

VEGETATION RESPONSE TO FIRE AND DOMESTIC
AND NATIVE UNGULATE HERBIVORY IN A
WYOMING BIG SAGEBRUSH ECOSYSTEM

By

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CHAPTER I

INTRODUCTION AND LITERATURE REVIEW

Introduction

The Northern Great Plains mixed-grass prairie has been home to large herds of native grazers for centuries and native vegetation has been exposed to grazing pressure from numerous animals, including bison (*Bison bison*), Rocky Mountain elk (*Cervus elaphus*), white tailed deer (*Odocoileus virginianus*), mule deer (*Odocoileus hemionus*), American pronghorn (*Antilocapra americana*), and black-tailed prairie dogs (*Cynomys ludovicianus*). Until European settlement, Native American tribes hunted and coexisted with these native herds. Reintroduction of horses to the North American continent in the 16th Century (Connelly *et al.* 2011) reached the northern Great Plains in 1715 (Erickson, Lee & Bertram 2000). Around the beginning of the 19th century, Europeans brought with them domesticated livestock which changed the face of the landscape as herd numbers increased (Belsky & Blumenthal 1997; Bork, West & Walker 1998; Valone *et al.* 2002; Sankey *et al.* 2006).

European settlement influenced additional changes to the Northern Great Plains with fire suppression (Cooper 1960; Savage & Swetnam 1990; Peet 2000; Borman 2005; Butler 2006). Prior to European settlement, Native Americans used fire to provide forage for native grazers and domesticated horses, and remove wooded understory (Brown *et al.* 1994; Boyd 2002; Boyd, Hamilton & Running 2006). As European settlement moved farther westward, so did fire

suppression. Settlers actively suppressed fire and, with the decreasing populations of Native Americans, fire decreased across the landscape, leading to woody plant encroachment, and alteration in species composition as fire dependent species decreased and were replaced by fire sensitive species (Miller & Rose 1999; Yanish 2002).

In addition to historic grazing and fire effects, other factors also contribute to changes in rangeland vegetation. Development of current plant communities of the northern Great Plains may largely be a reflection of the Little Ice Age of the 16th to 19th centuries and periodic drought during that time (Boyd, Hamilton & Running 2006; Nordt *et al.* 2008). Recent changes in climate (past 100 years) and future forecasts indicate warmer, drier summers and colder winters which has the potential to further alter the vegetation. Specifically, the potential for wildfire will likely increase and herbivore migrations may be altered (Millar & Woolfenden 1999).

Large predators also disappeared from the landscape as European settlement increased. Trophic cascades show predators influence vegetation composition and structure by altering grazer behavior and movement across the landscape (Fortin *et al.* 2009; Beschta & Ripple 2010). Prey populations have expanded with decreased predation (Berger 1999) leading to increased rates of herbivory on plant communities as herbivores alter foraging habits in the absence of predation (Fortin *et al.* 2009). Predator disappearance may have indirectly influenced fire behavior as decreased pressure from predators on grazing herds altered the distribution of herbivores which altered the distribution of fuel.

The arrival of domesticated livestock with European settlement further altered the landscape of the Northern Great Plains. Diet selection of different grazing species shapes plant communities at a landscape level (Hobbs *et al.* 1996; Belsky & Blumenthal 1997; France, Ganskopp & Boyd 2008). Grazer digestive physiology and ecophysiological developments of grazers allows for the coexistence of multiple grazing species on a landscape with reduced

competition, as described in the Jarman-Bell Principle which states a relationship between animal body size and choice of food supported by diet selection studies (Gwynne & Bell 1968), and theoretical foraging patterns (Belovsky 1997). Studying domestic and native grazers together has the potential to create a broader understanding of their relationship to each other and to the landscape, allowing for more scientifically sound management decisions by land managers.

Evolutionary history of grazing sites has been shown to be a strong variable predicting the importance of vegetation response to grazing treatment (Milchunas & Lauenroth 1993), while exotic plant encroachment has been shown to be primarily influenced by current grazing treatment and plant competition rather than topographic or demographic interactions (Milchunas, Lauenroth & Chapman 1992). Understanding historical landscape level disturbances, such as grazing intensity and fire regimes, is essential to management decisions for landscape heterogeneity and potential vegetation. Managing for heterogeneity includes the combination of biotic and abiotic factors affecting variability in vegetation stature, composition, density, and biomass (Fuhlendorf & Engle 2001) to support a diversity of species with varying habitat and dietary requirements (Du Toit & Cumming 1999; Fryxell et al. 2005; Cromsigt, Prins & Olff 2009). Interactions of spatial and temporal disturbances affect plant community composition and further explorations of these interactions are needed.

To test the influence of combined grazing and fire on vegetation response in sagebrush communities it is important to consider grazing as a complex variable with multiple levels as well as accounting for environmental variables affecting between year changes in data. Pre-existing, long-term grazing exclosures located on the Charles M. Russell, National Wildlife Refuge in central Montana offer a unique opportunity to examine the influence of domestic and native large grazers in combination with fire on herbaceous component of a sagebrush-bunch grass community in the northern Great Plains mixed-grass prairie. Established grazing exclosure studies on similar topographic sites with varying plant compositions allow us to analyze the

impact of site location to the historical grazing pressure across three sites. This analysis is essential to improve our understanding of the interactions of historic, current, and site location interactions affecting plant species composition.

The goal of this study is to determine the level of influence of fire, grazing, and their interactions on vegetation response, plant species composition, and biomass production. The selected long term grazing exclosures provide a unique opportunity to examine the effect of the interactions of fire, and native and domestic grazers, on sagebrush community vegetation. The land use history of the site is representative of much of the grazed landscape across the Northern Great Plains and findings from this study are easily translated to relevant landscape management on private and public lands.

