca hallii)*	0.64	0.02
			little club-moss		
			(*Selaginella densa)*	0.68	0.01
			Canada Goldenrod		
			(*Solidago Canadensis)*	0.56	0.05
			Prairie Crocus		
			(*Anenome patens)*	0.54	0.04
Healthy with Problems (n=16)			**(n=22)**		
Kentucky Bluegrass					
(Poa pratessis)	0.80	<0.001	<none>		
low goldenrod					
(*Solidago missouriensis)*	0.51	0.03			
Unhealthy (n=7)			**(n=5)**		
crested wheatgrass			Sandberg bluegrass		
(*Agropyron pectiniforme*)	0.50	0.03	(*Poa sandbergii*)	0.67	0.01
			common dandelion		
			(*Taraxacum officinale)*	0.78	0.02
			gumweed		
			(*Grindelia squarrosa*)	0.70	0.01

Table 2.12. Results of mixed model testing for the fixed effects of range health score, AHM, and ecosite groupings (ESG) on (a) total introduced plant species cover, and (b) the relative contribution of introduced species to the Simpson's index. Bold values indicate a significant relationship (p<0.05). AHM was rescaled (AHM_s) to more closely match range health.

	Df	(a) Introduced Cover			(b) Simpson's Index Influence		
		F	Df.res	Pr(>F)	F	Df.res	Pr(>F)
RH	1	27.70	110.16	**<0.001**	26.02	97.70	**<0.001**
AHM_s	1	5.35	104.57	**0.02**	5.65	92.49	**0.02**
ESG	1	8.80	112.49	**0.003**	3.60	116.00	0.06
RH:AHM_s	1	1.93	109.20	0.17	2.60	100.43	0.11
RH:ESG	1	5.17	109.87	**0.02**	2.53	115.40	0.11
AHM_s:ESG	1	0.14	108.76	0.71	0.13	114.28	0.72
RH:AHM_s:ESG	1	0.27	106.14	0.60	0.09	112.41	0.77
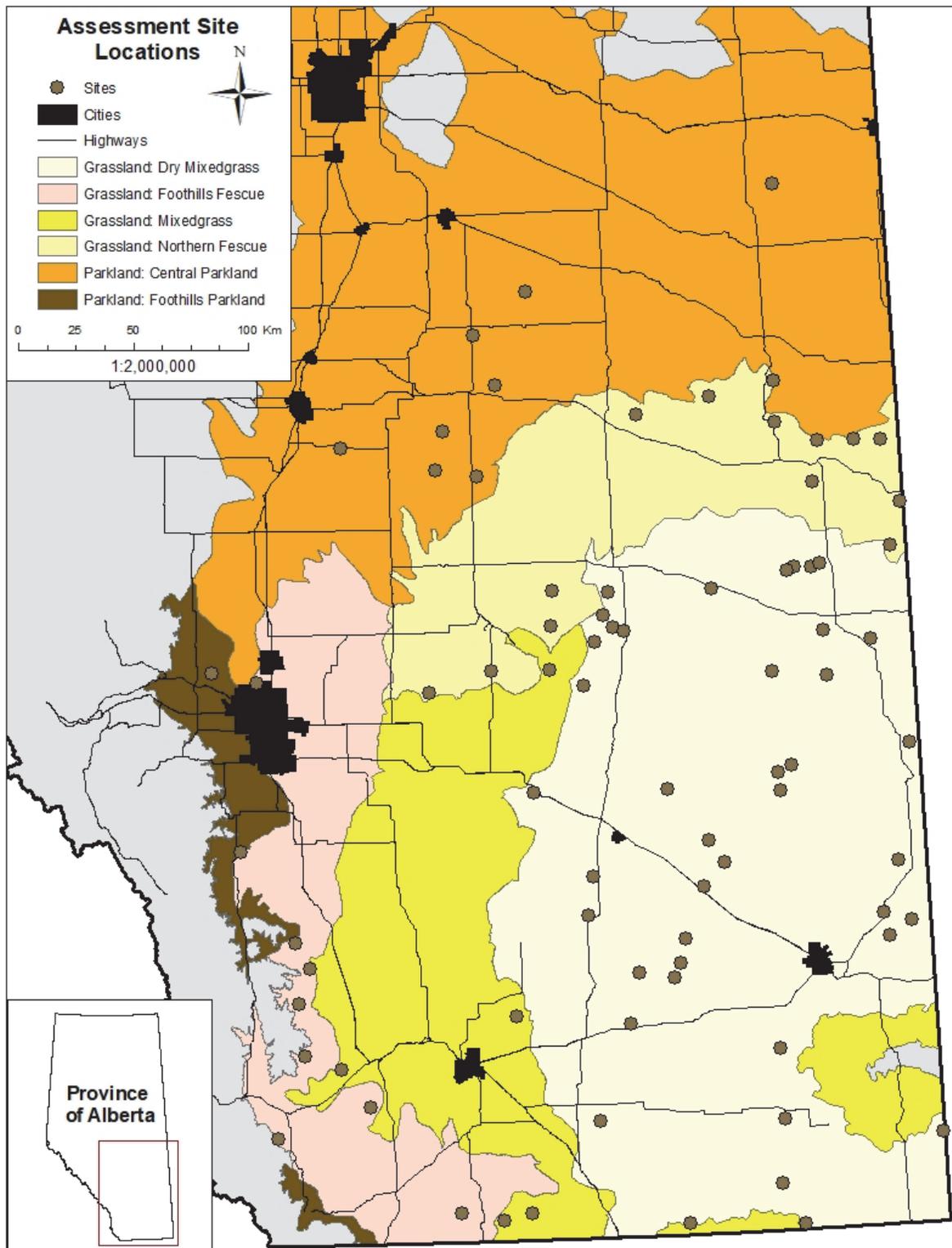
Figure 2.1. Public locations of grassland sites assessed for range health and vegetation diversity in southern Alberta. Natural regions and subregions are based on Alberta's natural subregion classification system (Downing and Pettapiece 2006).
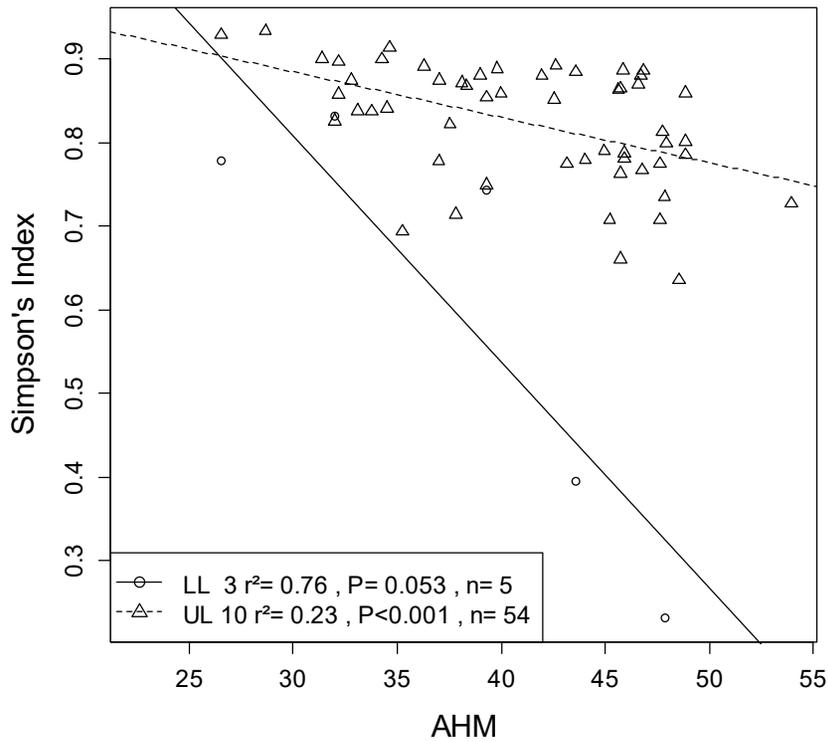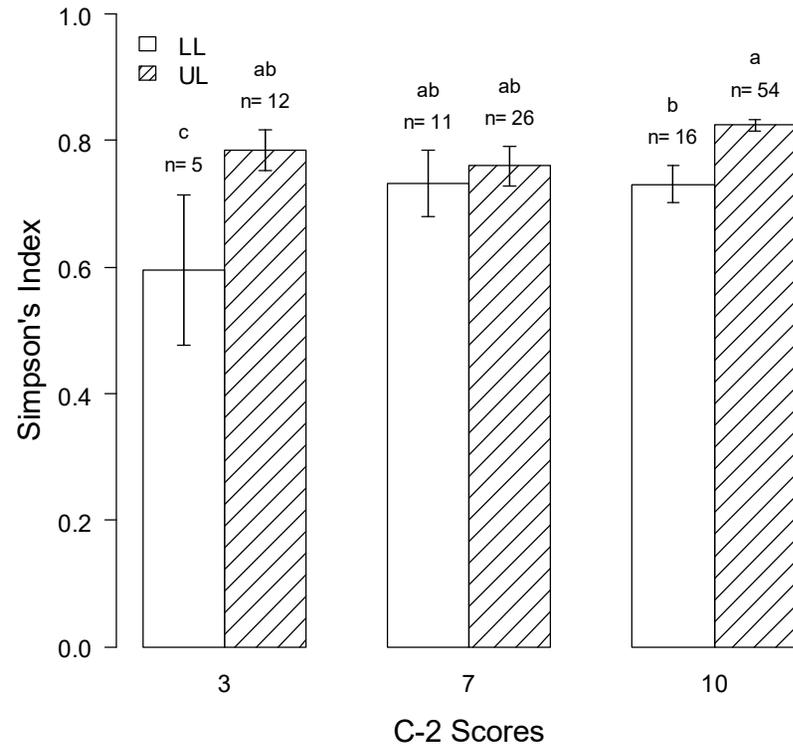
Figure 2.2. Relationship between Simpson's diversity index and (a) annual heat:moisture (AHM) index, presented for select significant (P < 0.10; see Table 2.5) ecosites by C-2 (Canopy structure) component score groupings, and (b) different ecosite groupings, lowlands (LL) and uplands (UL), for each C-2 component score category. Data for the latter are means ± SE, different letters indicate differences in mean values (P < 0.10).

a) AHM - Ecosite groups – C-3   b) AHM – C-3   c) Ecosite groups – C-3
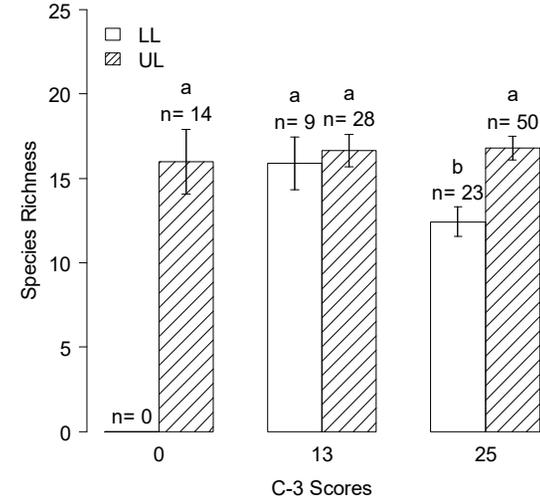
Figure 2.3. Variation in species richness response in locations assessed at one of three different categorical C-3 component scores (Nutrient cycling / hydrological function) along an annual heat:moisture index (AHM) presented (a) in a 3-way interaction with select significant ecosite groups (P < 0.10, Table 2.6) and (b) just by component scores themselves. Figure 2.3c shows means ± SE of different ecosite groupings for each C-3 component score category. Letters indicate differences in mean values (P < 0.10).

a) AHM - Ecosite groups – C-4.1   **b) AHM – C-4.1**   **c) Ecosite groups – C-4.1**
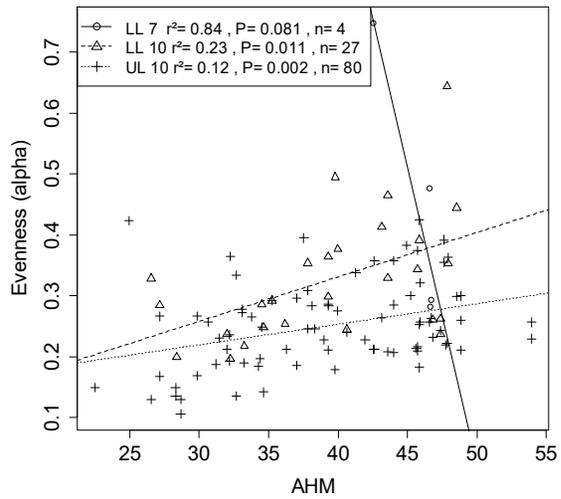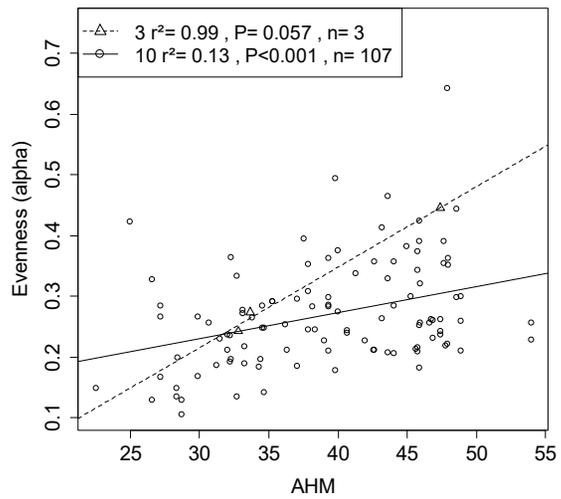
Figure 2.4. Relationship between species evenness and annual heat:moisture (AHM) index, presented (a) for select significant (P < 0.10; see Table 2.7a) ecosite by C-4.1 (Site stability – evidence of erosion) component score groupings, and (b) just C-4.1 significant scores themselves. Figure 2.4c shows means ± SE of different ecosite groupings across each C-4.1 component score category. Letters indicate differences in mean values (P < 0.10). Alpha is an evenness measure representing the slope of a linear curve of the log rank abundance distribution. Lower alpha means more evenness.

Figure 2.5. Relationship between the Simpson's index and annual heat:moisture (AHM) index, presented (a) for select significant (P < 0.10; see Table 2.7b) ecosite by C-4.1 (Site stability – evidence of erosion) component score groupings, and (b) just C-4.1 significant scores themselves. Ecosites are lowlands (LL) and uplands (UL). Figure 2.5c shows means ± SE of different ecosite groupings across each C-4.1 component score category. Letters indicate differences in mean values (P < 0.10).

a) AHM - Ecosite groups
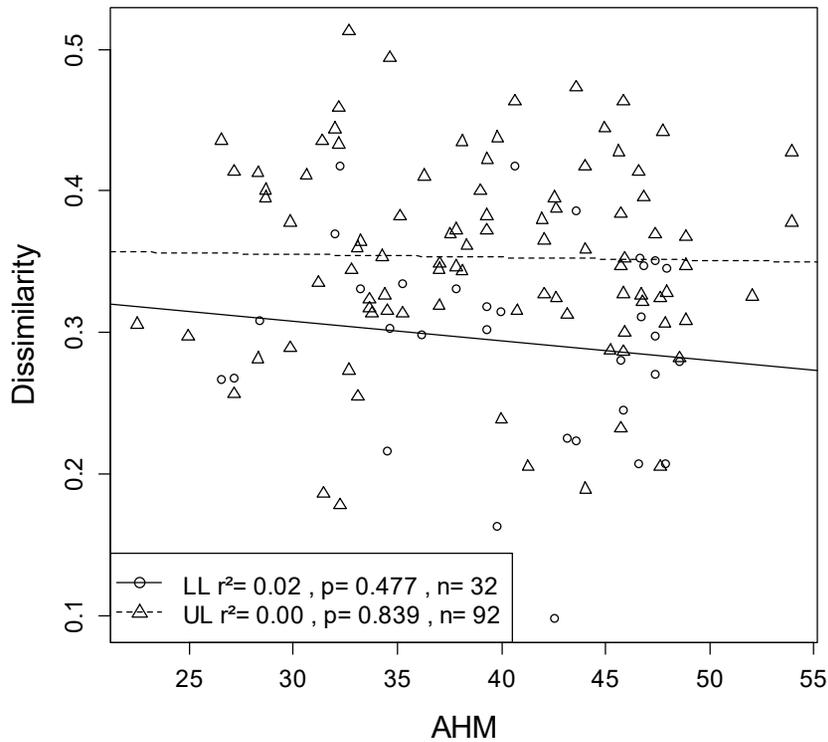
b) Ecosite groups – C-4.1
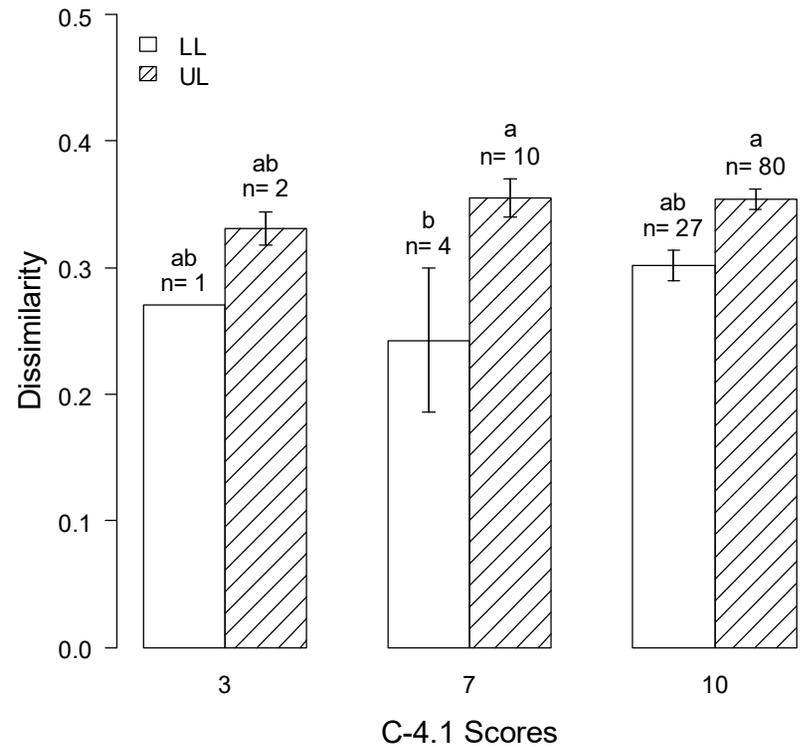
Figure 2.6. Variation in heterogeneity response in locations assessed at one of three different categorical C-4.1 component scores (Site stability – evidence of erosion), either (a) along an annual heat:moisture index (AHM), or (b) as means (± SE) for different combinations of upland (UL) or lowland (LL) ecosite groups by C-4.1 scores. Letters indicate difference in mean value (P < 0.10) for the latter. For Figure 2.6a, no linear models were significant (P ≥ 0.14, Table 2.7c) so only ecosite trends are shown. The dissimilarity index was used to measure heterogeneity.

a) AHM - Ecosite groups – C-5   b) AHM – C-5   c) Ecosite groups – C-5

Legend for (a):
- LL 2 r²= 0.98 , P= 0.097 , n= 3
- UL 6 r²= 0.95 , P= 0.004 , n= 5