Specific objectives are:

1. Determine vegetation response to fire in areas with: No Grazing, Wildlife Only, and Open Grazing treatments.
2. Determine the effect on biomass production of rabbit and grasshopper exclosures on vegetation response after fire in No Grazing, Wildlife Only, and Open Grazing large ungulate treatments.
3. Quantify grazer attraction to burned versus unburned treatments.
4. Quantify the change in biomass and percent crude protein before and after burning.
5. Determine percent area burned of fires in between and within grazing treatments.

Literature Review

Grassland and shrubland communities developed with and respond to various disturbances and their interactions. Changes in climate and management, expansion of agriculture, and settlement by humans have led to altered fire regimes, altered hydrologic and nutrient cycling, which have ultimately altered ecosystem structure and function on a landscape and community level (Noy-Meir 1995; Griffiths 2002; Butler 2006; Feaser & O'Connell 2009; Metera *et al.* 2010). Herbivory by grazers ranging in size from invertebrates to mega-fauna influence vegetation structure and composition in grasslands and shrublands worldwide (Archibald *et al.* 2005; Branson, Joern & Sword 2006; Jonas & Joern 2007; France, Ganskopp & Boyd 2008; Craine *et al.* 2009; Cromsigt, Prins & Olf 2009). Disturbance by fire is also common on rangelands worldwide and has been historically used by native cultures for food foraging, warfare, and management of game species (Boyd 2002; Griffiths 2002; Borman 2005).

Similar to other systems, fire and grazing disturbances created and maintained grassland and shrubland communities across the North America (Milchunas & Lauenroth 1993; Bond 2005). Before European settlement of the Western United States, 162 million ha of prairie were present in the Great Plains (Samson & Knopf 1994). Loss of grassland in the Great Plains has exceeded 70% with loss in some regions as high as 97%, making grasslands in North America one of the continent's most endangered resources (Samson, Knopf & Ostlie 2004). Sagebrush shrublands also once exceeded 62 million ha in North America, and estimates are that 44% of this has been lost to encroachment by tree species or non-native annual grass and remaining communities are highly fragmented (Davies *et al.* 2011). Further, changes in disturbance regimes have increased expansion of shrubs and trees into native grasslands (Brown & Archer 1989; Heyerdahl, Miller & Parsons 2006). Distribution of grasslands and sagebrush has likely changed due to alterations in fire regimes.

Grassland and shrubland communities across the Western United States have been largely affected by anthropogenic influences since European settlement in the 19th Century. Fire suppression from European settlement, population decrease in Native American communities, and introduction of domestic livestock has altered plant communities on a landscape scale. Native American communities used fire to perpetuate grasslands and provide grazing and foraging habitat for wildlife, but fire suppression following European settlement has altered temporal occurrence of fire events, leading to altered spatial distribution of vegetation communities and wildlife populations (Brown *et al.* 1994; Boyd 2002). European settlers brought with them horses, cattle, and sheep which would reshape the landscape as they competed with native grazers for forage (Boyd 2002). Additionally, introduction of exotic plant species, expansion of settlements, alteration of native plant communities for crop production, and have also led to expansion of shrubland into native grasslands (Gruell 1985; D'Antonio & Vitousek 1992; Lauenroth 1994; Miller, Svejcar & West 1994; Yanish 2002; Turnbull *et al.* 2010; McDonald & McPherson 2011). Native American tribes occupied the area surrounding the Missouri River prior to European westward expansion and settlement, and fire history of the area suggests high occurrence of fire in the area up to the time of European settlement, followed by fire suppression (Brown *et al.* 1994; Boyd 2002).

Topography of the northern Great Plains in Montana was formed by glacial advancement and recession leaving large areas of glacial till and moraine deposits from the Laurentide ice sheet during the late Pleistocene epoch (1.8 million – 10,000 years ago) (Fullerton *et al.* 2004; Davis *et al.* 2006). Glacial advancement and recession altered the course of the Missouri River from a northward flowing river and forced it south through North and South Dakota, forming horizontal strata of the plains sediments and characterizing the Missouri River breaks of central Montana (Barker & Whitman 1989).

Abiotic factors also influence composition, structure, and functioning of ecosystem processes. Soil type is considered constant in an ecosystem due to the time scale on which changes occur on soil. Effects of management on soil properties do not alter the soil type, though degradation of the topsoil and upper horizon layers may occur. Effects of fire and grazing on soil properties is not due to a change in soil characteristics, but the effect of an alteration in the plant community in the form of decreased insulation from plant material which leads to increased fluctuation in soil temperature, increased runoff, increased bulk density, increased water evaporation, decreased soil water, and decreased water infiltration (Teague *et al.* 2010b; Vermeire, Crowder & Wester 2011). Vegetation changes caused by management are often magnified by soil type (Turner 1971; Teague *et al.* 2010b) due to the influence of soil on vegetation community resilience (Turnbull *et al.* 2010).

Differences in vegetation communities on similar soil types have been shown to affect soil response to management practices. Disturbance often promotes erosion which can be in the form of water erosion or aeolian (wind driven) transport. Erosion from both wind and water can be trapped in vegetation and lead to fertile islands throughout dry regions (Schlesinger *et al.* 1990; Field *et al.* 2010). Changes from grassland to shrubland communities have been shown to encourage islands of fertility with positive feedbacks occurring under shrubs and continual degradation between islands (Charley & West 1975; Schlesinger *et al.* 1990; Schlesinger *et al.* 1996; Schlesinger & Pilmanis 1998). This increases runoff, erosion, and areas of bare ground and as flow pathways are more concentrated between plants and nutrients are removed by erosion (Gallardo & Schlesinger 1992; Turnbull *et al.* 2010). Aeolian suspension erosion (dust) has the potential to remove great quantities of nutrients because soil organic matter and nutrients are often associated with smaller soil particles (Field *et al.* 2010). Fire and grazing have been shown to lead to increased rates of sediment fluctuations in both wind and water erosion, with greatest rates present when the two disturbances were combined (Field *et al.* 2011). Decreased vegetation

height has been determined to increase aeolian erosion, while decreased plant basal area increases water erosion and (Field *et al.* 2011). Sagebrush steppe, in particular, has been shown to be highly sensitive to aeolian transport following burning compared to other arid environments (Sankey, Germino & Glenn 2009) because decreased soil moisture has promotes aeolian erosion and soil water decreases following fire (Neuman & Scott 1998; Stout 2001; Vermeire *et al.* 2005).

Prairies of the Great Plains are characterized by variation in species composition and regional community types are influenced by climate (Barker & Whitman 1989). Climate influences vegetation gradients and patterns in ecosystem process in the northern Great Plains and is the most important variable for explaining spatial patterns in ecosystem processes (Barker & Whitman 1989; Bradford *et al.* 2006). Drought and increased precipitation have been shown to influence establishment and persistence of invasive species (Millar & Woolfenden 1999; Davies, Svejcar & Bates 2009), and warming and cooling cycles affect distributions of warm season and cool season grasses (Cordova *et al.* 2011). Temporal and spatial scales of change are also affected by climate and occur across the Great Plains. Temporal changes in length of growing season occur from North to South, and fluctuations in ecosystem's response to long-term climate conditions occur on a spatial scale as a result of vegetation resilience (Bradford *et al.* 2006). Future variation in climate, in combination with management practices, may lead to increased variation in community composition.

Introduction of domestic livestock to the West beginning in the late 19th Century altered vegetation communities from grassland to shrubland (Laycock 1991). Bison historically occupied the Great Plains and Intermountain West (Samson & Knopf 1994) but as cattle and sheep ranching expanded across Montana, bison were replaced by their domestic counterparts. Increased grazing pressure in native grassland generally shifted from light and moderate grazing to heavy grazing with the introduction of livestock (Fleischner 1994) which contributes to alterations in community structure and composition of grassland ecosystems, and promotes

increase of woody species (Arno & Gruell 1983; Zimmerman & Neuenschwander 1984; D'Antonio & Vitousek 1992; Brown *et al.* 1994; Lauenroth 1994; Miller, Svejcar & West 1994; Davis, Grime & Thompson 2000; Heyerdahl, Miller & Parsons 2006; Brooks & Chambers 2011).

Increased grazing pressure also contributed to altered fire regimes and shifted many plant communities from bunch-grass dominated landscapes to those dominated by sagebrush and other woody species not preferred as forage by herbivores (Milchunas & Lauenroth 1993). Grazing alters the effects of fire by reducing fuel loads and removing understory fine fuels which reduces occurrence of fire and alters burn patterns and intensities (Milchunas & Lauenroth 1993; Brown *et al.* 1994; Miller, Svejcar & West 1994; Huber-Sannwald & Pyke 2005; Williams *et al.* 2006; Manier & Hobbs 2007; Leonard, Kirkpatrick & Marsden-Smedley 2010; Mbatha & Ward 2010), though, there is evidence that tree encroachment and cattle grazing do not correlate across decadal temporal scales (Sankey *et al.* 2006). Studies in Australia, South Africa, and the Mediterranean show grazing to be a primary factor in determining fuel loading as fire increases palatability of forage species and attracts grazers (Noy-Meir 1995; Scogings, Trollope & O'Connor 1996; Archibald *et al.* 2005; Williams *et al.* 2006; Leonard, Kirkpatrick & Marsden-Smedley 2010; Mbatha & Ward 2010), but grazing may reduce fire only where the majority of vegetation is palatable (Williams *et al.* 2006). The influence of this interaction – and the subsequent departure from historical disturbance regimes – has led to alterations in vegetation composition and resilience to disturbance.

Fire suppression since European settlement has also altered vegetation communities in the West. Fire across the Northern Great Plains has decreased as European settlement has increased (Brown *et al.* 1994; Heyerdahl, Miller & Parsons 2006). Regional fire history in Montana suggests fires were frequent until the turn of the 20th Century (Arno & Gruell 1983; Brown *et al.* 1994; Heyerdahl, Miller & Parsons 2006) and fire suppression has been suggested to have led to abundant cover of fire sensitive sagebrush (Blaisdell, Murray & McArthur 1982; Arno

& Gruell 1983; Miller, Svejcar & West 1994). Historical photos, fire scar chronologies, and oral histories recount an increase in sagebrush stand density over the past 100 plus years, and fire has been recognized as a major factor shaping the vegetation community change of the northern Great Plains (Frost 2010). Increase of fire sensitive species in fire dependent ecosystems suggests departure from historical fire return intervals as well as influence from changing climate and changes in herbivory patterns.