Legend for (b):
- 6 r²= 0.83 , P= 0.005 , n= 7
- 10 r²= 0.05 , P= 0.030 , n= 97
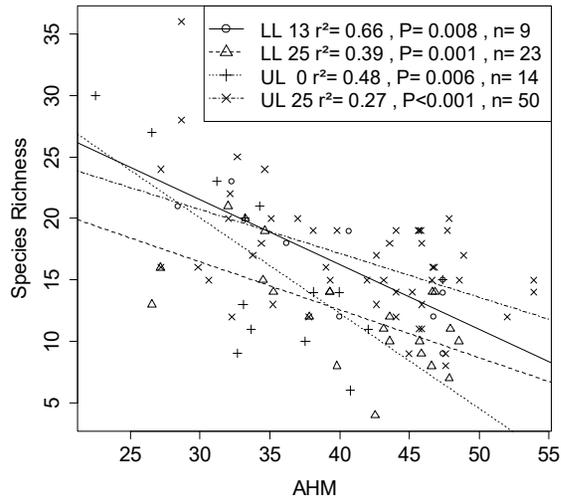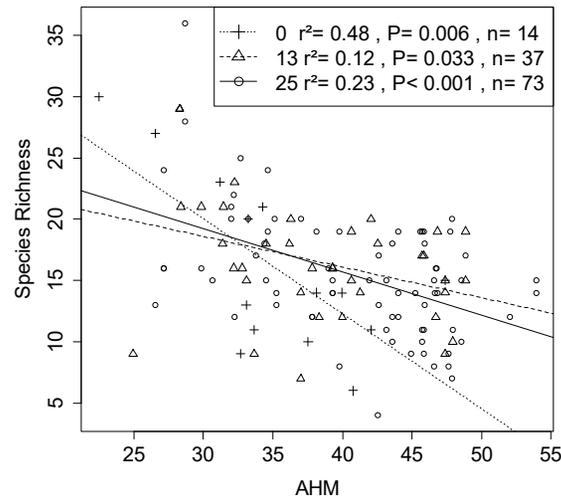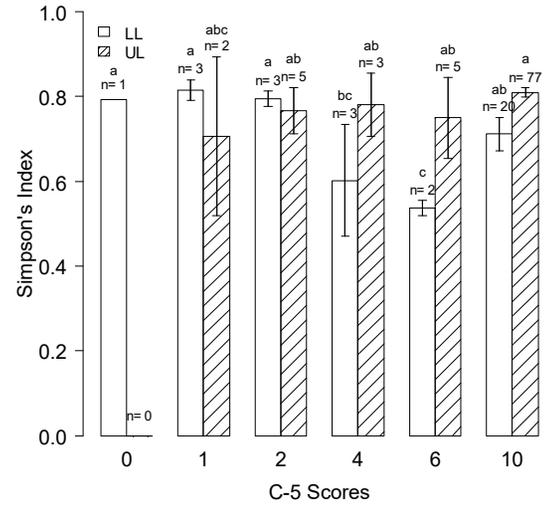
Figure 2.7. Relationship between Simpson's diversity index and annual heat:moisture (AHM) index, presented for (a) select significant (P < 0.10; see Table 2.8a) ecosite by C-5 (Prohibited noxious / noxious weeds) component score groupings, and (b) just C-5 significant scores themselves. Figure 2.7c shows means ± SE of different ecosite groupings across each C-5 component score category with different letters indicating mean differences (P < 0.10). Ecosites are upland (UL) or lowland (LL) sites.
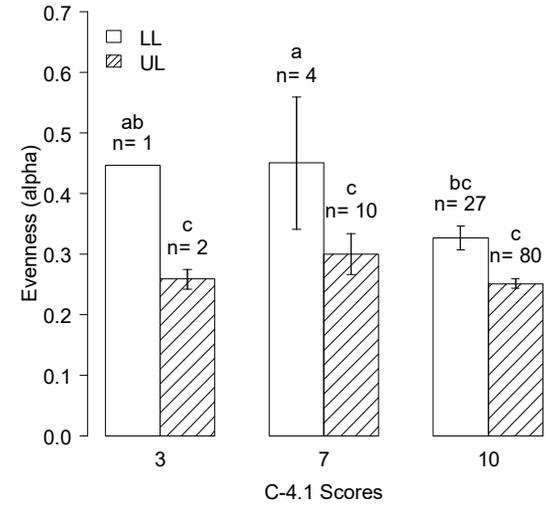
a) AHM - Ecosite groups – C-5
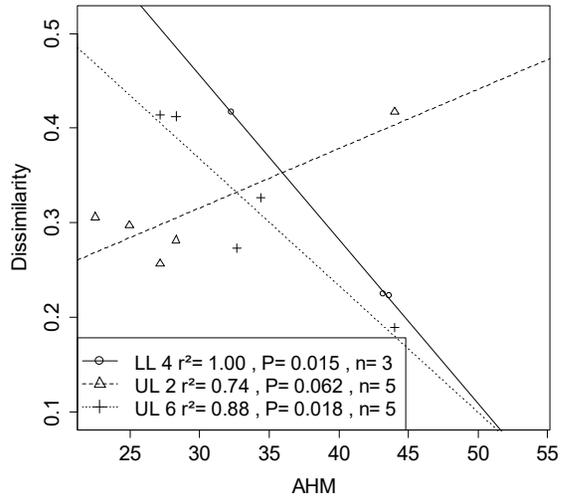
b) AHM – C-5
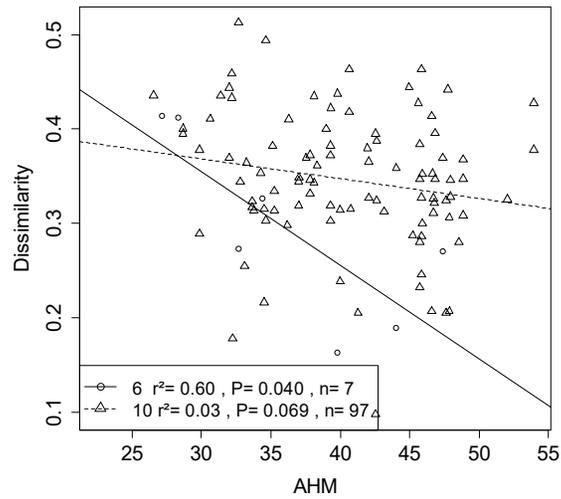
c) Ecosite groups – C-5
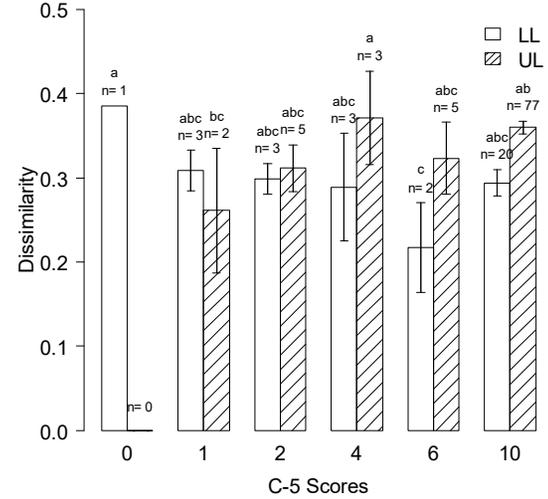
Figure 2.8. Relationship between heterogeneity and annual heat:moisture (AHM) index, presented for (a) select significant (P < 0.10; see Table 2.8b) ecosite by C-5 (Prohibited noxious / noxious weeds) component score groupings, and (b) just C-5 significant scores themselves. Ecosites are upland (UL) or lowland (LL) sites. Figure 2.8c shows means ± SE of different ecosite groupings across each C-5 component score category with different letters indicating mean differences (P < 0.10). The dissimilarity index was used to measure heterogeneity.
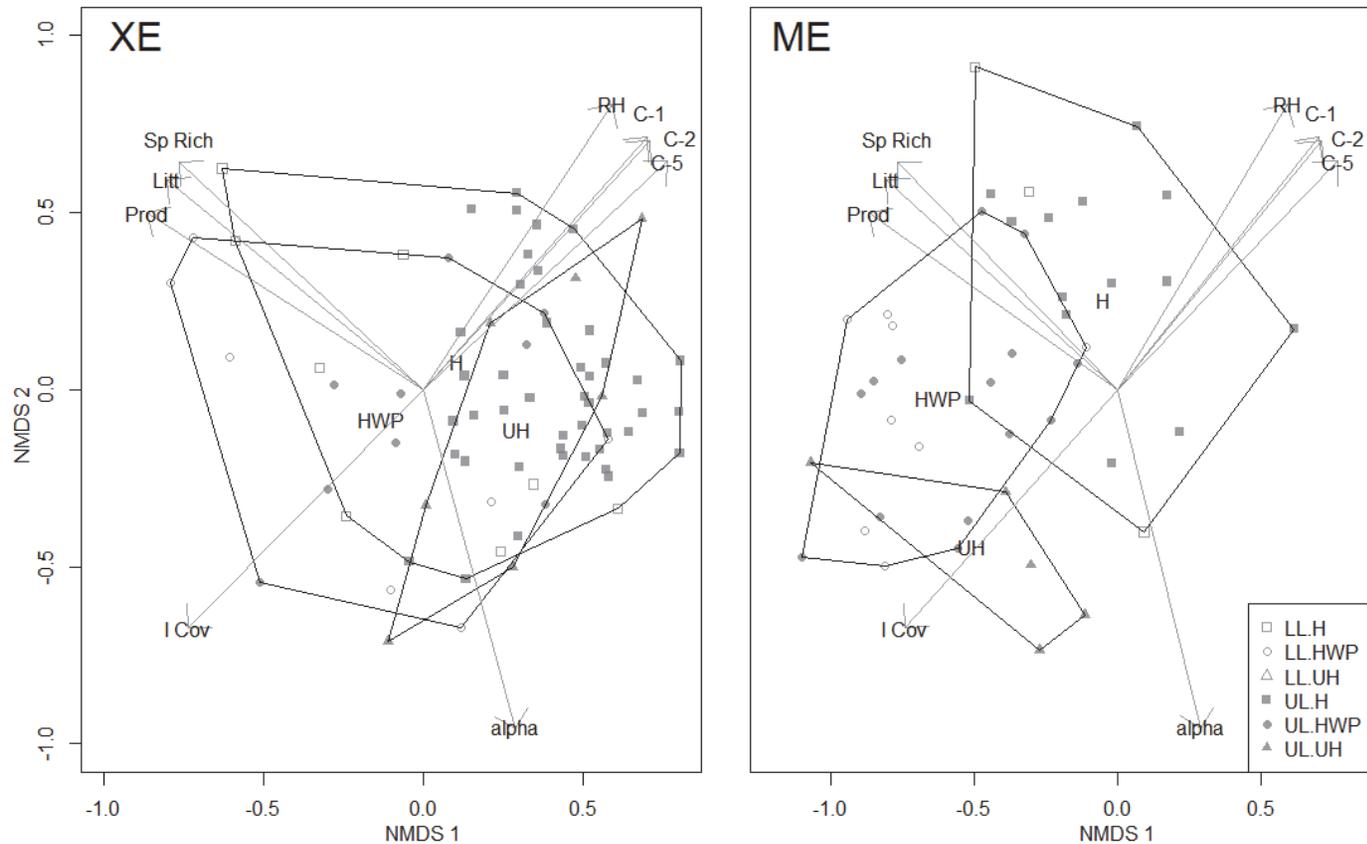
Figure 2.9. Non-metric multi-dimensional scaling ordination of sample site locations, depicting clustering by rangeland health categories. The ordination was split by xeric (XE) and mesic (ME) geographic moisture gradient groups created by amalgamating natural subregions for viewing, which include the mixed grass and dry mixed grass subregions (XE), and the mesic Foothills Fescue, Foothills Parkland, Northern Fescue and Central Parkland subregions (ME), respectively. Rangeland health categories are healthy (H), healthy with problems (HWP), and unhealthy (UH). Changes in point symbology denote range health categories, and upland (UL) and lowlands (LL) ecosite groupings. Total range health (RH), current annual production (Prod), litter biomass (Litt), species richness (SP Rich), evenness (alpha), cover of introduced species (I Cov), and correlated component score vectors (C-1,C-2, C-5) are also shown on the plots (Table 2.10).

a) Range health – Ecosite groups
b) AHM
c) Ecosite groups
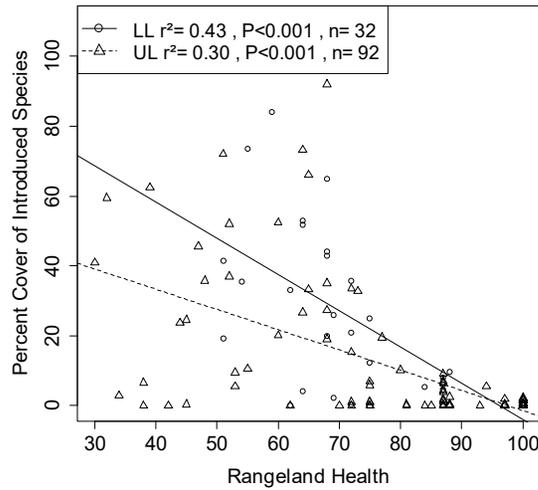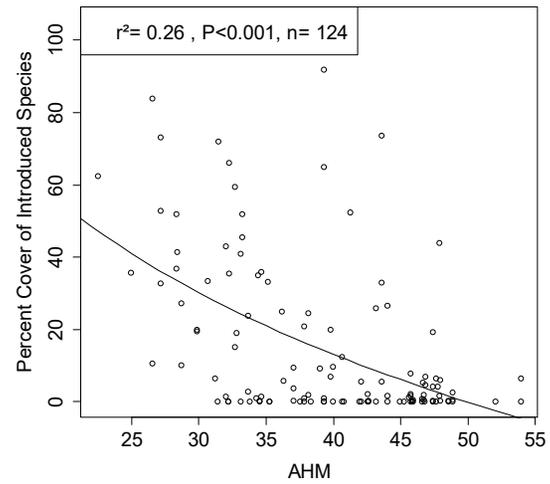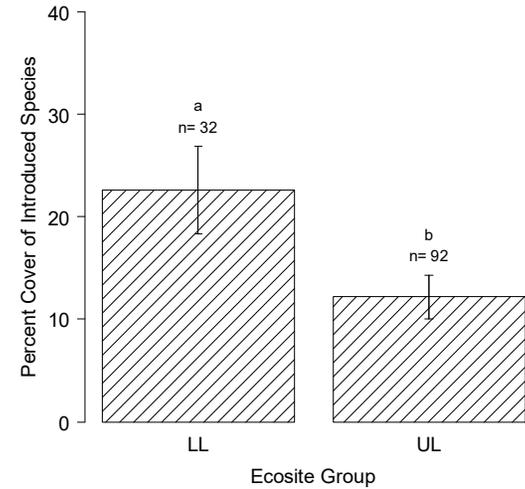
Figure 2.10. Relationship between the total cover of introduced plant species and (a) aggregate range health scores, further separated by ecosite groupings (ESG), (b) annual heat:moisture (AHM) index, and (c) individual ecosite groupings. Ecosite groupings are upland (UL) and lowland (LL), and are expressed as means ± SE with letters indicating mean difference (P < 0.10).
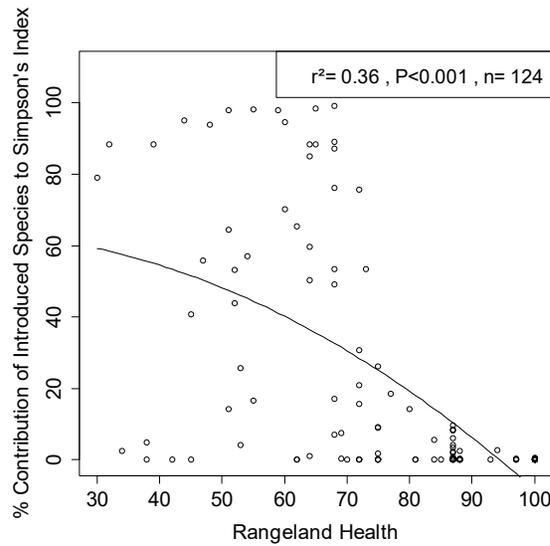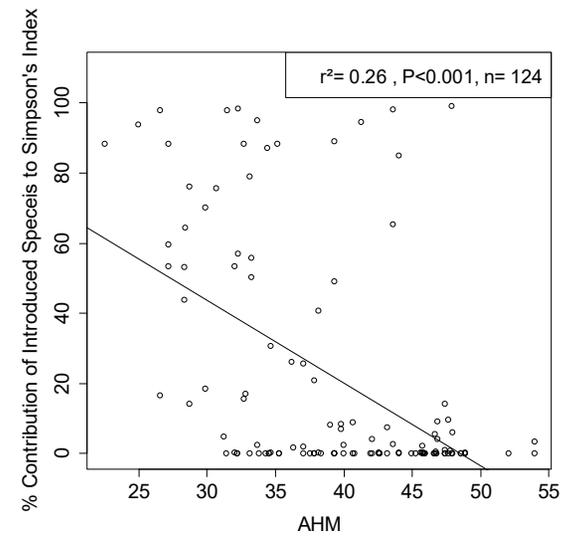
a) Range health   b) AHM   c) Ecosite groups
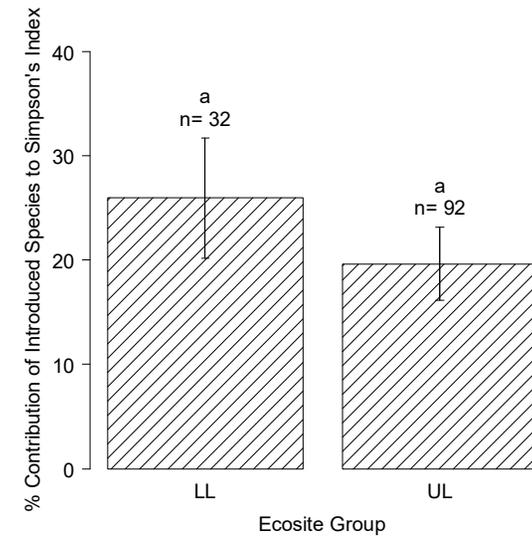
Figure 2.11. Relationship between the relative contribution of introduced plant species on the Simpson's index and (a) aggregate range health scores, (b) the annual heat:moisture (AHM) gradient, and (c) individual ecosite groupings. Ecosite groupings are upland (UL) and lowland (LL), and are expressed as means ± SE with letters indicating mean difference (P < 0.10).

3.0 Rangeland health assessments in pastures and their relationship with plant diversity and community patchiness.

3.1   Introduction

Differing plant communities within pastures containing native grassland are an expression of abiotic factors and ongoing disturbances such as grazing, both of which can affect inherent patterns of diversity (Adler et al. 2000). Pastures are seldom homogeneous, but instead are composed of a number of different plant communities varying discretely in plant species composition and productivity due to factors such as topography, soils and other environmental attributes (Bestelmeyer et al. 2009, CRC 1994, Holecheck et al. 2004). Moreover, varying availability and palatability of forage among plant communities will alter animal preference for individual communities, which then interacts with other factors such as proximity to water, slope and aspect to further influence animal use across different parts of the landscape (Briske et al. 2008, Roath & Krueger 1982, Senft et al. 1987). Ultimately, this creates a mosaic of plant communities within pastures, driven both by the inherent variation in abiotic conditions across the landscape, but also by cattle behavioral preferences in which cattle spatially impose varying levels of use over time. This leads to the creation of "patchiness" of plant communities within a pasture.

Selection of forage by grazing animals at multiple scales, including within individual plant communities and locations across the landscape (Adler et al. 2000, Senft et al. 1987), can increase the heterogeneity of vegetation within a pasture. Grazing directly influences plant

composition due to animals selectively defoliating and re-grazing preferred plants, thereby placing them at a competitive disadvantage (Briske et al. 2008, Collins & Barber 1986). This changes species composition from highly competitive late-seral plants to those species with greater tolerance of disturbance. Uneven utilization (i.e. animal distribution) throughout a pasture can further influence species composition, with substantial areas being repeatedly grazed while other areas remaining non-grazed (Briske et al. 2008). Over time, these composition changes within and across the landscape create community patchiness that have been shown to influence biological diversity (Collins & Barber 1986, Fuhlendorf & Engle 2001, Lwiwski et al. 2015, Sliwinski & Koper 2015).

Quantifying the spatial impact of livestock across heterogeneous pastures is difficult, in part due to the need to identify and define spatially distinct communities, which typically requires intensive sampling (Adler et al. 2000, Sliwinski & Koper 2015, Virk & Mitchel 2014). Often studies on spatial heterogeneity in vegetation relative to grazing are completed through intensive data collection at small spatial scales in single (or poorly replicated) pastures, and rarely are conducted across extensive land areas that include a large number of pastures distributed across multiple management units (i.e., Fynn & O'Connor 2000, Fuhlendorf & Smeins 1999, Lwiwski et al. 2015, Sliwinski & Koper 2015). The ability to measure grazing intensity at each location is a particular challenge.

A more rapid solution to indirectly estimate grazing intensity may be to utilize rangeland health assessments, and more specifically, quantify how plant communities within pastures have been influenced by grazing. Range health evaluations are used by land managers to measure the overall functional responses of plant communities arising from all past

disturbances. These include assessments of multiple indicators of plant community change, including plant community composition and structure, site stability, litter abundance and weeds (Adams et al. 2016, Pellant et al. 2005, TGUCT 1995). Health assessments are completed at the plant community level, and routinely averaged to obtain an overall metric of health for the entire pasture; however, spatial variation in health (i.e., patchiness) could also be used to provide an indication of how grazing is impacting vegetation heterogeneity across pastures.

Community types are abstract amalgamations of existing plant communities that actually occur on individual geographic units across rangeland landscapes. For the purposes of this study, a plant community type was defined as a relatively discrete (i.e., identifiable) state of vegetation, as represented by a range of specific vascular plant species composition and abundance. Although there could be subtle differences in the abundance of minor plant species, it was expected that community types (defined by dominant vegetation) would reoccur on geographic units with similar abiotic attributes and disturbance (Bestelmeyer et al. 2003, Laycock 1991, Westoby et al. 1985). These abiotic attributes are summed to define an ecological site (hereafter ecosite), which is a discrete physical area of the landscape containing unique physical properties (soil texture, salinity, landscape characteristics, and both microclimatic and macroclimatic features) capable of supporting a specific kind and productivity of plant community (Bestelmeyer et al. 2003, TGUCT 1995). In the absence of disturbance, or with a stable amount of disturbance for an extended period, individual ecosites can be expected to support a single uniform plant community in a relatively stable state, and which is repeated in separate geographic areas with the same ecosite properties (Bestelmeyer et al. 2003, Fuhlendorf et al. 2012, Pellant et al. 2005, Westoby et al. 1989).

Rangeland health assessments essentially measure the deviance in each existing plant community from the expected community type due to factors such as excessive grazing, with an emphasis on conserving plant composition and structure, as well as underlying hydrological function and site stability (Adams et al. 2016, Pellant et al. 2005, TGUT 2005). Above natural levels of grazing, a plant community is expected to deviate from the reference community towards early seral, disturbance tolerant plant species within a given climatic zone (Ellison 1960, Milchunas et al. 1988). In cases with even heavier grazing, soil erosion may occur and hydrological functions may be impaired (Ellison 1960, TGUT 2005). This retrogression, characterized by increasing vegetation and structural deviance from the expected community type, is then associated with lower range health scores (Adams et al. 2016, TGUT 2005). It should be noted that this method does not differentiate among different types of disturbances, as factors such as fire and prolonged drought may also alter range health. As a result, range health assesses the total aggregate effects of all disturbances on the plant community within a given ecosite.

This study examined the relationship between plant community patchiness (i.e., within pastures) and associated rangeland health using a large existing dataset previously created through an established inventory methodology. The goal was to test whether observed measures of rangeland health predicted the spatial patchiness in plant communities found across pastures spanning diverse landscapes. Our assumption was that along with the spatial heterogeneity in vegetation created by different ecosites, patchiness of plant community types would further increase within unevenly grazed pastures. Unevenly grazed pastures in turn, would promote a greater differentiation (co-occurrence) of plant community types on the same

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ecosite, with some more similar to the reference community and others more distinct in their successional status due to grazing. Thus, more unevenly grazed pastures would be represented by lower health scores and a larger range of scores therein. The overall hypothesis tested was that pastures with intermediate average health scores would have more patches of plant community types. This is relative to pastures that remained completely healthy, or those heavily grazed such that all ecosites and communities therein were consistently unhealthy; in both cases each ecosite within a pasture would have one consistent community type (healthy or unhealthy) associated with it. An illustration of this concept is provided in Table 3.1. In essence, greater differences in rangeland health within ecosites are hypothesized to be associated with more overall heterogeneity. In order to test this with the information on hand the hypothesis was divided into the following components:

- Different ecosites and community types are vegetationally unique from one another, such that if they occur in the same pasture, plant diversity increases.