Fire dependent communities evolved with fire as a disturbance. Most plant species in the Northern Great Plains region are adapted to fire (Augustine & Milchunas 2009). Fire dependent species often reproduce sexually and asexually, responding to fire events and removal of biomass by resprouting from belowground buds (Hajny, Hartnett & Wilson 2011) which are stimulated by increased solar radiation and increased nutrient availability (Hulbert 1969; Old 1969; Schacht & Stubbendieck 1985; Hulbert 1988; Willms, McGinn & Dormaar 1993; Shay, Kunec & Dyck 2001). Other plants are sensitive to fire and may only establish years following fire disturbances at the appropriate successional stage with a seed source nearby (Wright 1974). Shift toward fire sensitive plant communities has the potential to further alter fire regimes (Brooks & Chambers 2011) as fire sensitive species out-compete native grass and forb species and decrease fire return intervals (Miller, Svejcar & West 1994; Miller & Rose 1999; Briggs, Hoch & Johnson 2002; Yanish 2002; Keane *et al.* 2008; Davies *et al.* 2010; Bates, Davies & Sharp 2011). The subsequent result is an increase in concentration of woody species in the absence of fire. Changes in vegetation composition of fire dependent communities will continue to favor fire sensitive species, altering post-fire succession and reducing community resiliency.

Ecological disturbances influence wildlife habitat and population distributions. Fire and herbivory distribute grazers spatially and temporally across the landscape (Cromsigt, Prins & Olff 2009). Competition for forage, though minimal between native and domestic species in central Montana, has been shown to occur in fall between elk and cattle during seasonal die-back of forbs

and also at a time when grass quality is decreased (Mackie 1970; Dusek 1975). Forage quality has been shown to increase following fire (De Witt & Derby 1955; Van Dyke & Darragh 2007; Mbatha & Ward 2010; Allred et al. 2011) which may benefit both wildlife and cattle at a time of year when overall forage quality is decreasing. Sagebrush dependent sage grouse may also benefit from increased landscape heterogeneity from low-intensity burns. Sage grouse require vegetation of varying structure and composition to complete their life cycle (Battazzo 2007; Beck, Connelly & Reese 2009; Hess & Beck 2010) which may be provided by fire in the form of increased landscape heterogeneity, though burning has been shown to have both positive and negative effects on sage grouse habitat with increased forage but decreased sagebrush cover (Hess & Beck 2010) while disturbance patterns have the potential to benefit wildlife habit and promote heterogeneous landscapes, care should be taken manipulating habitat of species of concern.

Plant communities are in a constant state of change. Disturbance, climate variations, and resource availability all have an impact on species diversity, species richness, growth rate, and species ability to establish and persist in a community. Ecological models of plant communities have been developed to explain observed variations in plant communities and in an attempt to predict future change in community composition. Early models explained plant succession advancing to and recessing from a predictable climax community in a linear fashion (Clements 1916), followed by theories of plant community composition determined by chance and conditions at the time of disturbance (Gleason 1927; Whittaker 1953) and rejection of climatic influences and the “monoclimax” and introduction of concepts of trends (Whittaker 1953). Early models are still discussed today, as well as transient maxima theory (Seastedt & Knapp 1993) and intermediate disturbance hypothesis (Connell 1978). More recently, thresholds and steady states have been applied to rangeland management (Holling 1973; May 1977; Westoby, Walker & Noy-Meir 1989; Laycock 1991; Miller, Svejcar & West 1994). State and transition models suggest ecosystems and communities have multiple states in which they exist, based on different

disturbances, and transitions between different states depending on disturbances applied (Westoby, Walker & Noy-Meir 1989), and various stages of succession and recovery or decline from disturbance have been referred to as multiple steady states (Miller, Svejcar & West 1994). The “cup and ball” analogy (Hurd & Wolf 1974) identifies stable states as “troughs” in which a ball or marble is stable, with “hills” over which the marble or ball must be pushed which requires energy to transition communities to other states. An additional model to describe changes in plant communities due to disturbance is the transient maxima theory (Seastedt & Knapp 1993) which attributes short term response to an increase in limiting resources to a point that another resource becomes limiting and response declines. The short term benefits only last as long as both resources are abundant above community needs. Models are very useful for explaining responses, but understanding of the interaction of disturbance and the transitions which promote states of vegetation are necessary to understanding possible management outcomes (Laycock 1991). While no model is perfect and models continue to change, understanding the progression of ecosystem modeling facilitates future model development and improves management decisions.

Individual species respond in varying ways to disturbances of fire and herbivory. At our experimental sites, western wheatgrass, blue grama, Wyoming big sagebrush, yellow sweetclover, and Japanese brome were dominant. Western wheatgrass has been shown to tolerate low to moderate intensity grazing, but decrease under high intensity grazing (Launchbaugh 1967; Olson, Brethour & Launchbaugh 1993; Harmony 2007). Blue grama also tolerates grazing well (Milchunas et al. 1990; Hart & Ashby 1998; Vermeire, Heitschmidt & Haferkamp 2008). Wyoming big sagebrush decreases with fire, is used as winter forage, and generally increases under heavy grazing (Mackie 1970; Arno & Gruell 1983; Laycock 1991; Baker 2006). Exotic species are generally well adapted to climatic and biological conditions at sites they invade. Japanese brome has been shown to decrease under grazing and burning, but persists after years of disturbance and recovers rapidly (D'Antonio & Vitousek 1992; Ogle, Reiners & Gerow 2003;

Harmony 2007). The bi-annual legume yellow sweetclover tolerates grazing (Ogle, St. John & Tilley 2008; Pacanoski 2010), but limited research is available for response to fire. Understanding variations in response between species and within communities improves understanding of site specific responses to disturbance.