- The ratio of plant communities to ecosites within individual pastures is influenced by rangeland health.

- Some ecosites will be more affected than others by grazing within a pasture, and therefore will be more susceptible to forming patches of differing rangeland health.

- Vegetation and plant diversity within individual ecosite patches will be influenced by regional climatic attributes, as measured by variation in aridity.

3.2 Methods

3.2.1 Study Area and MULTISAR

Rangeland vegetation and health assessments were conducted across grasslands within ranches of various sizes located in the Dry Mixed Grass, Mixed Grass and Foothills Fescue natural subregions of southern Alberta (Figure 3.1) (Downing & Pettapiece 2006). All pastures were assessed with surveys completed through Habitat Conservation Strategies performed by the MULTISAR (Multiple Species At Risk) partnership. MULTISAR is a multi-agency, collaborative program that focuses on multiple species conservation at the landscape level through habitat determination and the incorporation of best management plans for species at risk (Downey et al. 2005). Southeastern Alberta has a high number of species at risk and many of their habitats overlap. The MULTISAR program was created as a holistic strategy where both habitat analysis (provided by rangeland assessments) and current wildlife species occurrences are incorporated into a Habitat Conservation Strategy (Rumbolt et al. 2011).

In total 83,933 ha were evaluated in this study encompassing 16 ranches. A total of 262 pastures were assessed that ranged in size from 2.6 to 2364 ha in size, with a mean of 320 ha (SE ±24.4) and median of 181 ha. Pastures were not evenly distributed around the mean however, with 31 % below 100 ha and 55 % below 200 ha in size (Figure 3.2). MULTISAR's objectives in conducting rangeland assessments were to classify plant communities, assess rangeland health, and attain carrying capacities for large herbivores for each pasture on a property to aid in making informed habitat management decisions (Rumbolt et al. 2011).

Pastures spanned a range of climatic conditions; 30 year (1984 -2013) mean annual temperatures ranged from 3.2 to 6.2 °C, with a mean of 5.4 °C (±0.02) and median of 4.1 °C. Mean annual precipitation (MAP) ranged from 290.6 mm to 544.7 mm, with a mean of 386.4 mm (±3.0) and median of 370.5 mm (Mbogga et al. 2010). As available moisture is known to be the primary driver of productivity in grasslands, especially when MAP is below 500 mm (Sims & Singh 1978), we used the annual heat:moisture index (AHM), an index of aridity, to represent the range of climatic conditions at these sites. Average 30 year (1984 -2013) AHM values were calculated from Climate AB ver. 3.21 data, which interpolates long-term climate measures from weather stations for individual locations (http://tinyurl.com/ClimateAB) (Mbogga et al. 2010). AHM is calculated from mean annual temperature (MAT) and precipitation (MAP) in the following formula:

$$(MAT + 10) / (MAP / 1000)$$

AHM therefore accounts for both moisture and temperature differences as the geographic location changes (Mbogga et al. 2010). A higher number indicates a warm, drier climate, whereas lower values indicate a cooler and wetter climate. Values of AHM ranged from 28.5 to 56.1 across assessed pastures, and had a mean of 44.0 (± 0.4) and median of 45.1. For some analysis where interactions occurred, AHM was split based on the subregions they represented (Downing & Pettapiece 2006), namely Foothills Fescue (FF, AHM = 28 – 33), Mixed Grass (MG, AHM = 35 – 44), and Dry Mixed Grass subregions (DMG, AHM = 44 – 56).

3.2.2 MULTISAR Pasture Assessment Methodology

Pasture assessments were conducted by first stratifying the pasture into polygons remotely, ground truthing the polygons, and then completing detailed assessments of those grasslands designated native. Initially, a desktop mapping exercise was conducted to stratify each pasture into polygons by tone, texture, spectral and topography differences as evident on 1 m resolution or finer orthographic or infrared imagery. One property was done manually, the remainder by utilizing polygons created previously by the Alberta Grassland Vegetation Inventory (GVI) Program (GOA 2011). The property completed manually was done to the specifications listed by the GVI Program. Resulting polygons were field verified by rangeland agrologists, and confirmed the attribution of basic site-types, including native grasslands, tame pasture, annual crops, open water, lentic wetlands, lotic riparian, and rural and industrial developments (GOA 2011).

Polygons field verified as native grasslands were further assessed by collecting field data to identify the ecosite and plant community type, as well as quantify rangeland health. In total, 84% of the study area was assessed with the protocol for native grassland (Adams et al. 2016, Rumbolt et al. 2011). Data collection for these polygons entailed either directly measuring them, or linking to a polygon already measured previously if deemed to be on the same ecosite and dominated by the same vegetation. Directly measured data entailed installing a 50 m long line transect for vegetation sampling within a uniform area of vegetation across a constant landscape position (i.e. at the same slope, relief and aspect). Foliar plant cover of grasses, forbs and ground components (litter, soil, rock and bryophytes) was then assessed in ten micro-plots (20 cm x 50 cm) located systematically at 3 m intervals along the line. Shrub cover was

measured at each sampling location within a 1 m² quadrat nested overtop each micro-plot. Physical site information, such as slope, aspect, soil classification, natural subregion, ecosite, and (more broadly) the nearby presence of shrubs and trees, was also recorded.

Each grassland polygon was first confirmed to an ecosite type using the physical site information and plant composition field data collected, as well as spatial information provided by the GVI and Alberta Agriculture Soil Inventory Database (AGRASID 3.0) (Brierley et al. 2001, GOA 2011). These were confirmed using Alberta's methodology for range plant community type classification (Adams et al. 2005, 2013a, 2013b). In Alberta, classification of grasslands includes a 'range-site' determination that once located geographically within a natural subregion, can be analogous to ecosite (Adams et al. 2013a, 2013b, 2003, TGUCT 1995); therefore the term ecosite was used for this study. A total of 14 ecosites were utilized in the surveys included in this study (Appendix B).

Using the ecosite and observed vegetation data, each assessment was classified to a plant community type using the same range plant community type classification methodology with the dominant vegetation of the assessment (Adams et al. 2005, 2013a, 2013b). This classification involved a combination of Cluster and De-trended Correspondence (DCA) analyses to group plant communities by their dominant species. Ward's method of cluster analysis was first completed to create clusters of various heights, and then a DCA was used to compare the uniformity of top species from eigenvalues (R Core Team 2018, Oksanen et al. 2017). Clusters with a low eigenvalue (i.e., <0.25) were deemed similar enough to be a single (i.e., consolidated) community type, while clusters with larger values were further split by a lower cluster height until smaller groups could be made. Clusters were also viewed with non-metric

multi-dimensional scaling (NMDS) for consistency in groupings (Oksanen et al. 2017). In total

327 plant community types were identified across all ecosites (Table 3.2). Of these, 141 of the

community types accounted for 85 % of the assessments, with the remaining 15 % of

assessments identified as singular community types.

Community types were nested within their respective ecosites and used to assess each

community types rangeland health. The rangeland health assessment entailed using the field

verified ecosite determination to find a description of the reference (i.e. expected) plant

community type from the relevant plant community classification guide (i.e., Adams et al. 2005,

2013a, 2013b). The field measured community type within each sampling polygon and the

associated reference community were then compared using the rangeland health criteria to

obtain a score (Adams et al. 2016).

Rangeland health criteria are intended to provide information on how well the existing

community is performing ecological processes compared to the reference community for that

ecosite. The criteria are divided into five unequally weighted questions with categorical scoring

in each (Adams et al. 2016). These categories are 1) ecological integrity (based on plant

composition), 2) vegetation structure, 3) hydrological function / nutrient cycling, 4) site

stability, and 5) prohibited noxious and noxious weed presence. Discontinuous areas (i.e.,

polygons) that were visually assessed to be a previous plant community type and ecosite were

either assigned the same range health score as others (15 % of polygons assessed), or could

have an independent assessment to describe subtle differences in the health score that

occurred without changing the community type (18 % of polygons assessed). The number of

rangeland health assessments completed ranged from 1 to 37 per pasture, depending on its

complexity (mean = 6.2 ±0.4, median = 4); each assessment was linked to a polygon that

represented a specific proportion of the area within a given pasture.

3.2.3   Data Analysis

3.2.3.1 Ecosite and Plant Community Type Composition

Ecosites were tested for intrinsic vegetational differences using the detailed plant

species composition derived from each assessment. In total 1245 vegetation assessments were

included in the analyses (Table 3.2). Compositional differences within ecosites were tested

using permutational multivariate analysis of variance (PERMANOVA) of Bray-Curtis distance

matrices (Anderson 2001), as well as with pair-wise permutation multivariate analysis of

variance, the latter identifying specific differences among ecosites (Hervé 2017). Non-metric

multi-dimensional scaling (NMDS) was used to visualize these differences (Oksanen et al. 2017).

3.2.3.2 Plant Diversity

Plant diversity metrics of species richness and evenness were computed for each

vegetation assessment (transect, and therefore polygon) completed in the study. Richness was

the sum of species per assessment, and ranged from 3 to 42, with a mean of 16.5 (±0.14) and

median of 16.   Evenness was calculated using the relative abundance distribution of each plant

species that occurred within an assessment (Magurran 2004), calculated by rank abundance

(Oksanen et al. 2017). The pre-emption model was used to fit a linear line to the log of the rank

abundance distribution (Oksanen et al. 2017, Wilson 1991). The slope of the line (called the

alpha value) described the evenness of the plant community. Comparatively high alpha values

indicate a higher rate of decay, a greater dominance of only a few plant species, and a rapid

reduction in species abundance, whereas lower alpha values indicate less decay and greater evenness. Evenness alpha values ranged from 0.09 to 0.83, with a mean and median of 0.26 (±0.002) and 0.24, respectively.

To establish relationships between diversity metrics among different ecosites and AHM, species richness and evenness were assessed using a linear mixed model and analysis of variance tables in R software (Bolker et al. 2008, Fox & Weisberg 2011). Individual pastures, grouped by ranch properties, were included as a random factor to account for the relatedness (i.e., blocked nature) of assessments that had the same landscape conditions and management influence. Relationships with richness and evenness were then tested with range health and AHM using the same mixed model properties within individual ecosites that had sample sizes greater than 30. These were Blowout, Clayey, Loamy, Overflow, Sand, Saline Lowland, Sandy and Thin Break ecosites (Table 3.2).

3.2.3.3 Patchiness within Pastures

To test whether rangeland health was associated with the heterogeneity of plant communities within ecosites, the patchiness of plant communities was calculated as the ratio of the number of plant community types to the number of ecosites (PCT:ES) within a pasture. One large pasture twice the size of all others was removed in the analysis (see Figure 3.2), leaving the upland native grassland area assessed across all pastures at 70,569 ha, ranging from 2.6 ha to 2208 ha of grassland per pasture.

To determine the health score for each pasture, both a simple mean health score was computed (average RH of all polygons regardless of size) and a score weighted by the size of

each polygon it represented. For the latter, the relative area of each vegetation polygon was used to adjust the contributing influence of range health for each plant community type to the overall pasture mean. Weighted range health averages were used for analysis as pastures were typically measured with unequal areas representing rangeland health values. Differences between weighted and average range health was particularly notable as pastures increased in size (Appendix C). Rangeland health differences (max – min) were also computed (represented by delta pasture health) as the difference between the maximum and minimum health scores among communities within a pasture.

PCT:ES, pasture rangeland health, and AHM were tested for relationships using a mixed model, with individual ranch properties as the random effect to account for pastures more closely associated geographically and with localized management (Bolker et al. 2008). AHM and pasture health demonstrated weak but significant collinearity (P < 0.001, r^2 = 0.04), and were therefore considered redundant in the model. As PCT:ES and AHM were not associated with one another (F = 0.28, Df = 1, Pr(>F) = 0.60), we removed AHM and tested rangeland health alone for its relationship with PCT:ES.

Lastly, we assessed individual ecosites that were represented by 30 or more pastures for total plant community types responses to rangeland health within each pasture. These included the Blowout, Loamy, Overflow, Saline Lowland and Thin Break ecosites. These were assessed by first identifying the number of plant community types (PCT) within an included ecosite per pasture. Pastures had from 1 to 5 or 6 Plant community types for each of these ecosites. The weighted range health and delta health (max-min) was calculated for these ecosites per pasture as well. Thereafter a generalized linear mixed model was used (due to

count data) to test the relationship by relating the number of community types per ecosite to the rangeland health metrics calculated per pasture with property as the random effect (Bolker et al. 2008). Rescaling of weighted range health was required to better balance the analysis. We then related pasture level (but within ecosite) species richness and evenness diversity metrics to PCT numbers, along with AHM, using a linear mixed model to test whether more plant community types in a pasture increased overall species richness and evenness (within a given ecosite), or if it was more effected by the moisture gradient.

3.3   Results

Grassland areas assessed within individual ranch properties ranged from 268 to 14,078 ha, with more ecosites occurring on larger properties. AHM values among the pastures within the 16 ranches ranged from 29 to 56, and rangeland health of assessed pastures averaged 75 % (±0.8), with the lowest average pasture health score on a ranch being 46 % (±0.4.6), and the highest being 81 % (±0.1.6).

Pastures contained from 1 to 9 ecosites (mean = 2.5 ±0.09, median = 2); the most common ecosite being Loamy, which covered 52 % of the native grassland area sampled (Table 3.2). Other common ecological sites were Overflow, Blowout and Thin Break. All remaining ecosites each comprised less than 5% of the area assessed. The number of plant community types identified per pasture ranged from 1 to 21 (mean=4.1 ±0.21; median=3), depending on the size and complexity of the pasture. Less common ecosites typically had a larger number of community types relative to their representation in the total sampling effort, indicating strong

uniqueness among the plant communities observed for those ecosites. For example, the

Badland ecosite had 17 different plant community types for 18 transects, whereas the Loamy

ecosite had 55 plant community types on 503 transects.

3.3.1 Ecosite and Plant Community Comparisons

Most ecosites were different in vegetation composition from one another (PERMANOVA

F = 13.8, P = 0.001), although pairwise comparisons showed there were some that did not differ

(P > 0.05). Vegetation composition within the Limy ecosite were similar to that in the Blowout,

Clayey, Loamy, Sand, Sandy, Overflow, Thin Break, Gravel and Shallow to Gravel ecosites

(Appendix D). Additionally, the Shallow to Gravel ecosite was similar in vegetation to Gravel,

Sand and Loamy ecosites. Limy and Shallow to Gravel ecosites had the fewest assessments

(Table 3.2). Finally, Choppy Sandhill and Sand ecosites were also similar in composition.

Visually the plant community composition of most ecosites had substantial overlap in the

supporting ordination (Figure 3.3).

Plant species richness measures generally differed among ecosites, although ecosite

identity also interacted with AHM to alter richness (F = 2.14, df = 13, P = 0.01). Further

inspection showed the Clayey, Loamy, Gravel, Overflow and Thin Break ecosites all exhibited a

reduction in richness as AHM (i.e., aridity) increased (Table 3.3a, Figure 3.4a). Results also

tended to showed that the Thin Break and Loamy ecosites generally had greater richness

throughout, while the Overflow and Gravel ecosites had lower richness. Species evenness

showed the same interaction between ecosites and AHM (F = 3.38, df = 13, P < 0.001), where

evenness decreased (alpha increased) as aridity increased within each of the Clayey, Loamy,

Overflow, and Thin Break ecosites (Table 3.3b, Figure 3.4b). As AHM increased, the Overflow

ecosite became the least even whereas the Thin Break ecosite remained the most even in

composition among these four ecosites. Both richness and evenness within the Clayey ecosite

remained erratic due to poor distribution of the data, while all other ecosites had no significant

relationship ($P \leq 0.05$, Table 3.3).

Permutational multivariate analysis of variance (PERMANOVA) indicated that the plant

community types documented differed in composition from one another within each of the 14

ecosites examined ($P \leq 0.05$) (Table 3.4). Further testing within each ecosite (with sample size

greater than 30) for diversity relationships with rangeland health and AHM indicated that

health and AHM influenced plant species richness and evenness within the Loamy, Overflow,

and Saline Lowland ecosites ($P \leq 0.05$, Table 3.5). An interaction of AHM and health was

evident on species evenness within the Loamy ecosite. Closer inspection of this interaction

indicated that evenness decreased (alpha increased) as range health increased, but only in

mesic areas (i.e., Foothills Fescue grasslands). However, the reverse occurred in more arid (i.e.,

Dry Mixedgrass) areas, with evenness increasing with greater range health (Figure 3.5a). AHM

alone was related to evenness in the Overflow ecosite, where the previous ecosite analysis

indicated that evenness decreased as AHM increased (Figure 3.4b). Species richness was

related to AHM in each of the Loamy, Overflow, and Saline Lowland ecosites; species richness

declined as AHM increased, but Loamy ecosites maintained higher richness throughout (Figure

3.4a), and no discernible trend noted for Saline Lowland ($r^2 = 0.03$, $P = 0.23$; data not shown).

This discrepancy between the mixed model and regression for the Saline Lowland ecosite may

be explained by the near interaction of range health and AHM ($P = 0.059$, Table 3.5). No plots

occurred in the mesic conditions of the Foothills Fescue subregion, and only in the intermediate

moisture conditions of the Mixed Grass did a strong trend exist for richness to decline with

increasing range health (Figure 3.5b).

3.3.2   Pasture Level Relationships

At the pasture level, plant community type to ecosite (PCT:ES) ratios (the sum of plant

community types to the sum of ecosites per pasture) ranged from 1 to 4.3 (mean = 1.5 ±0.04,

median = 1.3) across all 262 pastures, although 86 % of pastures had ratios at or below 2.0.

Mixed model analysis indicated there was a relationship between weighted range health scores

and plant community type patchiness at the pasture scale (F = 6.14, df = 1, P = 0.014).

Patchiness, as represented by the PCT:ES ratio, was greatest where weighted range health

scores were between 70 and 90 %, and declined as scores either neared 100 % or fell below 60

% (Figure 3.6a).   An even stronger relationship occurred between PCT:ES ratios per pasture and

differences in rangeland health scores computed using the maximum difference (i.e., delta) in

observed health scores within each pasture (F = 86.3, df = 1, P < 0.001). Here, the PCT:ES ratio

increased as the delta in these pastures increased (Figure 3.6b).

Common ecosites (those ecosites found in over 30 pastures) were isolated and assessed

independently for relationships between the number of plant community types (PCT) found

and rangeland health. There was no relationship between the number of plant community

types within these ecosites and pasture weighted range health scores (Table 3.6a, P > 0.17).

However, the opposite was true when the number of community types were related to delta

pasture health scores, which were highly associated for all ecosites (Table 3.6b, P < 0.001). For

each of these five ecosites, the number of plant community types increased as the range of scores (delta values) increased within a pasture (Figure 3.7). Moreover, this trend was strongest within the Overflow ecosite ($r^2 = 0.76$) and least apparent within the Blowout ($r^2 = 0.36$). The Saline Lowland ecosite had a high trend ($r^2 = 0.68$), but notably many of the pastures with this ecosite contained only 1 plant community type (n = 41).

Plant diversity metrics at the pasture level were related to the number of community types (PCT) but varied in response among each of the five most common ecosites, with additional influences of climatic conditions (Table 3.7). Species richness and evenness both responded to the interaction of PCT and AHM for the Blowout ecosite. Inspection of this interaction revealed that more arid pastures (i.e., those in the DMG) increased in species richness and evenness as the number of PCT's increased (Figure 3.8). A similar interaction occurred within the Loamy ecosite for species richness, although this increase appeared to be reduced in areas with greater aridity (Figure 3.9a). Evenness did not relate to PCT within the Loamy ecosite; however, evenness generally decreased (alpha increased) as pastures became more arid (Figure 3.9b). Species richness and evenness did not relate to PCT's in the Overflow ecosites, but did to AHM, however only indicated a weak trend (P = 0.05 - 0.10) wherein species richness and evenness decreased as AHM increased (Figure 3.10). Species richness varied with the number of PCT's in the Saline Lowland ecosite, but only in more arid pastures. Similarly, evenness responded to PCT within the Thin Break ecosite, though only in the MG and DMG regions. Inspection showed responses similar to the Loamy and Blowout ecosites, where richness in the Saline Lowland and evenness in the Thin Break ecosite increased as the number

of PCT's increased per pasture (Figure 3.11). There was no relationship between evenness in Saline Lowland or richness in Thin Break ecosites and the number of PCT's (Table 3.7).

3.4   *Discussion*

Current rangeland inventory methodology is designed to survey across a landscape to aid management decisions (Bestelmeyer et al. 2009, Holechek et al. 1989, Rumbolt et al. 2011). Specifically, the MULTISAR data utilized in this study was gathered to provide knowledge of the type of plant communities each ranch was comprised of, their productivity, and health so that management could sustain species at risk habitat (Rumbolt et al. 2011). To date, MULTISAR's habitat conservation strategy has covered 159,970 hectares of land, of which nearly 84,000 assessed hectares were used within this study (MULTISAR 2018). Typically, range inventory data is more generalized than specific research information, but may nevertheless be useful in understanding vegetation heterogeneity, defined as plant community patchiness at a coarse scale. In order to test this, we first quantified differences among the ecosites identified, including their plant community types as classified through rangeland inventory methodology. Next, we combined information on their identity to create a measure of patchiness within a pasture and tested whether this metric of plant community patchiness varied with range.

3.4.1   Ecosite and Plant Community Type Vegetation and Diversity Differences

Most ecosites identified and embedded plant community types had differing vegetation composition. Ecosites are defined by differences in environmental attributes and can be visually separated in the field or remotely by topographic patterns, landform, and soil

conditions (AEP 2018, Bestelmeyer et al. 2003, GOA 2011, Maynard et al. 2007); therefore, abiotic differences should be expected. Some ecosites not dissimilar could be explained by inadequate sampling (low sample size due to rare conditions) or the presence of relatively similar environmental (abiotic) factors. For example, Gravel and Shallow to Gravel ecosites, as well as Choppy Sandhill and Sand ecosites, would have similar textured soil to one another, in turn promoting similar vegetation (Adams et. al 2013a, Adams et al. 2013b).

Ecosites and community types also fundamentally differed in plant species diversity and evenness. Interestingly, in our study the ecosite characterized primarily by a steep slope had the highest species richness and evenness (i.e. Thin Break), while ecosites determined solely by high moisture accumulation in the landscape (i.e., Overflow) were lower in plant diversity. Similar findings were noted in the previous study (Chapter 2) where lowlands with greater moisture generally had lower diversity, and are consistent with results in other studies where differing diversity measures occur with topographical differences (Dorji et al. 2014, Hartnet et al. 1996, Lwiwski et al. 2015, Moeslund et al. 2013). Notably, this response is separate and unique from that associated with plant species changes due to regional climatic shifts, where more diversity was generally found in mesic than drier areas within the same ecosite. Ecosites in which diversity was related to AHM in our study generally exhibited increasing diversity from arid to more mesic conditions. This same environment gradient effect on vegetation is common in many other studies wherein diversity increases due to differences in regional climate, elevation, and topography (Dorji et al. 2014, Moeslund et al. 2013, Perelman et al. 2017).

Observed variation (i.e., heterogeneity) between ecosites provided a baseline assessment of landscape conditions within each pasture. Generally speaking, a pasture containing more ecosites should have more inherent heterogeneity. This is similar to other studies showing higher diversity in areas with more variable environmental conditions (Dufour et al. 2006, Fuhlendorf & Engle 2001, Moeslund et al. 2013). While it may be expected that larger pastures would have more ecosites, this was not always the case, as smaller heterogeneous pastures also occurred (Appendix E). Within ecosites, there were also notable differences in plant composition and diversity, suggesting these grasslands had experienced substantial disturbance-induced changes in vegetation that had the potential to be linked to ongoing land uses such as grazing. Grazing is a factor that can change species composition within a landscape (Briske et al. 2008, Grime 1973, Milchulnas et al. 1988). For example, grazing has been shown to affect diversity both positively and negatively, depending on the climate, evolutionary grazing history of the area, and degree of recent grazing (Milchunas et al. 1988). In this study, both rangeland health and AHM affected plant species richness and evenness for some ecosites, with both evenness and richness usually declining as aridity increased, and responding in a varied manner to range health depending on climate and the type of ecosite. Ecosites not affected (i.e., Blowout, Clayey, and Sands) occurred on limited ranches and may not have had enough range of variation to detect change.

3.4.2   Plant Community Type to Ecosite (PCT:ES) Ratio

Plant community type classification involves identifying unique community assemblages (plant composition and abundance) within ecosites that stabilize under the current climate and disturbance regime (Adams et al. 2003, 2013a, 2013b, Westoby et al. 1989). To assess grazing

effects on overall pasture heterogeneity as reflected by the inventory data collected, we computed a metric of community patchiness, characterized as the number of different plant community types per ecosite (PCT:ES), on the premise that if disturbances (largely grazing) were uniform (either heavy or light), these would be more likely to be equal (i.e., number of PCT's = ecosites). Conversely, we hypothesized that moderate levels of grazing would lead to an increase in community patchiness due to non-uniform utilization of the vegetation within each ecosite which affects its composition. Thus, moderate grazing would increase the number of plant community types found per ecosite, until such point that all areas were evenly (and negatively) impacted due to excessive disturbance. As range health was our measure of disturbance effects (in this case grazing), we then hypothesized that greater patchiness in plant community types would be associated with a greater breadth of associated range health values. We also performed this independently on single well represented ecosites to test whether all ecosites behaved similarly.

In this study, the least heterogeneous pastures (those with low PCT:ES ratio) occurred when overall pasture mean rangeland health was very high (above 90%) or unhealthy (below 50%). Other reports have similarly noted that both little to no grazing and heavily grazed areas are capable of reducing vegetation heterogeneity, and ultimately plant diversity (Fuhlendorf & Engle 2001, Knapp et al. 1999). In contrast, the greatest heterogeneity was evident in our study area within pastures whose average weighted range health scores were 70-90%. Once again, this finding is consistent with studies concluding that uneven grazing impacts across a single pasture are an important mechanism enhancing vegetation heterogeneity (Adler et al. 2000, Lwiwski et al. 2015, Sliwinski & Koper 2015, Teague & Dowhower 2003, Virk & Mitchel 2014).

Along with moderate health scores, this trend was further evident in our study where higher heterogeneity was associated with a broader range of rangeland health scores. This result suggests that increasing patchiness within pastures arises in large part due to more variable rangeland health, and presumably corresponding livestock use patterns, compared to pastures that are managed with a single level of health. This in turn related to higher overall diversity measures at the pasture level for most ecosites. It should be noted, however, that patchiness was at least partly a function of pasture size, with patchiness inherently lower in smaller pastures. In contrast, larger pastures are more susceptible to variable grazing effects as livestock distribution is less controlled than in smaller pastures (Holechek et al. 1989, Teague & Dowhower 2003, Teague et al. 2013).

The results found here indicate that existing contemporary range health protocols in use for grasslands in Alberta (Adams et al. 2016) are capable of capturing this basic vegetation heterogeneity in grasslands, and thus, may have application for enhancing biodiversity conservation. To this end, it seems that if there is an array of grassland plant communities throughout a pasture that are in reasonably high health (i.e., averaging 70-90%) that this is likely to lead to the greatest patchiness of plant community types within ecosites across the landscape. This increase in patchiness has been shown to provide a large diversity of habitat that increases the amount and kinds of wildlife such as songbirds (Bock et al. 1993, Sliwinski & Koper 2015). One specific species of bird may be obligate to a certain habitat, but having multiple habitats increases the opportunity for a diversity of bird species.

As the ecosites examined here differed in physical attributes, it should be expected that they responded differently to changes in rangeland health. For example, sandy soils react

differently compared to clayey soils due to differences in infiltration, moisture retention and compaction (Jabro et al. 2009, Larson et al. 1980). Similarly, Blowout communities are affected by hardpans as well as salts, and have high bare soil and limited vegetation to begin with (Adams et al. 2013a, Adams et al. 2013b). This was seen when ecosites were analyzed independently for plant community type changes and range health. As the range of health values increased (delta pasture health) within pastures, a stronger relationship occurred for the number of plant communities in the ecosite with the greatest local moisture supply (i.e., Overflow). Varied range health influenced these Overflow ecosites much more than in Blowout ecosites, where moisture supply was lower and growing conditions much poorer. An explanation for this discrepancy can be found in Milchunas et al. (1988), who suggest plant species composition changes can shift more readily with even modest grazing in areas with greater moisture. Other studies also indicate that uplands and lowlands react differently to grazing pressure (Hartnett et al. 1996, Lwiwski et al. 2015), as do studies in dry and mesic geographic areas (Milchunas & Lauenroth 1993, Willms et al. 1985, Willms et al. 2002).

Interestingly, while the Overflow ecosite may have been the most sensitive to rangeland health changes in this dataset, increases in plant community types within a pasture showed the least changes in plant diversity. In contrast, both the Blowout and Loamy ecosites showed notably higher pasture levels of species richness and evenness from increase in patchiness. This suggests that while differences in community types within the Overflow ecosite occurred due to dominant species shifts, the less dominant plant species within this particular ecosite may have remained, whereas dominant species shifts in Loamy and Blowout ecosites also incorporated a more substantial changeover of inherent plant species, including those less common. In

general, rare species have often been shown to define diversity (Whittaker 1965), and may be relatively more important in regulating diversity responses in drier ecosites. The link to diversity measures and landscape heterogeneity has been evident in other studies and often concluded that lands with heterogeneous vegetation are more diverse than those homogenous (Dufor et al. 2006, Fuhlendorf & Engle 2001, Perelman et al. 2017, Virk & Mitchel 2014). However, this data suggests that health changes to some ecosites (i.e., Loamy and Blowout ecosites) may be more apt to increase overall diversity than others.

3.5   *Conclusion*

Patchiness of vegetation occurred historically on North American grassland landscapes from uneven grazing by native large herbivores, and presently occurs in most rangelands grazed by livestock leading to increased overall heterogeneity (Fuhlendorf & Engle 2001, Knapp et al. 1999, Lwiwski et al. 2015, Teague & Dowhower 2003, Virk & Mitchel 2014). At the pasture scale, we defined community heterogeneity (i.e., patchiness) as the number of differing plant communities contained within each ecosite. To evaluate the effects of grazing-induced changes on plant communities, we assessed the variation in range health of plant communities at the pasture-scale. An increase in plant community types relative to the number of ecosites suggested that disturbances (such as grazing) were having variable spatial effects on a pasture relative to the baseline abiotic environmental factors (topography and soils).

This study found that differing amounts of plant communities are likely to occur within pastures due to the inherent heterogeneity of the pasture, and that a proportion of that

heterogeneity can be associated with grazing-induced changes in range health. This heterogeneity, in turn, could be further related to variation in plant diversity, including richness and evenness. Consequently, modifying the health of pastures through proper grazing practices may be an important means to both modify vegetation heterogeneity, as well as the underlying plant diversity associated with the communities therein. In order to manage patchiness through ongoing grazing, care must be taken to graze intense enough to facilitate some plant species composition changes among communities (i.e., patches), but not overgraze so extensively so that all plant communities are negatively effected. Additionally, it must be kept in mind that grazing effects on individual plant communities are unlikely to be consistent across ecosites. Utilizing range health assessments appears to be an effective way to quantify overall patchiness, where as long as overall pasture health is maintained, an acceptable level of health can be tied to the range (i.e., breadth) of health scores detected throughout the pasture, which in turn, is advantageous to diversity.

Table 3.1. Conceptual example of the proposed range health to community type patchiness concept. Assume a pasture is composed of 2 ecosites divided into 3 different areas (polygons). Scenario 1 shows all locations healthy and very little deviance of health. Scenario 2 shows the entire pasture as less healthy, yet the disturbance is equal across the pasture. Scenario 3 indicates the same overall health as Scenario 2, yet a wide variety of disturbance particularly in ecosite A. Lastly Scenario 4 shows an unhealthy pasture where all areas are significantly affected. The hypothesis here is that Scenarios 1,2, and 4 would have similar number of community types to ecosites (2), yet Scenario 3 (in bold) would have the highest changes to vegetation that would express as 3 different community types.

Area	Ecosite	Assessed Pasture Health			
		Scenario 1	Scenario 2	Scenario 3	Scenario 4
Polygon 1	A	90	70	**50**	40
Polygon 2	A	100	70	**90**	50
Polygon 3	B	95	70	**70**	45
Average Health		95	70	**70**	45
Potential number of Community Types		2	2	3	2

Table 3.2. Summary of the upland ecosites assessed for rangeland health in this study. Ecosites are each described by the total area assessed, number of transects completed, and number of different plant community types found therein.

Ecosite	Area (Ha)	Area (%)	Sampling transects	Community types
Badland	661.4	0.9%	18	17
Blowout	11014.6	15.6%	169	25
Clayey	3220.3	4.6%	38	7
Choppy Sandhills	691.7	1.0%	11	4
Gravel	829.8	1.2%	15	5
Limy	282.8	0.4%	7	6
Loamy	36655.0	51.9%	503	55
Overflow	6532.9	9.3%	161	53
Sand	1608.3	2.3%	41	15
Subirrigated	151.9	0.2%	10	7
Saline Lowlands	1724.6	2.4%	55	36
Shallow to Gravel	363.5	0.5%	9	5
Sandy	2790.2	4.0%	73	24
Thin Break	4042.3	5.7%	135	68
Totals	**70569.4**	**100%**	**1245**	**327**

Table 3.3. Modelled linear relationships between (a) plant species richness and (b) community evenness, with annual heat:moisture index (AHM), shown separately by individual ecosites. Results in bold denote significance (P ≤ 0.05).

		(a) Plant Species Richness vs AHM					(b) Community Evenness (alpha) vs AHM				
Ecosites	N	Estimate	Std. Error	t value	Pr(>|t|)	r²	Estimate	Std. Error	t value	Pr(>|t|)	r²
Badland	18	-0.286	0.221	-1.30	0.21	0.09	-0.001	0.003	-0.20	0.84	0.002
Blowout	169	0.032	0.076	0.43	0.67	0.67	-0.001	0.002	-0.51	0.61	0.001
Clayey	38	-0.926	0.295	-3.14	**0.003**	0.21	0.031	0.006	5.29	**<0.001**	0.44
Choppy Sandhills	11	0.271	0.318	0.85	0.42	0.07	-0.002	0.003	-0.52	0.61	0.03
Gravel	15	-0.464	0.183	-2.54	**0.02**	0.33	0.004	0.003	1.17	0.26	0.10
Limy	7	0.570	0.431	1.32	0.24	0.26	-0.008	0.006	-1.48	0.20	0.30
Loamy	503	-0.422	0.032	-13.0	**<0.001**	0.25	0.005	0.000	11.2	**<0.001**	0.20
Overflow	161	-0.383	0.066	-5.79	**<0.001**	0.17	0.007	0.002	4.50	**<0.001**	0.11
Sands	41	0.097	0.130	0.75	0.46	0.01	-0.003	0.002	-1.29	0.20	0.04
Subirrigated	10	-0.441	0.225	-1.96	0.08	0.32	0.018	0.008	2.12	0.07	0.36
Saline Lowland	55	-0.137	0.113	-1.21	0.23	0.03	0.005	0.004	1.28	0.21	0.03
Shallow to Gravel	9	-0.189	0.364	-0.52	0.62	0.04	0.003	0.006	0.42	0.69	0.02
Sandy	73	-0.254	0.212	-1.20	0.24	0.02	0.002	0.003	0.65	0.52	0.01
Thin Break	135	-0.323	0.074	-4.34	**<0.001**	0.12	0.003	0.001	3.31	**0.001**	0.08

Table 3.4. Results of the permutational multivariate analysis of variance (PERMANOVA) reporting on compositional differences among plant community types grouped within ecosites.

Ecosite	Df	Sums Of Sqs	Mean Sqs	F.Model	R2	Pr(>F)
Badland	16	4.88	0.305	2.01	0.970	0.03
Blowout	24	15.80	0.659	6.42	0.512	0.001
Clayey	6	3.24	0.540	5.40	0.511	0.001
Choppy Sandhills	3	1.65	0.551	4.80	0.673	0.001
Gravel	4	1.28	0.319	2.81	0.529	0.002
Limy	5	1.37	0.275	4.27	0.955	0.047
Loamy	54	75.62	1.40	11.95	0.590	0.001
Overflow	52	39.01	0.750	5.24	0.716	0.001
Sand	14	6.81	0.486	4.08	0.687	0.001
Subirrigated	6	2.84	0.473	6.01	0.923	0.002
Saline Lowland	35	16.65	0.476	4.08	0.882	0.001
Shallow to Gravel	4	1.31	0.326	3.45	0.775	0.020
Sandy	23	11.94	0.519	4.03	0.654	0.001
Thin Break	67	31.53	0.471	4.10	0.804	0.001

Table 3.5. Variation in plant species richness and community evenness in relation to AHM, total range health scores, and their interaction, based on a mixed model analysis. Analyses were conducted separately for each ecosite. Results in bold denote significance (P ≤ 0.05).

| Ecosite		Plant Species Richness			Community Evenness (alpha)		
		AHM	Health	AHM:Health	AHM	Health	AHM:Health
	Df	1	1	1	1	1	1
Blowout	F	1.42	2.02	1.82	1.77	2.22	1.87
	Pr(>F)	0.235	0.157	0.179	0.185	0.139	0.173
Clayey	F	0.00	0.02	0.01	0.01	0.08	0.08
	Pr(>F)	0.961	0.89	0.906	0.918	0.782	0.787
Loamy	F	7.57	0.01	0.10	30.9	9.46	12.2
	Pr(>F)	**0.006**	0.910	0.757	**<0.001**	**0.002**	**0.001**
Overflow	F	4.35	0.31	0.47	6.65	2.30	2.64
	Pr(>F)	**0.039**	0.576	0.494	**0.011**	0.131	0.106
Sands	F	0.30	0.01	0.03	0.06	0.01	0.01
	Pr(>F)	0.585	0.919	0.855	0.816	0.942	0.933
Saline	F	4.34	3.95	3.75	2.57	2.08	2.05
Lowlands	Pr(>F)	**0.043**	0.053	0.059	0.116	0.157	0.159
Sandy	F	1.07	0.72	0.62	0.11	0.04	0.02
	Pr(>F)	0.308	0.402	0.434	0.745	0.838	0.897
Thin Breaks	F	2.55	0.49	0.95	3.84	1.52	2.34
	Pr(>F)	0.113	0.483	0.332	0.052	0.220	0.128

Table 3.6. Relationship between the number of plant communities found and estimates of rangeland health based on (a) area weighted averages, and (b) the range of health scores (delta health) found within a pasture. Data were tested by a generalized linear mixed model for each ecosite separately. Weighted Range health was rescaled for balance. Bold values denote significance (P ≤ 0.05).

Ecosites	Fields	(a) Weighted Rangeland Health (rescaled)				(b) Delta Health (max-min)			
		Estimate	St. Error	z-value	Pr(>|z|)	Estimate	St. Error	z-value	Pr(>|z|)
Blowout	90	0.014	0.085	0.17	0.87	0.020	0.005	3.62	**<0.001**
Loamy	221	0.048	0.053	0.91	0.36	0.020	0.003	7.48	**<0.001**
Overflow	97	0.114	0.083	1.37	0.17	0.028	0.004	7.74	**<0.001**
Saline Lowland	45	0.042	0.146	0.29	0.77	0.034	0.009	3.74	**<0.001**
Thin Break	85	0.101	0.087	1.15	0.25	0.026	0.005	5.46	**<0.001**

Table 3.7. Pasture level variation in plant species richness and community evenness within ecosites occurring in 30 or more pastures. Variation was measured in relation to the number of different plant community types (PCT) that occurred within the ecosite, AHM, and their interaction, based on a mixed model analysis. Analyses were conducted separately for each ecosite. Results in bold denote significance ($P \leq 0.05$).