Species response to fire and grazing in rangelands influences community response to invasive species, whether exotic or native. Woody plant encroachment into grasslands is increasing under suppressed fire regimes in the West (Lauenroth 1994; Miller, Svejcar & Rose 2000; Heyerdahl, Miller & Parsons 2006). The expansion of juniper into shrub land communities has been documented since the late 1800s (Stewart, Cottam & Butchings 1940; Miller & Wigand 1994; Miller & Rose 1999; Weisberg, Lingua & Pillai 2007) and the causes have been linked to cattle grazing, climate change, and fire suppression (Stewart, Cottam & Butchings 1940; Burkhardt & Tisdale 1976; Miller & Wigand 1994). Historical mean fire return interval in sagebrush communities has been estimated between 12 (Miller & Rose 1999) and 40 years (Houston 1973; Arno & Gruell 1983; Heyerdahl, Miller & Parsons 2006), where as fire suppression over the past 100 years in the northern Great Plains from European expansion and current land management have led to increased tree and shrub cover encroachment on landscapes previously dominated by grassland vegetation, altering vegetation composition on many sites in the Intermountain West (Weaver 1943; Cooper 1960; Belsky & Blumenthal 1997). Maintaining disturbance patterns which promote native plant resiliency and decrease invasibility will reduce both woody plant encroachment and establishment of exotic species.

Studies of species and community response to varying levels of grazing pressure have been studied through the use of grazing exclosures. Exclosure studies increase understanding of plant community recovery rates, as well as allowing for examination of threshold levels in ecosystems to determine if or when a tipping point can be established (Laycock 1991; Lauenroth 1994; Hart 2001). Arguably, grazing exclosure studies are not perfect, they are often small in size

in relation to the surrounding landscape (Lauenroth 1994), have homogeneous grazing treatments applied which are difficult to apply to landscapes with dynamic spatial and temporal variability (Fuhlendorf *et al.* 2009), and most have been grazed prior to exclosure, giving false status to the name “ungrazed” (Fleischner 1994). However, grazing exclosures provide opportunities to study paired treatments and observe recovery following select disturbance in a controlled environment.

Sagebrush communities are also important for rangeland carbon sequestration (Cleary *et al.* 2010). Aboveground biomass is responsible for carbon accumulation and can increase potential carbon losses during fire (MacNeil *et al.* 2008; Cleary *et al.* 2010). Accumulation of carbon in sagebrush landscapes may be due to aboveground shrub growth, while carbon flux rates depend on management treatment. Grazed and burned plots released less carbon than undisturbed plots in the Northern mixed grass prairie (MacNeil *et al.* 2008), and grazing has been shown to be a carbon neutral management practice (Frank *et al.* 1995; Owensby, Ham & Auen 2006; Risch & Frank 2006). Flux rates in management treatments may be related to sequestration of carbon as plants recover following treatment. Further research in carbon sequestration and flux rates on grazed and burned prairie ecosystems is needed to understand potential interactions of grazing and burning on carbon storage and sequestration.

Vegetation composition in rangelands is a result of interactions of disturbance and biogeographical history. Fire, herbivory, climate, historical disturbance patterns, and current land use make understanding current interactions a necessity for land managers. As a result, examining interactions of disturbances at different spatial and temporal scales is necessary to accurately interpret ecological data, including influences of domestic and native herbivory and fire on vegetation response.

CHAPTER II

METHODOLOGY

Study Area

The Charles M. Russell National Wildlife Refuge (CMR) lies on of 445,000 hectares of upland sagebrush, mixed conifer forests, and Missouri river breaks 105 km northeast of Lewistown, MT. Of the 445,155 ha (plus 101,000 ha Ft Peck Reservoir) are managed in combination by the U.S. Army Corps of Engineers and the U.S. Fish and Wildlife Service. State lands comprise 14,425 ha and another 20,234 ha are private in-holdings.

CMR was established in 1936 as the Fort Peck Game Range by President Franklin D. Roosevelt who signed Executive Order 7509 establishing the game range as critical habitat for approximately 400,000 sharptail grouse and 1,500 antelope. The Bureau of Land Management was assigned management of livestock forage produced above and beyond the needs of wildlife, while the United States Fish and Wildlife Service (FWS) was assigned wildlife management responsibility. The multiple use management did not meet the wildlife management requirements set forth by Fish and Wildlife Service, and in 1976, Congress passed Public Law 94-223 giving exclusive management of wildlife ranges – including CMR – to the FWS. This transferred grazing management from the Taylor Grazing Act – which allotted 61.3 percent of the total available AUM's to livestock grazing – to the National Wildlife Refuge System Administration Act of 1966. Transfer of jurisdiction to the FWS also required an Environmental Impact

Statement (EIS) be drafted to serve as a master plan for management of the CMR. The Draft EIS was completed in 1980 following a range survey in 1978. One provision of the proposal was reducing average livestock grazing AUM's on the refuge by 33% from roughly 4 ha/AUM to 6.3ha/AUM.

In 1986 the Record of Decision (ROD) for the Final CMR EIS was signed. The ROD selected the proposed action alternative with implementation beginning in 1987 and full implementation by 1991. Livestock grazing reductions were implemented at a reduction rate of 20% of the previous year, per year, until optimal stocking rates were achieved with final forage allotments of two thirds of the forage to wildlife and one third of the forage to livestock.

Climate is characterized by cold winters and warm, dry summers. Average temperatures range from -12°C in January to +27°C in July with the greatest amounts of precipitation occurring in May and June with average rainfall accumulation of 22 mm to 32 mm (Department of Commerce, 2000). July and August are the driest and warmest months on the CMR when isolated dry-lightening strikes and wildland fires can occur. Above average precipitation was recorded in both 2010 and 2011 at 150% above normal in 2010 and 224% above normal in 2011 (NOAA 2010; NOAA 2011).

Four major soil orders were identified in the 1985 EIS: Entisols, Aridisols, Mollisols, and Vertisols. Mollisols are generally found in areas of higher precipitation and are very limited on the refuge. Aridisols are found on the more gentle slopes of the refuge. Entisols are found on the steep Missouri River breaks and are poorly developed. Vertisols are found on sloping sedimentary uplands and are characterized by high salt or clay, shrink-swell properties, making this soil order poorly suited to agriculture. Water infiltration is slow on Vertisols and roads built on this soil order become impassible with any measurable precipitation (Dailey *et al.* 1985). The 2010 Soil Survey Geographic Database for Phillips Counry, MT identified three major soil types

on CMR (NRCS 2010): Cabba soil series which are loamy, mixed, superactive, calcareous, frigid, shallow Typic Ustorthents; Vida soil series which are fine-loamy, mixed, superactive, frigid Typic Argiustolls; and Zahill soil series which are fine-loamy, mixed, superactive, frigid Typic Calciustepts. These series range from shallow to deep profiles, deposited from glacial till, colluvial, and alluvial deposits on hillslopes, escarpments, and sedimentary plains.

The 1978 Range Survey (Dailey *et al.* 1985) determined five major vegetation types on the refuge including sagebrush-greasewood-grassland type, ponderosa pine-juniper type, grassland, riparian, and cultivated land: Sagebrush-greasewood-grassland type comprised more than 60 percent of the upland areas at the time of the survey, ponderosa pine-juniper type comprised about 35 percent, and grassland, riparian, and cultivated land make up the remaining 5 percent.

The potential native vegetation is described as little bluestem (*Schizachyrium scoparium*), western wheatgrass (*Pascopyrum smithii*), needle-and-thread (*Stipa comata*), prairie sandreed (*Calamovilfa longifolia*), bluebunch wheatgrass (*Pseudoroegneria spicata*), green needlegrass (*Stipa viridula*), plains muhly (*Muhlenbergia cuspidate*), and blue grama (*Bouteloua gracilis*) (NRCS 2010). Current vegetation is dominated by Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*), western wheatgrass, bluebunch wheatgrass, blue grama, fringed sagebrush (*Artemisia frigida*), and Rocky Mountain and creeping juniper (*Juniperus* spp).

Animal species that utilize the rangelands on the CMR include Rocky Mountain elk, American pronghorn, white-tailed deer, mule deer bighorn sheep (*Ovis canadensis*), sharp-tailed grouse (*Tympanuchus phasianellus*), Greater sage-grouse (*Centrocercus urophasianus*), mountain lion (*Puma concolor*), black-tailed prairie dogs, and black-footed ferrets (*Mustela nigripes*).

This study focused on three pre-established grazing exclosures located north of the Missouri River and west of UL Bend in the Missouri River breaks on the CMR refuge.

Vegetation and topography in the breaks ranges from steep, barren dolomite clay hillsides to Ponderosa pine and juniper dominated coulees with sagebrush/mixed-prairie uplands. All three exclosures are located in upland Wyoming big sagebrush/mixed-grass prairie classified vegetation but the topography and vegetation cover varies between the exclosures. Two of the three exclosures – Agate Ridge (6 ha) and Opuntia (3 ha) exclosures – were built in 1967 to “determine the response of vegetation (density and composition) ... in the Nichols Coulee rest Rotational pastures” using vegetation, pellet, and browse transects (CMR Records). The third exclosure – Bell Ridge (3 ha) – was built in 1984 as a cattle exclosure, and converted into a split wildlife and cattle exclosure in 2005 to mimic the exclosures built in 1967. Each exclosure is divided into three parts. One section is fenced with 3.3 m tall hog wire with t-posts and wooden posts at corner braces to exclude native ungulate grazers and domestic cattle, the second section is fenced with four strands of barbed wire with t-posts and wooden corner posts, and the final section is unfenced as a control.

Agate Ridge is located on a narrow ridge-top and is the largest exclosure. Dominant vegetation cover is Wyoming big sagebrush and bunch-grass species with deep juniper coulees tracking down to two creek bottoms to the northeast and southeast which converge to the east in a riparian area. The topography within the fenced exclosure is steep with barren dolomite clay slopes, dense greasewood (*Sarcobatus vermiculatus*) stands, and juniper communities. The upland vegetation consists of sagebrush, plains pricklypear (*Opuntia polyacantha*), Gardner’s saltbush (*Atriplex gardneri*), winterfat (*Krascheninnikovia lanata*), western wheatgrass and dense clubmoss covering with pebbly desert pavement. Sideslope vegetation consists of bunchgrasses and forbs with prairie mugwort (*Artemisia ludoviciana*), Western snowberry (*Symphoricarpos occidentalis*), currant (*Ribes* spp.), and skunkbrush sumac (*Rhus aromatica*) present in coulees.

Bell Ridge exclosure is also dominated by Wyoming big sagebrush and bunch grass, but the topographic features are gently sloping with sparse juniper and few Ponderosa pine (*Pinus*

ponderosa) trees. The vegetation is visibly different between the grazing treatments at Bell Ridge, with grass dominating the enclosure in which only wildlife graze, and Wyoming big sagebrush dominating the control area. Prairie rose (*Rosa arkansana*), purple prairie clover (*Dalea purpurea*), plains prickly pear, yellow sweetclover (*Melilotus officinalis*), and western wheatgrass grow between abundant sagebrush plants.