	Max PCT's		(a) Plant Species Richness			(b) Evenness (alpha)		
			PCT	AHM	PCT:AHM	PCT	AHM	PCT:AHM
		Df	1	1	1	1	1	1
Blowout (n=90)	5	F	10.2	9.39	13.8	3.04	3.36	4.37
		Pr(>F)	**0.002**	**0.004**	**<0.001**	0.085	0.071	**0.040**
Loamy (n=221)	5	F	27.5	1.29	11.20	2.59	11.7	0.01
		Pr(>F)	**<0.001**	0.253	**0.001**	0.109	**0.001**	0.943
Overflow (n=97)	6	F	2.73	6.97	0.05	0.20	12.7	1.64
		Pr(>F)	0.102	**0.012**	0.832	0.657	**0.001**	0.203
Saline Lowland (n=45)	5	F	7.68	8.32	8.42	0.04	0.18	0.08
		Pr(>F)	**0.009**	**0.006**	**0.006**	0.845	0.677	0.776
Thin Break (n=85)	6	F	2.57	1.18	0.21	3.56	1.53	8.60
		Pr(>F)	0.113	0.284	0.645	0.063	0.223	**0.004**
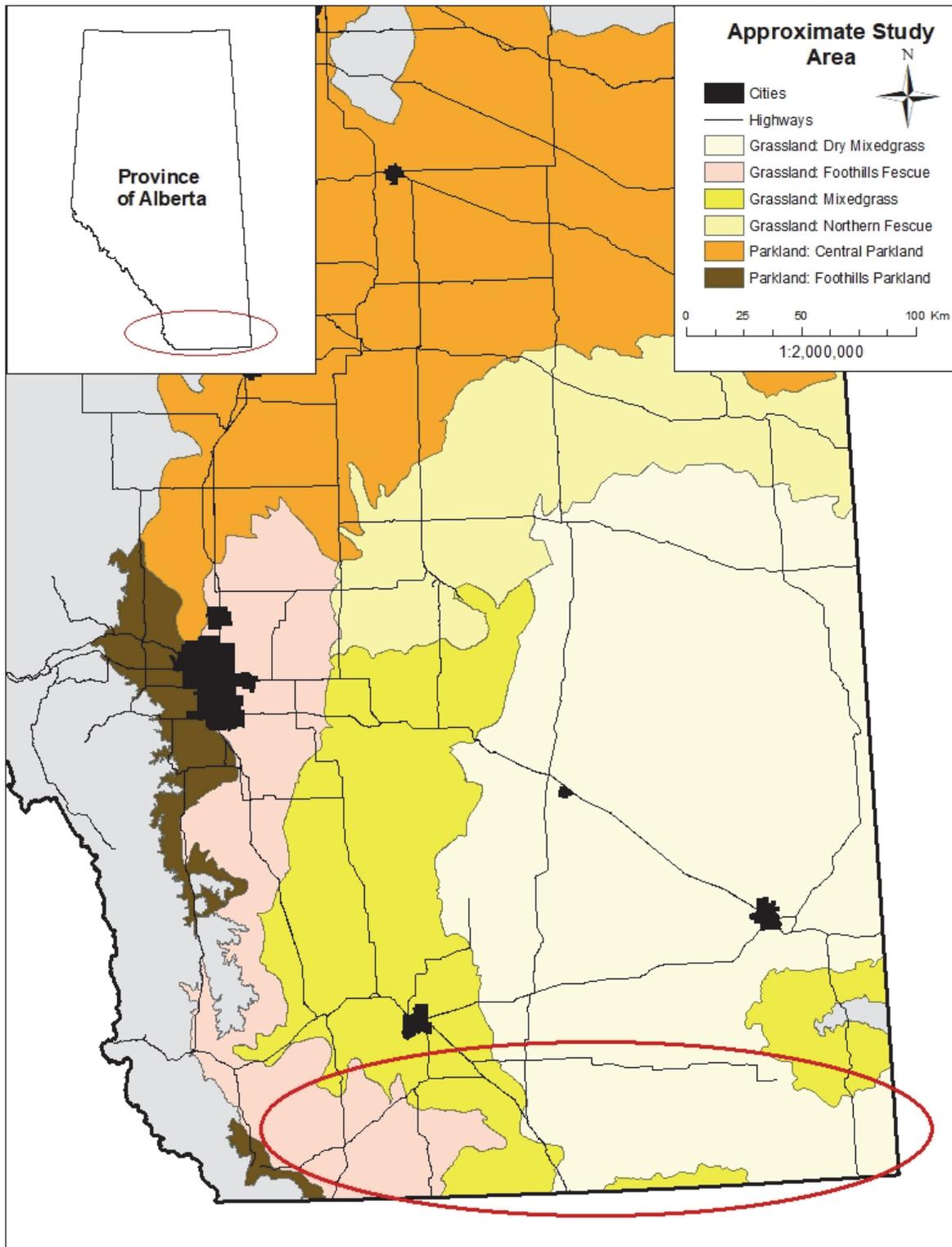
Figure 3.1. General area of MULTISAR pasture locations used in the pasture heterogeneity study. Natural regions and subregions are based on Alberta's natural subregion classification system (Downing and Pettapiece 2006).
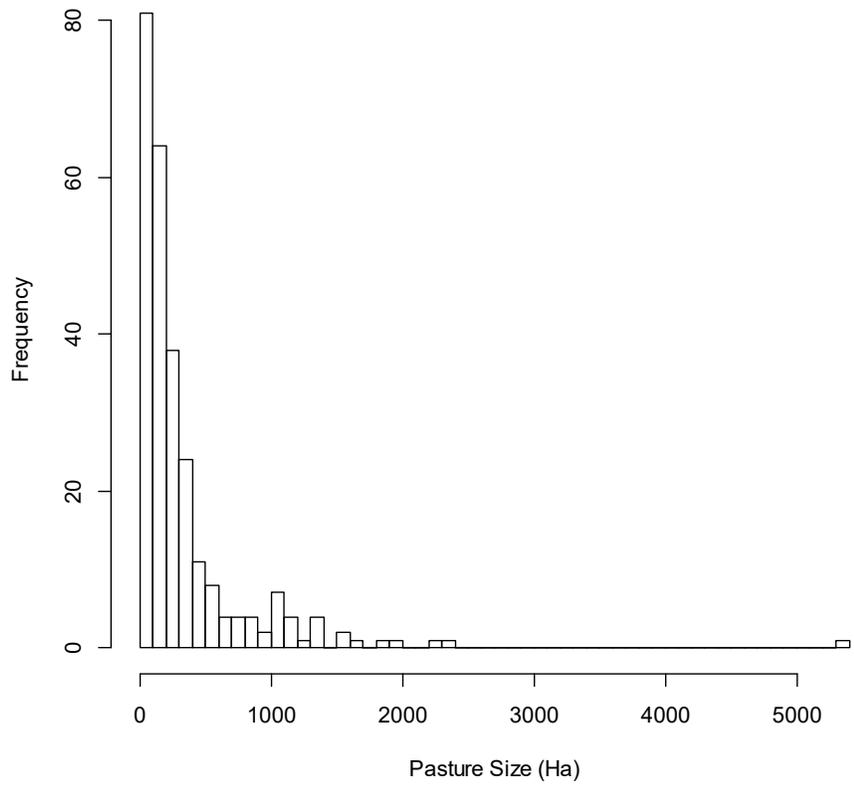
Figure 3.2. Histogram of the number of pastures sampled by different pasture size classes (hectares). The lone pasture greater than 5000 ha was considered an outlier from the rest of the pastures and was removed from the analysis.
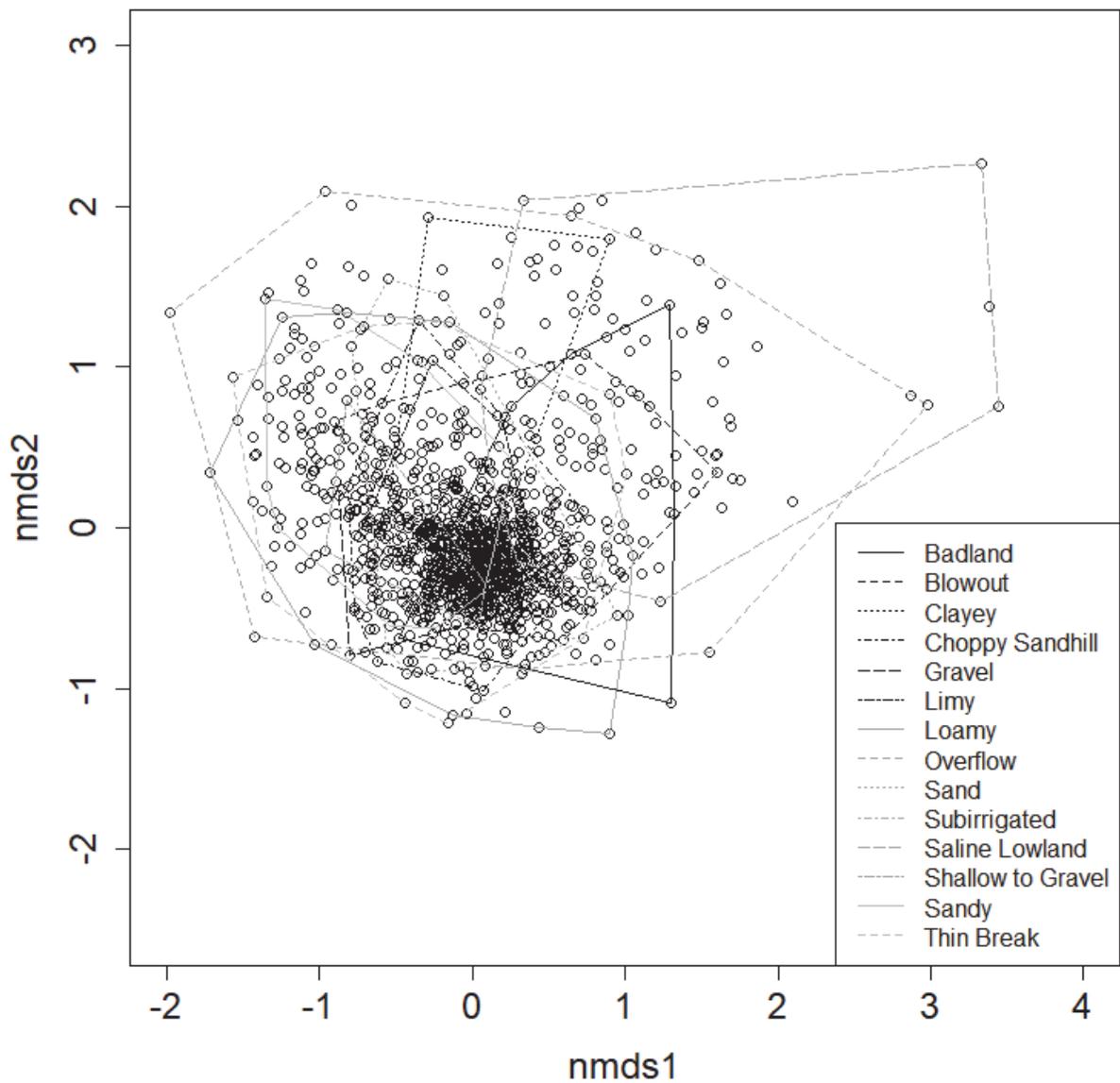
**Figure 3.3.** NMDS ordination of plant communities that occurred on ecosites across the study area. Polygons delineate the ordination space occupied by communities from each type of ecosite, as indicated in the caption. Although permutational multivariate analysis of variance (PERMANOVA) indicated significant differences between groups (P=0.001), visually there is marked overlap among ecosites, and a common centroid area.

a) Plant Species Richness: AHM - Ecosite
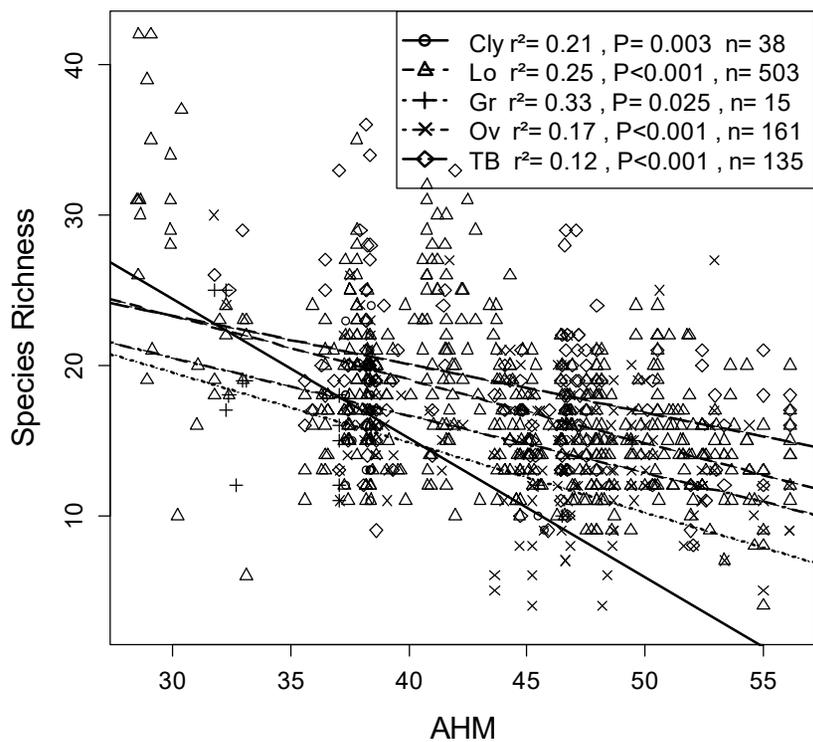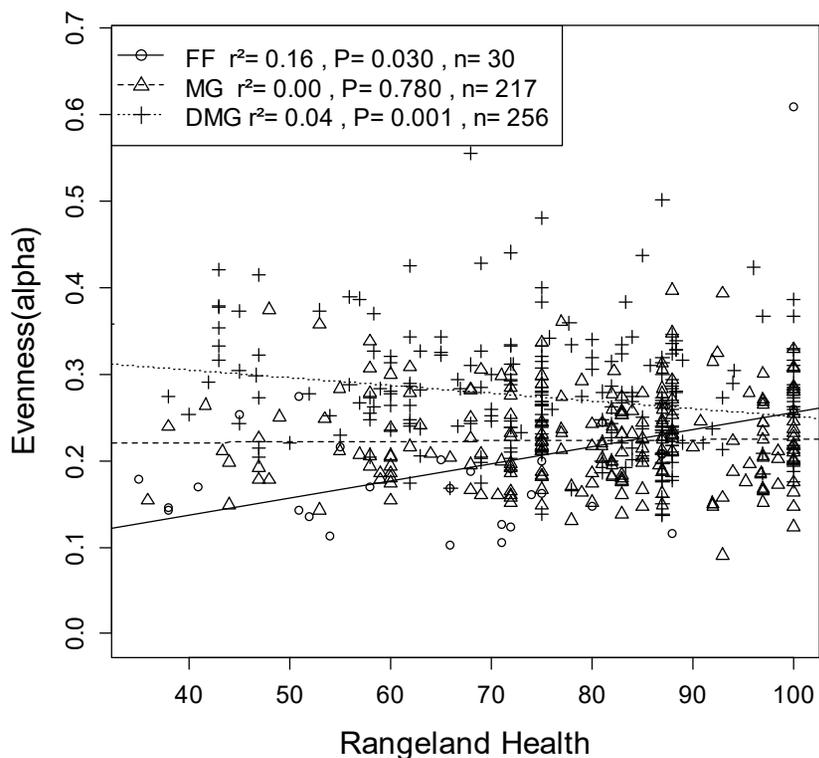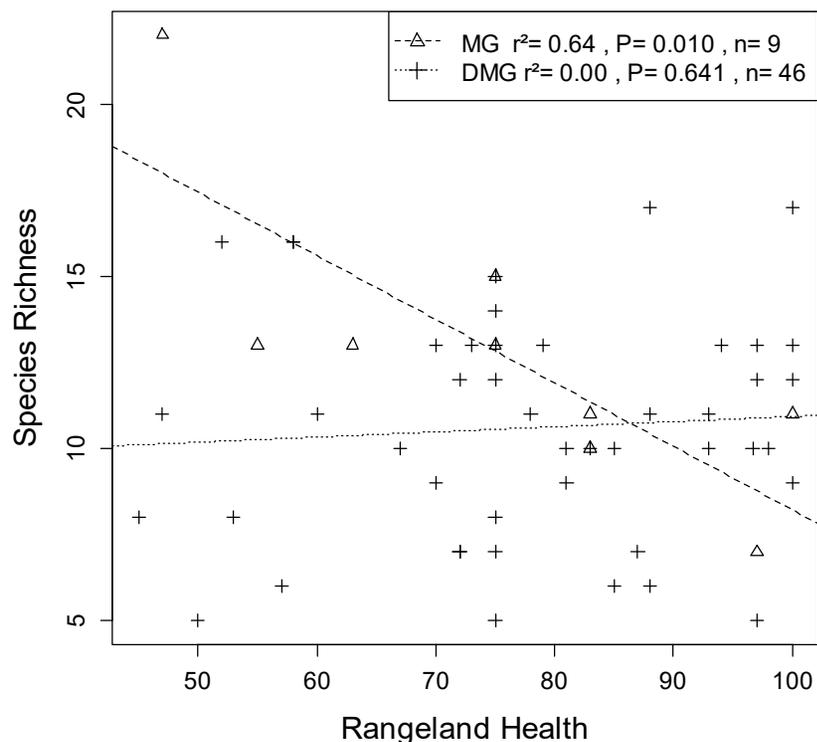
b) Community Evenness: AHM - Ecosite
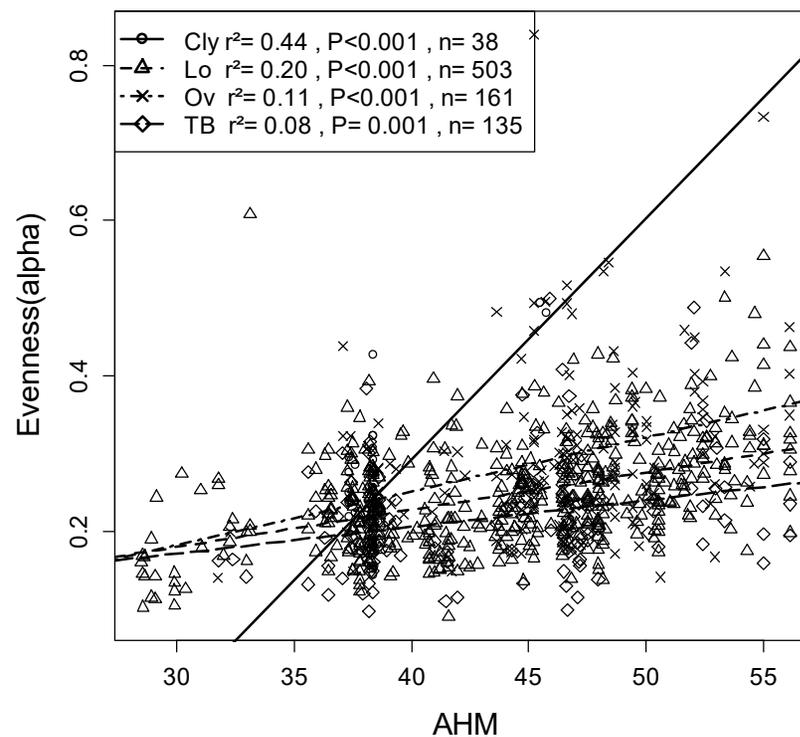
Figure 3.4. Relationship between (a) plant species richness and (b) plant community evenness (alpha) and the annual Heat:Moisture index. Responses are shown separately for ecosites that had a significant relationship (P ≤ 0.02, Table 3.3) across the AHM gradient. Evenness is measured by an alpha value where a lower alpha indicates greater evenness.

a) Loamy ecosite: AHM – Range health   b) Saline Lowland ecosite: AHM – Range health

Figure 3.5. Relationship between (a) plant community evenness, range health and AHM across Loamy ecosites, and (b) species richness, range health and AHM for Saline Lowland ecosites. To show the AHM by RH interaction with AHM the data are presented by different Natural Subregions, where AHM values were as follows: FF = 28 – 33, MG = 35 – 44, and DMG = 44 – 56. Evenness is measured by alpha values where a higher number indicates more dominance (i.e., lower evenness). For Saline Lowlands, the interaction of rangeland health and AHM was significant (trend, P=0.06, Table 3.5), and therefore these data are shown separately for the MG and DMG areas as the relationship with AHM yielded no relationship (P = 0.23). Included ecosites had relationships that were significant in a mixed model with individual pastures from each ranch property included as a random factors (P ≤ 0.05, Table 3.5).

a) Weighted Average Pasture Rangeland health
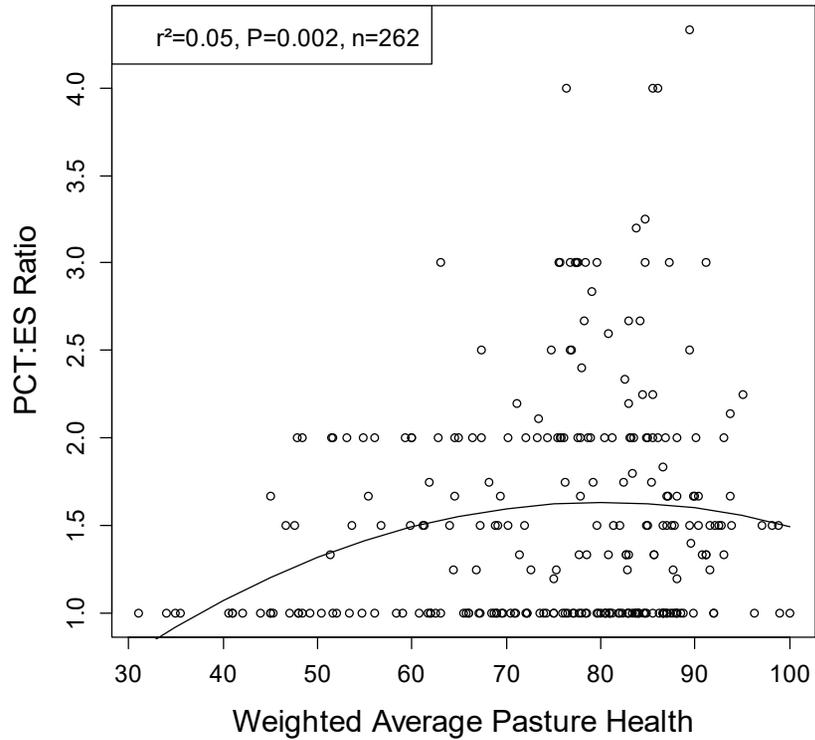
b) Delta Pasture Rangeland health (max – min)
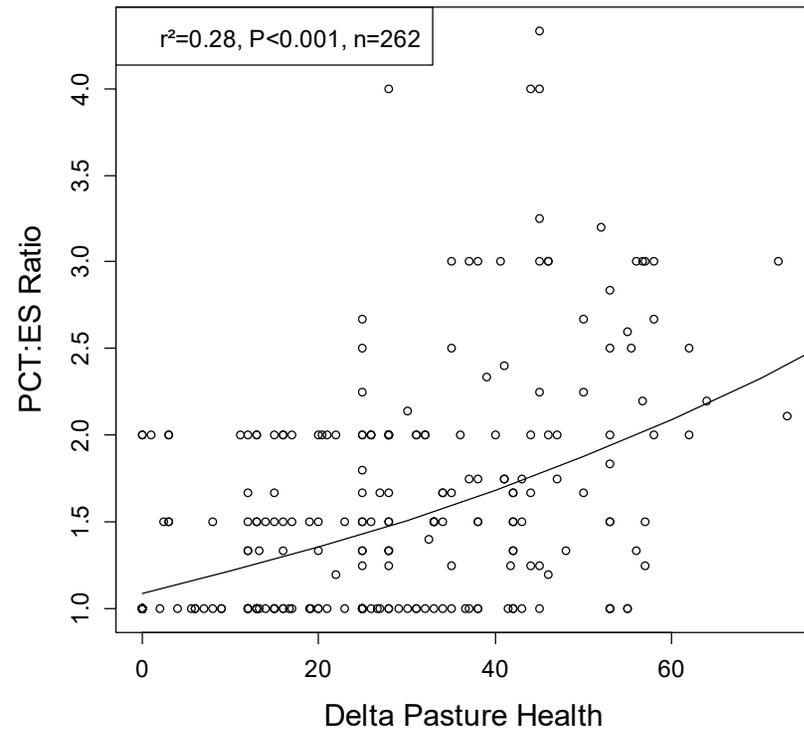
Figure 3.6. The relationship between plant community type to ecosite ratio (PCT:ES) and either (a) weighted range health or (b) the difference in range health scores occurring across a pasture (Delta Pasture Health).
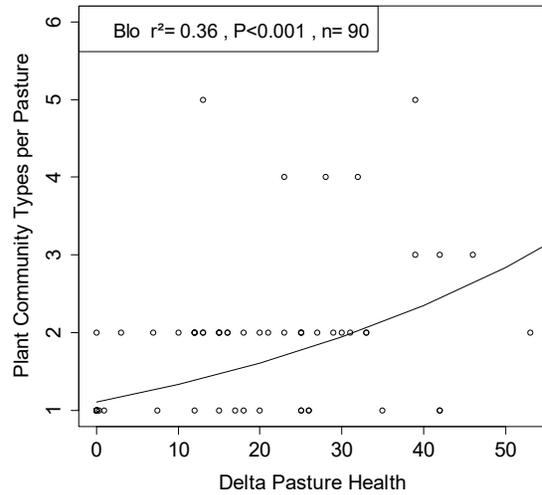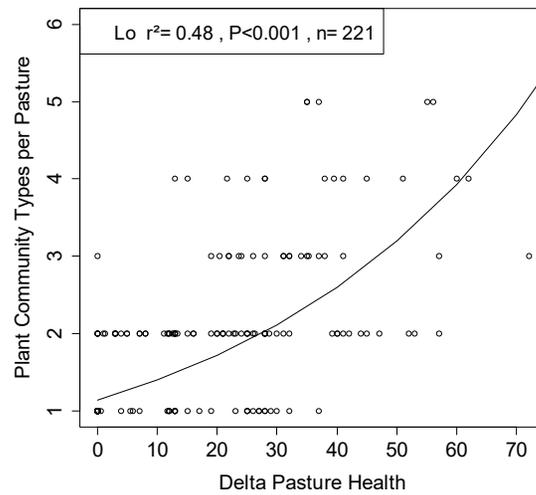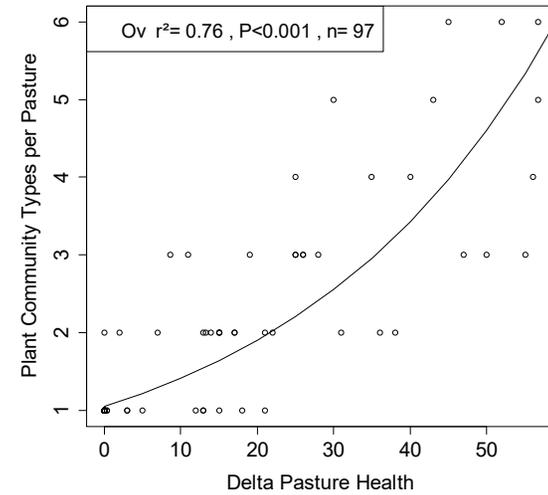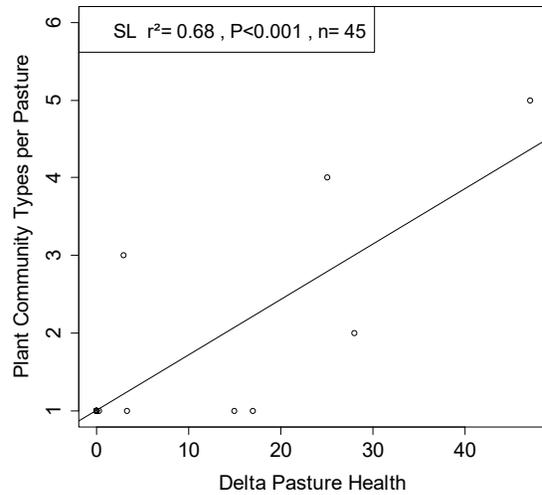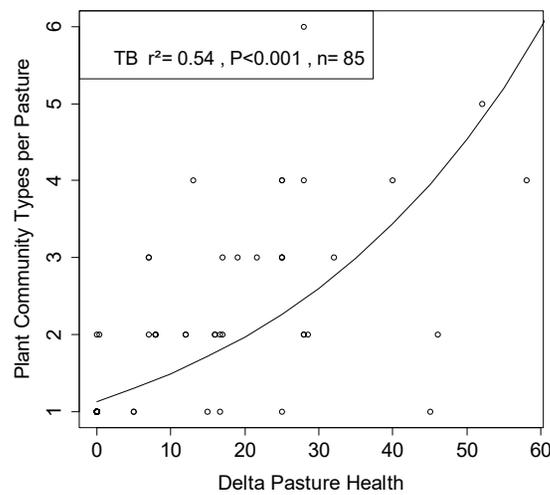
Figure 3.7. The relationship between the number of plant communities per pasture and delta pasture health (max vs min differences in range health) across a pasture. The ecosites presented are those found in at least 30 pastures across the study.

a) Blowout – Species Richness – AHM - PCT   b) Blowout - Community Evenness – AHM - PCT
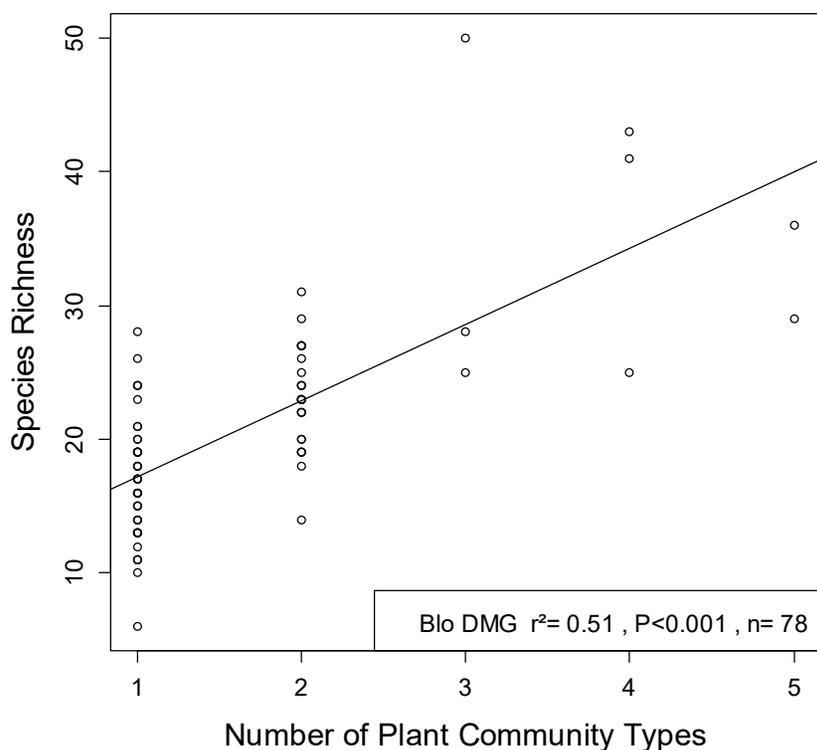
Blo DMG r²=0.28, p<0.001, n=78

Blo DMG r²= 0.51 , P<0.001 , n= 78
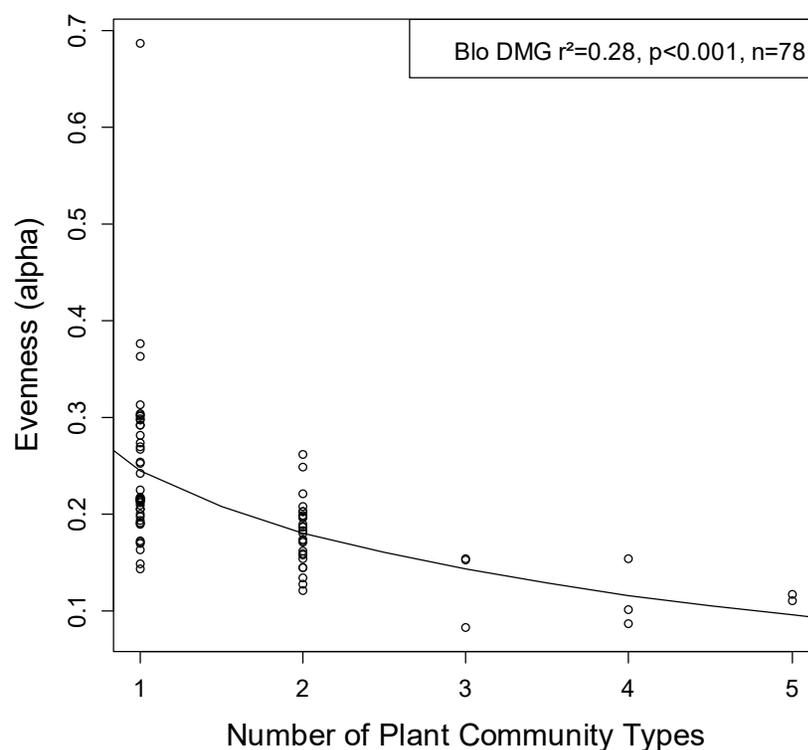
Figure 3.8. The relationship between (a) plant species richness, or (b) community evenness, and the number of plant community types found within the Blowout ecosite at the pasture level. Plant community types interacted with AHM (P ≤ 0.04, Table 3.7) so data are presented by different Natural Subregions. Ranges were as follows: Foothills Fescue (FF) = 28 – 33, Mixed Grass (MG) = 35 – 44, and Dry Mixed Grass (DMG) = 44 – 56. Only the DMG is shown as the MG did not relate to species richness (P=0.83, r²=0.004) or evenness (P=0.30, r²=0.01), and there were no plots in Blowout ecosites within the FF.

a) Loamy - Species Richness – AHM - PCT
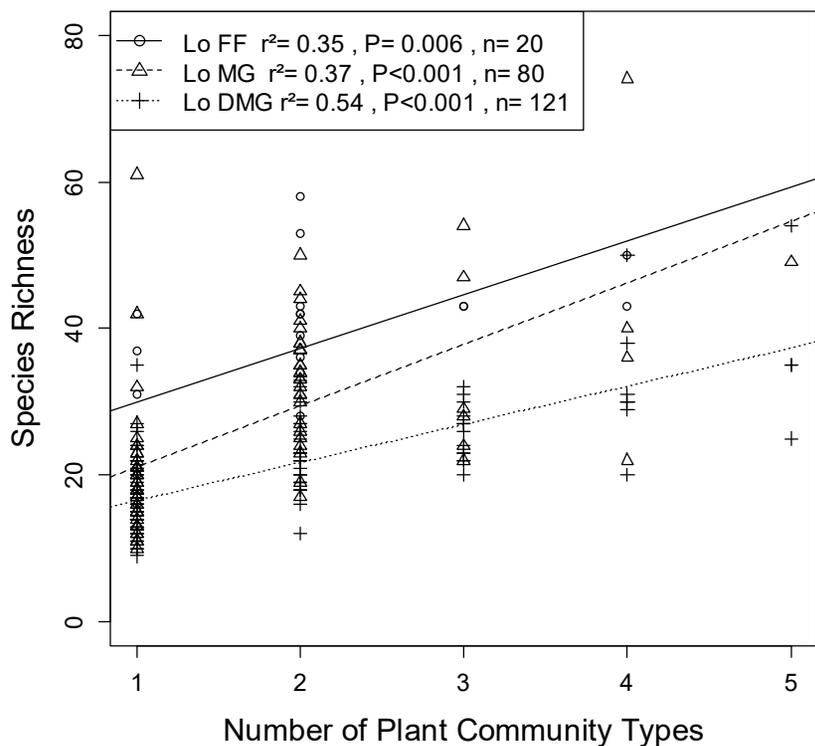
b) Loamy - Community Evenness – AHM
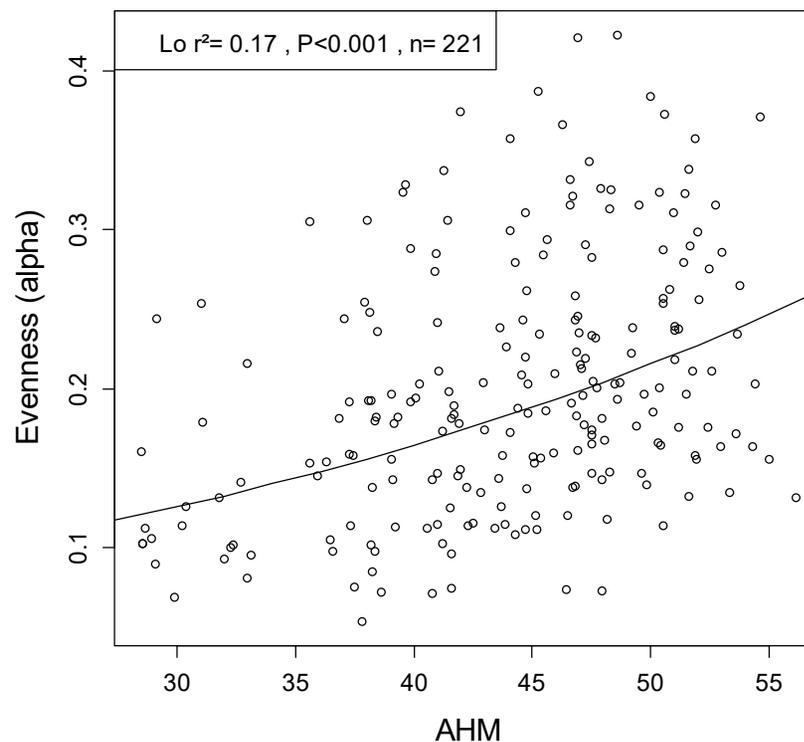
Figure 3.9. The relationship between (a) species richness and the interaction between the annual heat:moisture region and number of plant community types (PCT) within the Loamy ecosite at the pasture level, and (b) species evenness associated related to AHM, as evenness did not relate PCT (P = 0.11, Table 3.7). To show the interaction between PCT and AHM, data are presented by different Natural Subregions. Ranges were as follows: Foothills Fescue (FF) = 28 – 33, Mixed Grass (MG) = 35-44, and Dry Mixed Grass (DMG) = 44-56.

a) Overflow - Species Richness – AHM
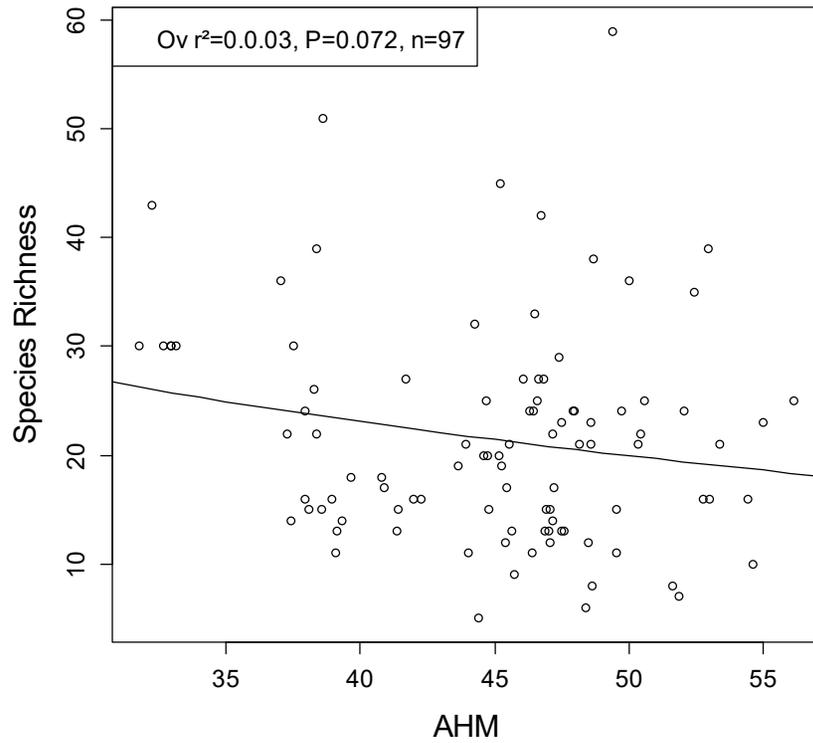
b) Overflow - Community Evenness - AHM
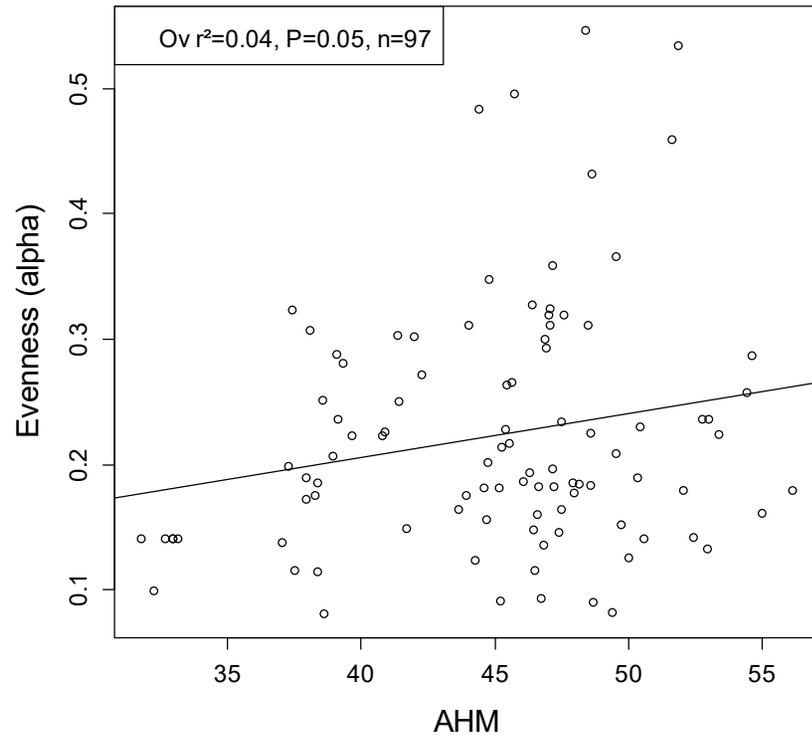
Figure 3.10. Relationships between a) plant species richness or b) evenness, and AHM across the overflow ecosites. These did not relate to the number of plant community types (PCT) so only AHM is shown (Table 3.7).

a) Saline Lowland - Species Richness – AHM - PCT
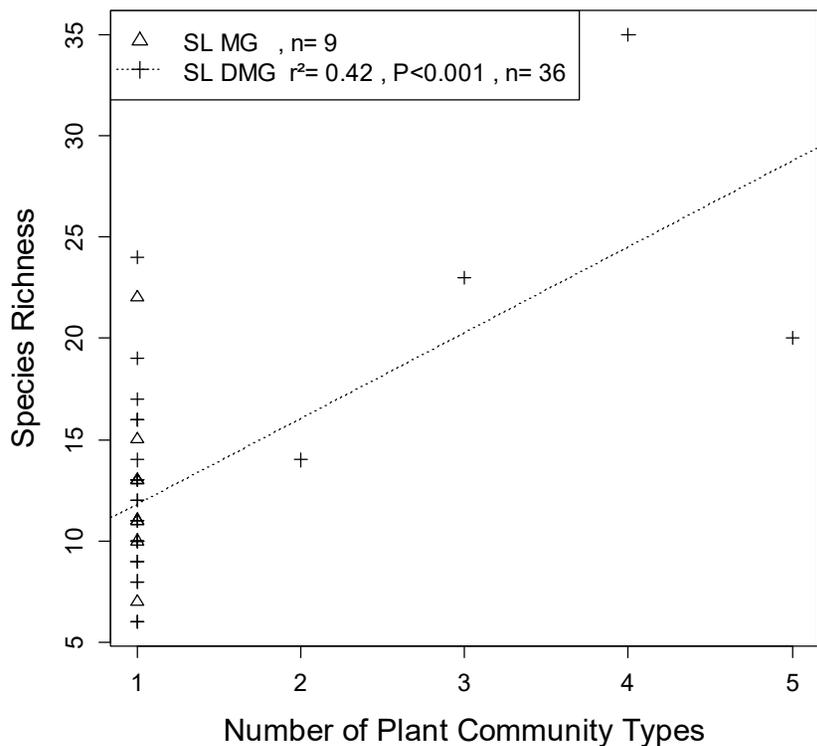
b) Thin Break – Community Evenness – AHM - PCT
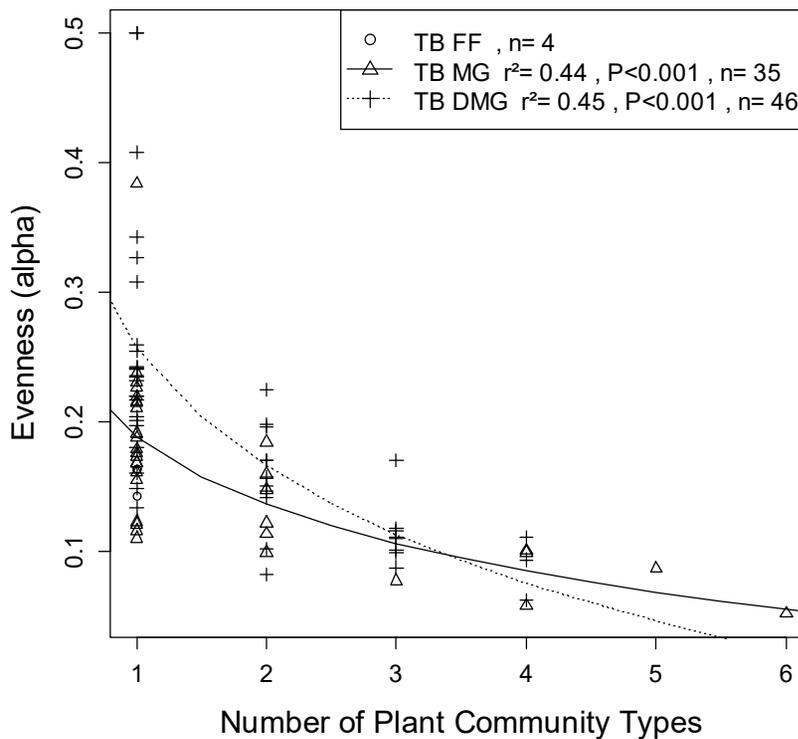
Figure 3.11. The relationship between (a) plant species richness in Saline Lowland, and (b) community evenness in Thin Break ecosites related to the number of plant community types (PCT) and the annual heat:moisture gradient (AHM) at the pasture level. To view the interaction, data are presented by different Natural Subregions to account to AHM. Ranges were as follows: Foothills Fescue (FF) = 28 – 33, Mixed Grass (MG) = 35 – 44, and Dry Mixed Grass (DMG) = 44 – 56. MG in Saline Lowland and FF in Thin Break only had pastures where PCT=1 so no trend line was created. There were no Saline Lowland plots in the FF.