Opuntia enclosure is aptly named as it has a high occurrence of plains prickly pear and is dominated by Japanese brome (*Bromus japonicus*) and western wheatgrass. The topography at the site is primarily flat with low draws with higher forb abundance and few Wyoming big sagebrush plants. Greasewood, saltbush, prairie mugwort, and snowberry are present in low abundance. All three enclosure sites are classified into the same vegetation class of sagebrush/mixed-grass prairie, but the vegetation composition at each site varies. Distance between sites ranges from 11 km between the two closest sites, to 37 km between the two farthest sites (Appendix A).

Methods

We established a two by three factorial arrangement of treatments arranged in a split plot in a randomized complete block design experiment. Each of three sampling locations served as a block for burning treatment with grazing treatments as subplots. Two fire treatments were burned and unburned. Three grazing treatments were defined by grazer type: Open Grazing (grazed by cattle and native ungulates), Wildlife Only (grazed by native ungulates only), and No Grazing (neither grazed by cattle nor native ungulates).

Burning was conducted on August 3 and 4, 2010. Burn conditions included humidity ranges from 31% - 60% relative humidity, temperature ranges from 68 °F – 87 °F, and winds 0 – 10 mph and variable. Fire rate of spread ranged from 0.5 – 257.2 chains/hour. Measures of

fireline intensity (Btu/ft/s) ranged from 1(Btu/ft/s) – 7577 (Btu/ft/s), and flame length (ft) ranged from 0.4 ft – 27.4 ft. Fire behavior variability between exclosure sites was analyzed by BehavePlus 4.0.0 for four fuel models with corresponding site and weather conditions. Fuel model 1 (short grass) occurred on 0% – 30% slopes with 6% 1-hour fuel moisture. Fuel model 4 (Chaparral) occurred on 0% - 30% slopes with 6% 1-hour fuel moisture, 10% 10-hr fuel moisture, 12% 100 –hr fuel moisture, and 95% live woody moisture. Fuel model gr1 (short, sparse, dry climate grass) occurred on a 5% slope with 9% 1-hour fuel moisture, and 70% live herbaceous moisture. Fuel model gr4 (moderate load, dry climate grass) occurred on a 5% slope with 11% 1-hour fuel moisture and 85% live herbaceous moisture.

Crude protein content of vegetation was sampled to quantify change in nutritional quality as a function of time since fire. Vegetation sampling for crude protein content occurred twice in 2010 (June and September) and twice in 2011 (June and August). Three 20cm X 50cm frames per treatment combination were randomly located and clipped to ground level for each sampling period and values averaged. The clipped vegetation was weighed wet, dried for 24 hours in a drying oven, weighed again, and mailed to the Oklahoma State University campus in Stillwater, Oklahoma, USA to the Soil, Water, and Forage Analytical Laboratory to test for crude protein content. Cage biomass and crude protein data were analyzed using the SAS/GLIMMIX ® procedure for linear mixed models. A Gaussian distribution with an identity link function was used to describe the data.

An herbivore exclusion experiment was designed and biomass was sampled to determine if main effects or interactions occurred between fire treatments, grazing treatment, time, and cage effect. Cage treatments were grasshopper excluded, rabbit excluded, and no treatment. Eight sets of cages (“set” consisting of one grasshopper, one rabbit, and one control) cages were established at each grazing exclosure site: one set in each Burned/No Grazing, Burned/Wildlife Only, Unburned/No Grazing, and Unburned/Wildlife Only exclosures for a subtotal of four sets, and

two sets at each grazing exclosure site in Burned/Open Grazing and Unburned/Open Grazing treatments accounting for the remaining four sets of cages.

Grasshopper exclosures were made from 1m² frames, 61 cm tall, made of 19 mm diameter PVC pipe covered in plastic mesh and secured with eight spikes to the ground. Rabbit exclosures were made from 1.5 m tall welded wire, bent to create 1m² exclosures. Each pair of rabbit and grasshopper exclosures was set side by side with a third 1m², uncaged control plot. Cages in unburned treatments were established in late June 2010 at all sites and cages in burned treatments were established the day following burn completion on August 5, 2010. In 2011 all cages were established in late June on similar vegetation cover as 2010, and all vegetation within the exclosures and control plots was clipped to ground level within a 1m² area and weighed and dried for an estimation of grams of biomass/m². Cage means were computed at each site for each of the treatment combinations and were analyzed for the effects of year, site, grazing treatment, and burn treatment and associated interactions.

Continuing the herbivore exclusion experiment, ten random points were distributed in each of the large ungulate exclosure grazing/burning treatments at each of the three exclosure sites. ArcMAP® 9.3 was used to randomly distribute ten points per treatment for a total of 60 points per site and 180 total sampling points across all exclosures. Points were uploaded into a hand-held Garmin® GPS unit to place and locate points in the field where one 61 cm rebar post was used to mark each point. At each point, a Daubenmire cover class frame (20 cm X 50 cm) (Daubenmire 1959) was sampled in each of eight compass directions (N, NE, E, etc.) placed 2 m from the random point. Data were collected to include percent cover of species composition, percent bare ground, percent litter, percent moss and crust, total grass cover, total forb cover, height (cm) of tallest plant and species of tallest plant. Data were collected in late June 2010 and late June 2011 (data collection periods one and three) as two repeated measures with points as subsamples within subplots, and Daubenmire frames as sub-subsamples within points.