4.0 Synthesis: Rangeland health as a tool for quantifying plant diversity

The terms biodiversity and rangeland health have inherent conservation values but are also attainable metrics (Adams et al. 2016, Baumgaertner et al. 2016, CRC 1994, Magurran 2004, West 1993). With these constructs comes an underlying assumption that maintaining or enhancing biodiversity and healthy rangelands is beneficial to overall conservation goals, and therefore the failure to value either can compromise ecological integrity (Adams et al. 2016, Aguiar 2005, Glowka et al. 1994, GOA 2014). It is important to separate studies of fact from projects practicing conservation strategies in order to maintain scientific credibility in the measures examined (Baumgaertner et al. 2016); therefore, to promote biodiversity or healthy rangelands, the empirical testing of meaningful relationships between these two metrics should be performed. It is often stated that conserving range health promotes a healthy and thus properly functioning ecosystem, and as such, if measures of range health are acceptable it is assumed that biodiversity is conserved (Adams et al. 2016, Symstad & Jonas 2011, TGUCT 1995). In practice, however, these two measures may not directly relate to one another. For example, a diverse community may have undesirable non-native plants, and controlling these plants may increase range health, but in the process, paradoxically reduces overall diversity. The objective of this study was to test the relationship between plant diversity and rangeland health using Alberta's range health assessments (Adams et al. 2016), with a primary focus on plant community diversity and community heterogeneity across pastures in native grasslands.

The relationship between diversity and rangeland health is important due to increased focus on the limited amount of native grasslands remaining in Western Canada. The conversion

of grasslands to cultivation, rural and industrial uses has reduced the cover of native vegetation and is a leading cause of declines in wildlife habitat (Bailey et al. 2010, Bennett 2003, CRC 1994, Lawler et al. 2014). As most remaining native grasslands are grazed by livestock, management of the latter is often implicated in biodiversity conservation (CRC 1994, West 1993). Grazing native grasslands are sometimes generalized with intensive livestock grazing systems that lead to reduced biodiversity (Eshel et al. 2014). Additionally, livestock grazing on native grasslands has been considered detrimental to wildlife habitat, with poorly managed areas in particular used as examples (Fleischner 1994, Vavra 2005, West 1993), despite mounting evidence that some grazing is often needed to maintain optimal ecosystem function (Hewins et al. 2018, Knapp et al. 1999), including supporting biodiversity (Collins & Barber 1986, Fuhlendorf & Engle 2001, Milchunas et al. 1988, West 1993). Range health measures serve to rapidly assess the ecological impacts of disturbance, including grazing, on native vegetation, thereby promoting healthy ecosystem functions (CRC 1994, TGUCT 1995, Adams et al. 2016, Pellant et al. 2005). Establishing a relationship between biodiversity and range health is beneficial as biodiversity is an important metric for ecosystems, but is often onerous to collect (Magurran 2004).

Perhaps surprisingly, this study found that range health scores were not strongly associated with plant diversity at the plant community level. This finding is in contrast to other studies (i.e., Bai et al. 2001) that have shown direct linkages between plant diversity and 'Clementsian style' range condition scores (Dyksterhuis 1949). This result suggests that while the assessment of rangeland health has evolved, its implications to directly monitor plant diversity may have become less clear. Range health concepts have moved from strictly examining plant species composition, to the inclusion of more interpretative measures of

rangeland sustainability, such as vegetation structure, nutrient cycling, and site stability (Adams et al. 2016, CRC 1994, Pellant et al. 2005, TGUCT 1995). In addition, current range health assessments may consider sustainable (i.e., light) grazing as an acceptable component of the 'desired plant community', thereby providing a very different benchmark for interpreting biodiversity. Finally, in this study plant diversity was found to be influenced by local variation in topographic, soil and climatic differences, further adding complexity to the nature and interpretation of range health relationships with diversity.

Notably, plant diversity metrics did relate to select components of range health, but rather than community composition, it reflected features such as vegetation structure and site stability/erosion. Once structure was notably reduced, or soil erosion became apparent, select diversity measures declined. Also, understanding the implications of introduced plant species was beneficial, as they may of acted as substitutes for some ecological attributes such as diversity, yet in Alberta's range health assessment system these species consistently caused range health reductions (Adams et al. 2016). These patterns suggest that while range health is not an indicator of plant diversity in general, there is opportunity to identify and refine specific indicators of the health protocol that are reflective of ongoing grazing use.

On a landscape scale (pasture units), range health was found to be an indicator of community patchiness within ecosites. The highest levels of overall community patchiness were attained by having a wide quantitative range of health scores, although pasture averages needed to be maintained at 70 to 90% to promote patchiness yet still maintain 'healthy' grasslands overall. This follows other studies where differing grazing disturbances created increased patchiness in some climates that in turn, led to increased landscape diversity (Collins

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& Barber 1986, Fuhlendorf & Engle 2001, Milchunas et al. 1988, Lwiwski et al. 2015).

Therefore, management or policy scenarios may target lower levels of range health in select

areas in order to promote varying patches at the community level that produce a more diverse

landscape for a variety of wildlife species (Bock et al. 1993, Fuhlendorf et al. 2012, Swilinski &

Koper 2015). To manage this effectively, both knowledge of the inherent heterogeneity of the

pasture from abiotic attributes, and an understanding of the response of individual plant

communities to different levels of grazing is required, as this study showed the patchiness

effects among plant community types varied with the abiotic parameters the community

occurred on. Range health measures these responses and therefore has applicability as a

communication and monitoring tool to balance health and biodiversity.

As efforts continue to promote native grassland conservation, inventory work that

includes rangeland health is becoming common to quantify habitat or grazing capacity and

measure the current health of ecosystems (AEP 2018, Bestelmeyer et al. 2009, Pellant et al.

2005). This work is not just to ensure sustainable livestock production, but to facilitate the

proper management of other land use interests such as industrial disturbance (e.g., energy

extraction and development) and wildlife habitat. Examples in Alberta are the reclamation

criteria policy that uses a variant of range health to quantify the trajectory of recovery for

reclaimed lands (ESRD 2013), or the resource inventories MULTISAR have completed for

quantifying wildlife habitat and promoting management goals associated with biodiversity,

including species at risk (Rumbolt et al. 2011). Although not completed purely for scientific

study, MULTISAR inventories provide structured and detailed data that this study used to

scientifically test for management (i.e., grazing) effects on plant diversity. This could be further

developed to encompass specific diversity metrics and other taxa (such as wildlife data), so future inventories could use range health metrics for biodiversity management.

This study contributed to our overall understanding of the relationship between range health and grassland biodiversity by scientifically identifying the value and limitation of the range health assessment and its inherent components as potential proxies for plant diversity measures at two scales. Many questions, however, generated from this work are still left unanswered. Significant climatic variability affected plant diversity across the 124 plots and 16 ranches assessed, as well as varied responses due to the local moisture, soil and topographic attributes of each site. Critically, these factors overrode much of the observed relationships between plant diversity and rangeland health. Future work may entail a more regional approach in experimental design to isolate those areas most utilized and responsive to the relationship (i.e., Loamy ecosites), to better understand their immediate diversity responses to grazing . Moreover, the contribution of introduced species, especially in mesic areas, to diversity should be further explored. Finally, although the opportunistic sampling used in both data chapters showed that Alberta's native grasslands most often have acceptable range health scores (average 76% health across assessments and 75% across ranches sampled), a more systematic approach is recommended for future work to provide greater representation of the biodiversity-health relationships for those lands with lower health scores.

References

Adams, B.W., G. Ehlert, , C. Stone, M. Alexander, D. Lawrence, M. Willoughby, D. Moisey, C. Hincz, A. Burkinshaw, J. Richman, K. France, C. DeMaere, T. Kupsch, T. France, T. Broadbent, L. Blonski, and A.J. Miller. 2016. Rangeland health assessment for grasslands, forest & tame pasture. Field Workbook, Rangeland Resource Stewardship Section, Alberta Environment and Parks, Edmonton Alberta. 124 p.

Adams, B.W., J. Carlson, D. Milner, T. Hood, B. Cairns and P. Herzog. 2004. Beneficial grazing management practices for Sage-Grouse (*Centrocercus urophasianus*) and ecology of silver sagebrush (*Artemisia cana*) in southeastern Alberta. Technical Report Pub. No. T/049, Public Lands and Forests Division, Alberta Sustainable Resource Development, Lethbridge, Alberta. 60 p.

Adams, B.W., J. Richman, L. Poulin-Klein, K. France, D. Moisey, and R.L. McNeil. 2013a. Rangeland plant communities for the Dry Mixedgrass natural subregion of Alberta. 2nd Approximation. Pub. No. T/040, Rangeland Management Branch, Policy Division, Alberta Environment and Sustainable Resource Development, Lethbridge, Alberta. 135 p.

Adams, B.W., J. Richman, L. Poulin-Klein, K. France, D. Moisey, and R.L. McNeil. 2013b. Rangeland plant communities for the Mixedgrass natural subregion of Alberta. 2nd Approximation. Pub. No. T/039, Rangeland Management Branch, Policy Division, Alberta Environment and Sustainable Resource Development, Lethbridge, Alberta. 103 p.

Adams, B.W., R. Ehlert, D. Moisey, and R.L. McNeil. 2005. Rangeland Plant Communities and Range Health Assessment Guidelines for the Foothills Fescue Natural Subregion of Alberta. 2nd Approximation. Pub. No. T/044, Rangeland Management Branch, Public Lands Division, Alberta Sustainable Resource Development, Lethbridge, Alberta. 85 p.

Adler, P.B., D.A. Raff and W.K Lauenroth. 2000. The effect of grazing on the spatial heterogeneity of vegetation. Oecologia 128:465-479.

Aguiar, M.R. 2005. Biodiversity in grasslands. Current changes and future scenarios. Pp. 261-280, in: Chapter 11: Grasslands: Developments, Opportunities, Perspectives. Reynolds, S.G. and J. Frame (editors), Science Publishers Inc., Enfield, New Hampshire.

Alberta Biodiversity Monitoring Institute (ABMI). 2014. Terrestrial field data collection protocols (abridged version) 2015-02-19. Alberta Biodiversity Monitoring Institute, Alberta, Canada. 48 p.

Alberta Environment and Parks (AEP). 2018. Range inventory manual for forest reserve allotments and grazing leases within Rocky Mountain, Foothills, Parkland and Grassland natural regions. Government of Alberta, Edmonton, Alberta. 74 p.

Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology 26(1):32-46.

Anderson, M.J., K.E. Ellingsen, and B.H. McArdle. 2006. Multivariate dispersion as a measure of beta diversity. Ecology Letters 9:683–693.

Bai, Y., Z. Abouguendia and R.E. Redmann. 2001. Relationship between plant species diversity and grassland condition. Journal of Range Management 54(2):177-183.

Bailey, A.W., D. McCartney, and M.P. Schellenberg. 2010. Management of Canadian prairie rangeland. AAFC No. 10144, Agriculture and Agri-food Canada, Canada. 58 p.

Baumgaertner, B. and W. Holthuijzen. 2016. On nonepistemic values in conservation biology. Conservation Biology 31(1):48–55.

Belcher, J.W. and S.D. Wilson. 1989. Leafy spurge and the species composition of a mixed-grass prairie. Journal of Range Management 42(2):172-175.

Bennett, A.F. 2003. Linkages in the landscape. The role of corridors and connectivity in wildlife conservation. International Union for Conservation of Nature, Gland, Switzerland and Cambridge, England. 254 p.

Bestelmeyer, B.T., A.J. Tugel, G.L. Peacock, Jr., D.G. Robinett, P.L. Shaver, J.R. Brown, J.E. Herrick, H. Sanchez and K.M. Havstad. 2009. State-and-transition models for heterogeneous landscapes: a strategy for development and application. Rangeland Ecology and Management 62(1):1-15

Biodiversity Convention Office (BDO). 1995. Canadian biodiversity strategy. Canada's response to the Convention on Biological Diversity. Catalogue No. En21-134/1995E, Minister of Supply and Services Canada, Quebec, Canada. 86 p.

Bock, C.E., V.A. Saab, T.D. Rich, and D.S. Dobkin. 1993. Effects of livestock grazing on neotropical migratory landbirds in western North America. Pp. 296 to 309, in: Status and management of neotropical migratory birds: September 21-25, 1992, Finch, Deborah M. and Stangel, Peter W. (editors). General Technical Report RM-229, Rocky Mountain Forest and Range Experiment Station, U.S. Dept. of Agriculture, Forest Service, Fort Collins, Colorado.

Bolker, B.M., M.E. Brooks, C.J. Clark, S.W. Geange, J.R. Poulsen, M.H.H. Stevens and J.S. White. 2008. Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology and Evolution 24(3):127-135.

Briske, D.D., J.D. Derner, J.R. Brown, S.D. Fuhlendorf, W.R. Teague, K.M. Havstad, R.L. Gillen, A.J. Ash, and W.D. Willms. 2008. Rotational grazing on rangelands: Reconciliation of perception and experimental evidence. Rangeland Ecology and Management 61(1):3-17.

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

Chillo, V., R.A. Ojeda, M. Anand and J.F. Reynolds. 2015. A novel approach to assess livestock management effects on biodiversity of drylands. Ecological Indicators 50:69-78.

Collins, S.L. 1987. Interaction of disturbances in tallgrass prairie: a field experiment. Ecology 68(5):1243-1250.

Collins, S.L., and S.C. Barber. 1986. Effects of disturbance on diversity in mixed-Grass prairie. Vegetatio 64(2/3):87-94.

Committee on Rangeland Classification (CRC). 1994. Rangeland health: new methods to classify, inventory, and monitor. National Academy Press., Washington, D.C. 200 p.

Connell, J.H. 1978. Diversity in tropical rainforests and coral reefs. Science 199:1302 -1310.

Davis, M.A. , K. Thompson, J.P. Grime and A. Liebhold. 2005. Invasibility: the local mechanism driving community assembly and species diversity. Ecography 28(5):696-704.

Davis, M.A. 2003. Biotic globalization: does competition from introduced species threaten biodiversity? Bioscience 53:481-489.

De Caceres, M. and F. Jansen. 2016. Relationship between species and groups of sites. R Package verson 1.7.6. URL: https://CRAN.R.

DeMaere, C., M. Alexander, and M. Willoughby. 2012. Range plant communities and range health assessment guidelines for the Foothills Parklands Subregion of Alberta. Pub. No. T/274, Alberta Environment and Sustainable Resource Development, Pincher Creek, Alberta. 136 p.

Dorji, T., S.R. Moe, J.A. Klein and Ø. Totland. 2014. Plant species richness, evenness, and composition along environmental gradients in an alpine meadow grazing ecosystem in Central Tibet, China. Arctic, Antarctic, and Alpine Research 46(2):308-326.

Downey, B.L., Downey B.A., Quinlan, R.W., and Jones, P.F. 2005. MULTISAR: a multiple-species conservation strategy for species at risk. Year 3 Report. Alberta Species at Risk Report No. 98, Alberta Sustainable Resource Development, Fish and Wildlife Division, Edmonton, Alberta. 56 p.

Dufour, A., F. Gadallah, H.H. Wagner, A. Guisan and A. Buttler. 2006. Plant species richness and environmental heterogeneity in a mountain landscape: effects of variability and spatial configuration. Ecography 29(4):573-584.

Dyksterhuis, E. J. 1949. Condition and management of range land based on quantitative ecology. Journal of Range Management 2:104-115.

Ellison, L. 1960. Influence of grazing on plant succession of rangelands. Botanical Review 26(1):1-78.

Environment and Climate Change Canada (ECCC). 2016. 2020 biodiversity goals & targets for Canada. Government of Canada, ISBN: 978-0-660-04248-0.

Environment and Sustainable Resource Development (ESRD). 2013. 2010 reclamation criteria for wellsites and associated facilities for native grasslands, (July 2013 Update). Government of Alberta, Edmonton, Alberta. 92 p.

Eshel, G, A. Shepon, T. Makov, and R. Milo. 2014. Land, irrigation water, greenhouse gas, and reactive nitrogen burdens of meat, eggs, and dairy production in the United States. Proceedings of the National Academy of Sciences of the United States of America (PNAS) 111(33):11996-12001.

Evans, S. M. 2001. Grazing the grasslands: exploring conflicts, relationships and futures. Prairie Forum 26(1):67-84.

Fleischner, T.L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8(3):629-644.

Fox, J. and S. Weisberg. 2011. An {R} companion to applied regression, 2nd Edition. Thousand Oaks CA: Sage.   URL: http://socserv.socsci.mcmaster.ca/jfox/Books/Companion

Friedel M.H. 1991. Range condition assessment and the concept of thresholds: a viewpoint. Journal of Range Management 44(5):422-426.

Fuhlendorf, S.D. and D.M. Engle. 2001. Restoring heterogeneity on rangelands: ecosystem management based on evolutionary grazing patterns. BioScience 51(8):625-632.

Fuhlendorf, S.D. and D.M. Engle. 2004. Application of the fire - grazing interaction to restore a shifting mosaic on tallgrass prairie. Journal of Applied Ecology 41:604-614.

Fuhlendorf, S.D. and F.E. Smeins. 1999. Scaling effects of grazing in a semi-arid grassland. Journal of Vegetation Science 10: 731-738.

Fuhlendorf, S.D., D.M. Engle, R.D. Elmore, R.F. Limb, and T.G. Bidwell. 2012. Conservation of pattern and process: developing an alternative paradigm of rangeland management. Rangeland Ecology and Management 65(6):579-589.

Fynn, R.W.S and T. G. O'Connor. 2000. Effect of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. Journal of Applied Ecology 37(3):491-507.

Glowka, L., F. Burhenne-Guilmin, H. Synge, J.A. McNeely, and L. Gundling. 1994. A Guide to the convention on biological diversity. Environmental Policy and Law Paper No. 30, International Union for Conservation of Nature, Gland, Switzerland and Cambridge, England. 161 p.

Goswami, M., P. Bhattacharyya, I. Mukherjee, P. Tribed. 2017. Functional diversity: an important measure of ecosystem functioning. Advances in Microbiology 7:82-93.

Government of Alberta (GOA). 2004. Forest Reserves Act. Alberta Queen's Printer, Edmonton, Alberta. 4 p.

Government of Alberta (GOA). 2011. Grassland vegetation inventory (GVI) specifications, 5th edition. Alberta Sustainable Resource Development, Edmonton, Alberta. 88 p.

Government of Alberta (GOA). 2014. South Saskatchewan regional plan 2014-2024. Edmonton, Alberta. 201 p.

Greene, R.P., and J. Stager. 2001. Rangeland to cropland conversions as replacement land for prime farmland lost to urban development. The Social Science Journal 38:543-555.

Grime, J.P. 1973. Competitive exclusion in herbaceous vegetation. Nature 242: 344-347.

Gurevitch, J. and D.K. Padilla. 2004. Are invasive species a major cause of extinctions? Trends in Ecology and Evolution 19:470-474.

Hartnett, D.C., K.R. Hickman and L.E. Fischer Walter. 1996. Effects of bison grazing, fire, and topography on floristic diversity in Tallgrass Prairie. Journal of Range Management 49(5):413-420.

Hejda, M., P.Pyšek and V. Jarošík. 2009. Impact of invasive plants on the species richness, diversity and composition of invaded communities. Journal of Ecology 97(3):393-403.

Hervé, Maxime. 2017. RVAideMemoire: Testing and plotting procedures for biostatistics. R package version 0.9-68. URL: https://CRAN.R-project.org/package=RVAideMemoir

Hewins, D.B., M.P. Lynseng, D.F Schoderbek, M. Alexander, W.D. Willms, C.N. Carlyle, S.X. Chang, and E.W. Bork. 2018. Grazing and climate effects on soil organic carbon concentration and particle-size association in northern grasslands. Scientific Reports 8:1336.

Holechek, J.L., R.D. Pieper, and C.H. Herbel. 1989. Range management principles and practices. Pearson Prentice Hall, Upper Saddle River, New Jersey. 504 p.

Jabro, J.D., R.G. Evans, Y. Kim, and W.M. Iversen. 2009. Estimating in situ soil–water retention and field water capacity in two contrasting soil textures. Irrigation Science 27:223-229.

Knapp, A.K., J.M. Blair, J.M. Briggs, S.L. Collins, D.C. Hartnett, L.C. Johnson, and E.G. Towne. 1999. The keystone role of bison in North America tall grass prairie. BioScience 49(1):39-50.

Kostenko, O., P.P.J. Mulder, M. Courbois, and T.M. Bezemer. 2017. Effects of plant diversity on the concentration of secondary plant metabolites and the density of arthropods on focal plants in the field. Journal of Ecology 105:647-660.

Kupsch, T., K. France, H. Loonen, A. Burkenshaw, M.G. Willoughby and R.L. McNeil. 2013. Range plant communities and range health assessment guidelines for the Central Parkland subregion of Alberta, 2nd Approximation. Pub No. T/125, Alberta Sustainable Resource Development, Public Lands and Forests Division, Edmonton, Alberta. 213 p.

Kupsch, T., K. France, J. Richman, and R.L. McNeil. 2012. Range plant communities and range health assessment guidelines for the Northern Fescue subregion of Alberta, 1st approximation. Pub. No. T/265, Alberta Sustainable Resource Development, Red Deer, Alberta. 77 p.

Lamb, E.G. 2008. Direct and indirect control of grassland community structure by litter, resources, and biomass. Ecology 89(1):216-225.

Larson, W.E., S.C. Gupta, and R.A. Useche. 1980. Compression of agricultural soils from eight soil orders. Soil Science Society of America Journal 44:450-457.

Lawler, J.J., D.J. Lewis, E. Nelson, A.J. Plantinga, S. Polasky, J.C. Witheyf, D.P. Helmers, S. Martinuzzi, D. Pennington, and V.C. Radeloff. 2014. Projected land-use change impacts on ecosystem services in the United States. Proceedings of the National Academy of Sciences of the United States of America (PNAS) 111(20): 7492–7497.

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.

Loreau, Michel. 2000. Biodiversity and ecosystem functioning: recent theoretical advances. Oikos 91(1):3-17.

Lwiwski, T.C., N. Koper, and D.C. Henderson. 2015. Stocking rates and vegetation structure, heterogeneity, and community in a northern mixed-grass prairie. Rangeland Ecology and Management 68(4):322-331.

Magurran, A.E. 2004. Measuring biological diversity. Blackwell Publishing, Malden, Massachusetts, USA. 256 p.

Martin-Fores, I., G.R. Guerin, and A.J. Lowe. 2017. Weed abundance is positively correlated with native plant diversity in grasslands of southern Australia. Public Library of Science (PLOS) ONE 12(6):e0178681.

Marty, J.T. 2015. Loss of biodiversity and hydrologic function in seasonal wetlands persists over 10 years of livestock grazing removal. Restoration Ecology 23(5):548-554.

Maynard, C.L., R.L. Lawrence, G.A. Nielsen, and G. Decker. 2007. Ecological site descriptions and remotely sensed imagery as a tool for rangeland evaluation. Canadian Journal of Remote Sensing 33(2):109-115.

Mbogga, M., T. Wang, C. Hansen and A. Hamann. 2010. A comprehensive set of interpolated climate data for Alberta. Pub. No. T/235, Alberta Sustainable Resource Development, Edmonton, Alberta. 7 p.

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

Milchunas, D.G. and W. K. Lauenroth. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecological Monographs 63(4):327-366.

Moeslund, J.E., L. Arge, P. Klith Bøcher, T. Dalgaard, R. Ejrnæs, M. Vestergaard Odgaard, J. Svenning. 2013. Topographically controlled soil moisture drives plant diversity patterns within grasslands. Biodivers Conserv 22:2151-2166.

Morgan, R.G. 1980. Bison movement patterns on the Canadian plains: an ecological analysis. Plains Anthropologist 25(88):143-160.

Moss, E.H. 1983. The Flora of Alberta. University of Toronto Press, Toronto, Ontario. 687 p.

MULTISAR. 2018. MULTISAR: A Multi-Species Conservation Strategy for Species at Risk in Grassland Natural Region of Alberta 2017-2018 Report. Alberta Environment and Parks, Fish and Wildlife Policy Branch, Alberta Species at Risk Report No. 162, Edmonton, AB. 39 p.

Narwani, T, and W.E. Schieck. 2015. The Alberta biodiversity monitoring institute: value to Alberta. Alberta Biodiversity Monitoring Institute, Edmonton, Alberta. 8 p.

Downing, D.J. and W.W. Pettapiece. 2006. Natural regions and subregions of Alberta. Pub. No. T/852. Natural Regions Committee, Government of Alberta, Edmonton, Alberta. 254 p.

Noss, R.F. 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology 4(4):355-364.

Noss, R.F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the greater Yellowstone ecosystem. Conservation Biology 16(4):895-908.

Oksanen, J. 2015. Multivariate analysis of ecological communities in R: vegan tutorial. URL: http://cc.oulu.fi/~jarioksa/opetus/metodi/vegantutor.pdf

Oksanen, J., G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P.R. Minchin, R.B. O'Hara, G.L. Simpson, P. Solymos, M.H.H. Stevens, E. Szoecs and H. Wagner. 2017. vegan: community ecology package. R package version 2.4-1. URL: https://CRAN.R-project.org/package=vegan

Pellant, M., P. Shaver, D.A. Pyke, and J.E. Herrick. 2005. Interpreting indicators of rangeland health, version 4. Technical Reference 1734-6, U.S. Department of the Interior, Bureau of Land Management, National Science and Technology Center, Denver, CO. 136 p.

Perelman, S.B., S.E. Burkart, M. Oyarzabal, C. Bagnato and W. Batista. 2017. Climatic and land-use drivers along a latitudinal gradient: species diversity in temperate grasslands on agricultural soils. Journal of Vegetation Science 28:1230-1239.

Peterson, G., C.R. Allen, and C.S. Holling. 1998. Ecological resilience, biodiversity, and scale. Ecosystems 1:6-18.

Pohl, M., D. Alig, C. Korner, and C. Rixen. 2009. Higher plant diversity enhances soil stability in disturbed alpine ecosystems. Plant Soil 324:91-102.

Quijas, S., B. Schmid and P. Balvanera. 2010. Plant diversity enhances provision of ecosystem services: a new synthesis. Basic and Applied Ecology 11:582-593.

R Core Team. 2018. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: https://www.R-project.org/

Roath, L.R. and W.C. Krueger. 1982. Cattle grazing behavior on a forested range. Journal of Range Management 35(3):332-338.

Rumbolt, K.S., F. Blouin, B.A. Downey, B.L. Downey, C.A. Koenig, D.J. Jarina, P.F. Jones, J.P. Landry-DeBoer and E.R. Wesley. 2011. MULTISAR: A multi-species conservation strategy for species at risk, 2010-2011 Report. Alberta Species at Risk Report No. 141, Alberta Sustainable Resource Development, Fish and Wildlife Division, Edmonton, Alberta. 84 p.

Sasaki, T., S. Okubo, T. Okayasl, U. Jamsran, T. Ohkuro, and K. Takeuchi. 2009. Management applicability of the intermediate disturbance hypothesis across Mongolian rangeland ecosystems. Ecological Applications 19:423-432.

Schieck, J., P. Solymos and D. Huggard. 2014. Human Footprint in Alberta. ABMI Science Letters, Alberta Biodiversity Monitoring Institute, Edmonton Alberta. 6 p.

Senft, R.L., M. B. Coughenour, D.W. Bailey, L R. Rittenhouse, O. E. Sala and D.M. Swift. 1987. Large herbivore foraging and ecological hierarchies. BioScience 37(11):189-795.

Sims, P.L. and J.S. Singh. 1978. The structure and function of ten western North American grasslands: III. Net primary production, turnover and efficiencies of energy capture and water use. Journal of Ecology 66:573-597.

Sliwinski, M.S. and N. Koper. 2015. Managing mixed-grass prairies for songbirds using variable cattle stocking rates. Rangeland Ecology and Management 68(6):470-475.

Symstad, A.J. and J.L. Jonas. 2011. Incorporating biodiversity into rangeland health: plant species richness and diversity in Great Plains grasslands. Rangeland Ecology and Management 64(6):555-572.

Task Group on Unity in Concepts and Terminology (TGUCT). 1995. New concepts for assessment of rangeland condition. Journal of Range Management 48:271-282.

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

Teague, W.R., and S.L. Dowhower. 2003. Patch dynamics under rotational and continuous grazing management in large, heterogeneous paddocks. Journal of Arid Environments 53:211-229.

Tilman, D. 1999. The ecological consequences of changes in biodiversity: A search for general principles. Ecology 80(5):1455-1474.

Tilman, D., J. Knops, D. Wedin, P. Reich, M. Ritchie and E. Siemann. 1997. The influence of functional diversity and composition on ecosystem processes. Science 277:1300-1302.

Tilman, D., C.L. Lehman, and C.E. Bristow. 1998. Diversity-stability relationships: statistical inevitability or ecological consequence? The American Naturalist 151(3):277-282.

Vavra, M. 2005. Livestock grazing and wildlife: developing compatibilities. Rangeland Ecology and Management 58(2):128-134.

Virk, R. and S.W. Mitchell. 2014. Effect of different grazing intensities on the spatial-temporal variability in above-ground live plant Biomass in North American mixed grasslands. Canadian Journal of Remote Sensing 40(6):423-439.

Weersta, B.G. 1986. Rocky Mountains Forest Reserve: livestock grazing history and *Festuca scabrella* Torr. range recovery. M.Sc. Thesis, University of Calgary, Calgary, Alberta.

West, A.L., C.B. Zou, E. Stebler, S.D. Fuhlendorf, and B Allred. 2016. Pyric-herbivory and hydrological responses in tallgrass Prairie. Rangeland Ecology and Management 69(1):20-27.

West, N.E. 1993. Biodiversity of rangelands. Journal of Range Management 46(1):1-13.

Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management 42(4):266-274.

Whittaker, R.H. 1965. Dominance and Diversity in Land Plant Communities. Science 147(3655):250-260.

Willms, W.D., J.F. Dormaar, B.W. Adams and H.E. Douwes. 2002. Response of the Mixed Prairie to protection from grazing. Journal of Range Management 55:210-216.

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

Willms, W.D., S. Smoliak and A.W. Bailey. 1986. Herbage production following litter removal on Alberta native grasslands. Journal of Range Management 39:536-540.

Willms, W.D., S.M. McGinn, and J.F. Dormaar. 1993. Influence of litter on herbage production in the Mixed Prairie. Journal of Range Management 46:320-324.

Willoughby, M.G., and M.A. Alexander. 2005. Long-term trends in rangeland health of the rough fescue ecological site in the Montane subregion. Pub No. T/069, Alberta Sustainable Resource Development, Public Lands and Forests Division, Edmonton, AB. 20 p.

Wilson, J.B. 1991. Methods for fitting dominance / diversity curves. Journal of Vegetation Science 2:35-46.

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

Appendix A. Number of field assessments completed in the first study, stratified by natural subregion and ecosite (Adams et al. 2005, Adams et al. 2013a, Adams et al. 2013b).

	Upland soil type defined			Location defined		
	Blowout	Loamy	Sandy	Overflow	Saline Lowland	**Total NSR**
Dry Mixed Grass	16	27	9	14	6	72
Mixed Grass		7		1		8
Foothills Fescue		10		2		12
Foothills Parkland		2				2
Central Parkland		6	1	2		9
Northern Fescue	1	13		7		21
Total Ecosites	17	65	10	26	6	**124**

Appendix B. List of ecological sites and their descriptions. Descriptions are from Adams et al. 2013a.

Badlands/Bedrock (BdL): Applies to all inclined to steeply sloping landscapes with greater than 10% bedrock exposures of softrock or hardrock. Slopes generally range from 15% to 60% (in isolated cases 7% to 100%).

Blowouts (BlO): Applies to all SLMs where soils from the Solonetzic order are dominant (>50%) or co-dominant (30 to 50%). Solonetzic soils have an impervious hardpan layer (Bnt horizon) in the subsoil that is caused by excess sodium (Na+). The land surface is frequently characterized by eroded pits.

Clayey (Cy): Applies to all non-saline and non-gleyed Chernozemic soils (soils with A, B and C horizons), and non-saline and non-gleyed Regosolic soils (soils that lack a B horizon >5 cm, and may lack an A horizon) with soil textures in the fine or very fine textural subgroups (i.e. clay and silty clay in >40% clay).

Choppy Sandhills (CS): Applies to all non-saline and non-gleyed Chernozemic soils (soils with A, B and C horizons), and non-saline and non-gleyed Regosolic soils (soils that lack a B horizon >5 cm, and may lack an A horizon) with soil textures in the very coarse (loamy sand) textural subgroup. CS applies to soils that occur on duned landscapes.

Gravel (Gr): Applies to any soil with less than 20 cm of a surface mantle of any textural class over very gravelly or very cobbly (>50% gravel or cobbles) material.

Limy (Li): Applies to all immature or eroded soils with free lime (calcium carbonates) at the soil surface or in the B horizon. Free lime is detected by effervescence when soil is treated with 10% hydrochloric acid (HCl). Limy soils include Rego or Calcareous Chernozemics, eroded phases, and subgroups from the Regosolic order if they are calcareous.

Loamy (Lo): Applies to all non-saline and non-gleyed Chernozemic soils (soils with A, B and C horizons), and non-saline and non-gleyed Regosolic soils (soils that lack a B horizon >5 cm, and may lack an A horizon) with soil textures in the medium and moderately fine textural subgroups (i.e. loam and clay loam).

Overflow (Ov): Applies to non-saline Chernozemic (soils with A, B and C horizons) and/or Regosolic soils (soils that lack a B horizon >5 cm thick, and may lack an A horizon) on landscapes that are low-relief inclines in valley or basinal settings. Overflow sites are usually fan or apron deposits, where upslope streams enter lowland areas and experience a marked decrease in gradient. Slopes generally range from 2% to 9% (in isolated cases from 0.5% to 15%). Overflow occurs only on lower slope positions or adjacent to stream(s), and the percentage of eligible overflow ranges from 10% to 50%.

Sands (Sa): Applies to all non-saline and non-gleyed Chernozemic soils (soils with A, B and C horizons), and non-saline and non-gleyed Regosolic soils (soils that lack a B horizon >5 cm, and

may lack an A horizon) with soil textures in the very coarse (loamy sand) textural subgroup. Sands does not apply to duned landscapes.

Sub-irrigated (Sb): Applies to all Gleyed, non-saline, medium to very coarse textured soils. Gleyed soils occur where the water table occurs near the soil surface, but does not often occur above the soil surface. Gleyed subgroups have faint to distinct mottles within 50 cm, or prominent mottles between 50 and 100 cm.

Saline Lowland (SL): Applies to all salt-enriched soils, including Saline phase Chernozemic, Saline phase Regosolic, and Saline phase Gleysolic soils. Saline phase soils have an electrical conductivity greater than 4.0 dS/m, which retards most plant growth.

Shallow to Gravel (SwG): Applies to any soil with 20 to 50 cm of a surface mantle of any textural class overlying gravelly or very gravelly or cobbly to very cobbly (>20% gravel or cobbles) material.

Sandy (Sy): Applies to all non-saline and non-gleyed Chernozemic soils (soils with A, B and C horizons), and non-saline and non-gleyed Regosolic soils (soils that lack a B horizon >5 cm, and may lack an A horizon) with soil textures in the moderately coarse (sandy loam) textural subgroup.

Thin Breaks (TB): Applies to: 1) all steeply-sloping landscapes with less than 10% bedrock exposures; or largely vegetated areas with bedrock at or near (within 1.0 m of) the surface.

Appendix C. Comparison of the simple average range health score and weighted range health scores across all plant communities sampled within a pasture, plotted against progressive increases in pasture size.
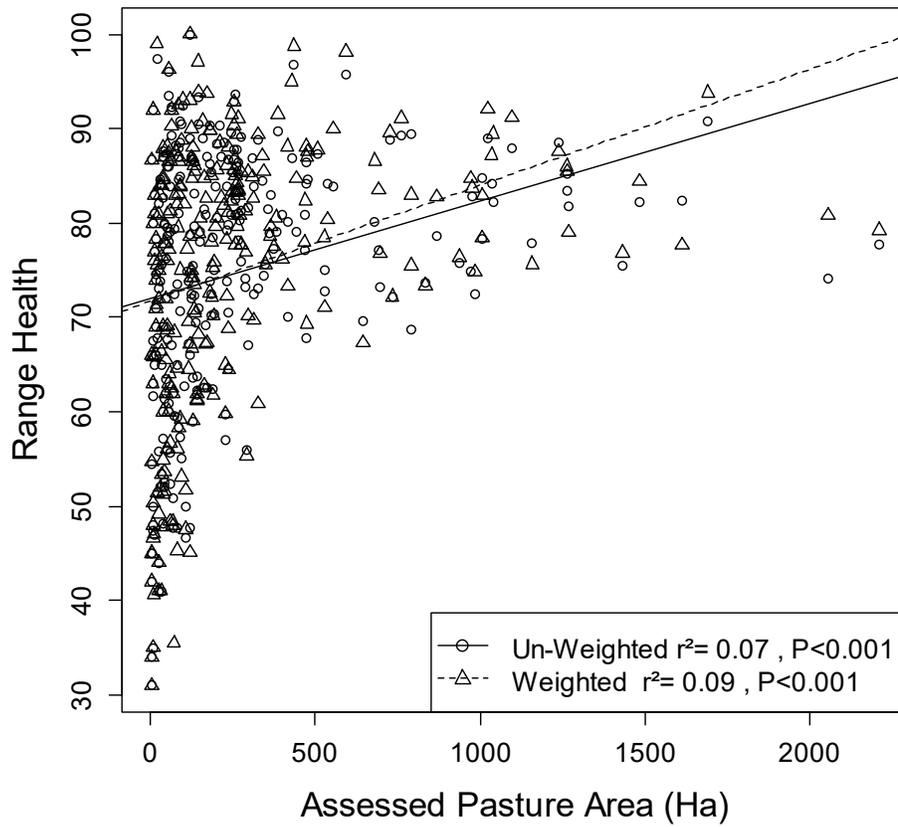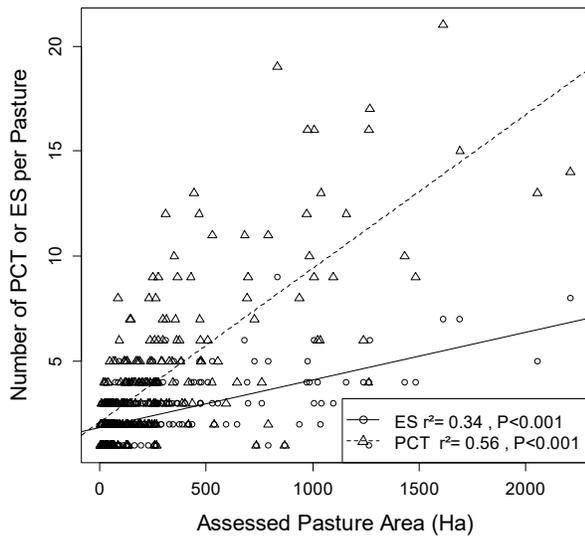
Appendix D. Results of the pair-wise permutational multivariate analysis of variance (PERMANOVA) comparing different ecosites for plant compositional differences. All comparisons indicate significant differences, except those not bolded (P < 0.05).

	BDL	Blo	Cly	CS	Gr	Li	Lo	Ov	Sa	SB	SL	SwG	Sy
Blo	<0.001	-	-	-	-	-	-	-	-	-	-	-	-
Cly	<0.001	<0.001	-	-	-	-	-	-	-	-	-	-	-
CS	<0.001	<0.001	<0.001	-	-	-	-	-	-	-	-	-	-
Gr	<0.001	**0.059**	<0.001	<0.001	-	-	-	-	-	-	-	-	-
Li	0.007	**0.059**	**0.376**	0.003	**0.062**	-	-	-	-	-	-	-	-
Lo	0.001	<0.001	<0.001	0.001	0.003	**0.798**	-	-	-	-	-	-	-
Ov	0.044	<0.001	<0.001	0.001	<0.001	**0.391**	<0.001	-	-	-	-	-	-
Sa	<0.001	<0.001	<0.001	**0.147**	<0.001	**0.063**	<0.001	0.000	-	-	-	-	-
SB	<0.001	<0.001	<0.001	0.001	<0.001	0.003	<0.001	0.016	<0.001	-	-	-	-
SL	0.002	<0.001	<0.001	<0.001	<0.001	0.009	<0.001	<0.001	<0.001	0.002	-	-	-
SwG	0.001	0.039	<0.001	0.001	**0.063**	**0.299**	**0.659**	0.011	**0.076**	<0.001	<0.001	-	-
Sy	<0.001	<0.001	<0.001	<0.001	<0.001	**0.466**	<0.001	<0.001	<0.001	<0.001	<0.001	0.011	-
TB	0.039	<0.001	<0.001	<0.001	<0.001	**0.308**	<0.001	<0.001	<0.001	<0.001	<0.001	0.004	<0.001

Appendix E. The relationship of assessed pasture size (independent variable) and (a) the number of plant community types (PCT) and ecosites (ES) per pasture, (b) the Plant Community Type:Ecosite ratio (PCT:ES) within each pasture, (c) weighted range health (by polygon area) per pasture, and (d) the difference (max vs min) of observed rangeland health scores across each pasture.
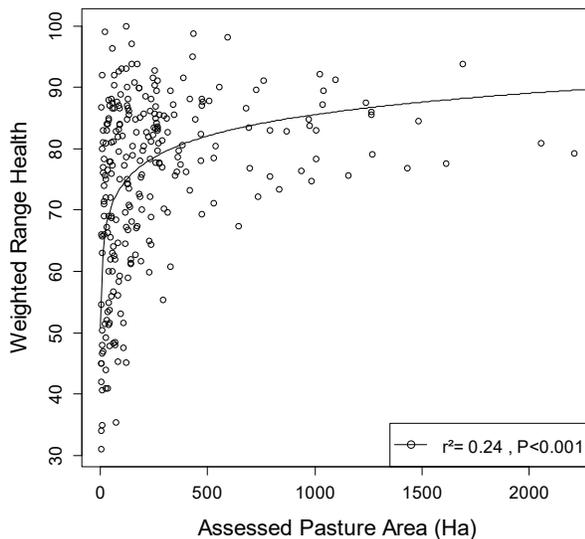
a) Plant community and Ecosite   b) PC:ES ratio
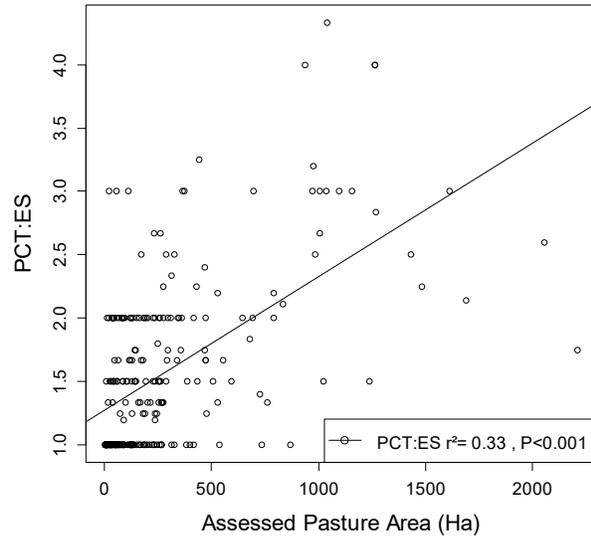
c) Weighted pasture rangeland health   d) Delta pasture health