A five meter buffer was created in ArcMAP® to ensure proper space between points and eliminate overlapping. Additional data were collected in a five meter circle at each point included percent cover of shrubs by species, percent area burned at each point, and presence of herbivores. Percent area burned was recorded as a ground cover variable in September, 2010 – six weeks following burning – and was analyzed as a cover class variable to determine percent area burned as a function of grazing treatment. Four photos were taken at each point (north, south, east, and west) for photo records. Data were collected in late June and September of 2010 and June and August of 2011 (data collection periods one, two, three, and four) as four repeated measures with points as subsamples within subplot.

Presence of herbivores was documented at each random sample point to determine attraction to sites before and after burning. Within the 5 meter radius from each point presence of grazers was marked by noting presence of grazers in the area, grasshopper presence, hoof prints, fecal pats, elk and grouse bed sites, and grazed plant material. Hoof prints were identified when possible, as were fecal pats including hare and grouse, and sage grouse turpene excretions. Grazed plant material was noted, including grazing from invertebrates. Data collection following burning also included a count of number of sagebrush seedlings and percent area burned as a cover class variable. Coverage data were analyzed using generalized linear mixed models (SAS/GLIMMIX®) with repeated measures. Due to the unequal spacing of sampling times, an unstructured covariance method was used. A beta distribution with a logit link function was used to describe the cover data.^{1,2}

¹ The data analysis for this paper was generated using SAS software. Copyright, SAS Institute Inc. SAS and all other SAS Institute Inc. product or service names are registered trademarks or trademarks of SAS Institute Inc., Cary, NC, USA.

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CHAPTER III

RESULTS

The effects of large ungulate herbivory and fire are difficult to isolate because the presence of different herbivores directly affected percent area burned across all sites and the burns altered herbivore distributions. Percent area burned exhibited a significant grazing treatment main effect ($p = 0.0255$), where mean area burned was highest in the No Grazing treatment, intermediate in the Wildlife Only grazing treatment, and lowest in the Open Grazing treatment (Fig. 1, Fig. 10). Observational evidence of animal trampling and fecal pats suggested that several herbivores preferred to forage in recently burned treatments over unburned treatments (Table 1).

Forage Quality and Biomass

Mean crude protein (CP) increased following burn treatment from below 10% CP to around 20% CP immediately following burning. Crude protein and biomass exhibited a significant interaction between time and fire treatment ($p \leq 0.0008$). Crude protein increased significantly following fire in late summer 2010, but returned to pre-burn levels by early summer 2011 (Fig. 2). Similarly, biomass exhibited a significant decrease immediately following burning in late summer 2010, but recovered to pre-burn levels by early summer 2011 and was no longer

statistically different from unburned treatments (Fig. 3). There was significant variation across our sampling times but the dominant effect was the increase in crude protein and decrease in biomass when sampled six weeks following fire. Prior to sampling and one year post sampling there were no significant differences among treatments. A nonlinear relationship between forage quality and forage quantity ($r^2 = 0.49$, Figure 4) existed where high forage quality (over 12% crude protein) only occurred on very low biomass measurements.

Our herbivore exclusion experiment resulted in different end of season standing biomass measures. Grasshopper cage biomass was higher than rabbit or control cages in all large ungulate grazing treatments ($p \leq 0.0317$). There were no significant differences between control and rabbit cages ($p > 0.05$), and biomass averaged across cages within large ungulate grazing exclosures increased between 2010 and 2011 in No Grazing/Burned treatment and was not significantly different from Unburned treatments one growing season following fire ($p > 0.05$, Fig. 3). The No Grazing/Burned treatment recovered more rapidly to a level no longer significantly different from the Unburned treatments.

Effects of Fire and Grazing on Composition

Vegetation composition is highly variable in space and time and influenced by grazing, fire and their interaction. While composition is variable over time and difficult to isolate from grazing, the dominant effects of our treatments were from fire. This significance was explained statistically by a significant fire by time interaction for litter, bare ground, crust, and total grass ($p \leq 0.0162$). In the year following fire, litter, crust, and total grass cover were significantly lower, and bare ground was significantly higher than pre-burn data collection ($p \leq 0.0079$). Grazing treatment had no significant effect on litter, total grass, or bare ground ($p > 0.05$). Additionally, a significant collection time by grazing treatment interaction was detected for crust ($p = 0.0200$).

Crust decreased significantly following fire in No Grazing and Wildlife Only grazing treatments that were burned ($p \leq 0.0101$).

In general, native herbaceous species and total grass were most responsive to fire treatments though the effect was largely significant for the time – fire interaction. Total grass cover from year one to year two decreased in the burn treatments ($p = 0.0012$) regardless of grazing treatment. An interaction of burn treatment by year was observed for total grass, western wheatgrass, and Sandburg's bluegrass ($p < 0.0476$). Western wheatgrass exhibited an increase in cover in the unburned treatment across sampling years ($p = 0.0031$) and did not change significantly in the burned treatment ($p > 0.05$). Sandburg's bluegrass cover increased significantly over time in both burned and unburned treatments ($p < 0.0001$), but a significantly greater increase was observed in areas that were burned ($p = 0.0476$). Blue grama was unaffected by burning treatment, but was significantly more abundant in the Open Grazing treatments ($p = 0.0419$) compared to No Grazing and Wildlife Only grazing treatments. No effects were significant for bluebunch wheatgrass or needle-and-thread ($p > 0.05$). Forbs were highly variable across replications and largely unresponsive to our treatments. Total forb cover showed no significant main or interaction effects ($p > 0.05$). Textile onion showed a significant interaction of grazing treatment and fire ($p=0.0190$) and unburned treatment had significantly higher cover in the Open Grazing treatment ($p = 0.0421$). All other native forb species were either of limited abundance for statistical analysis or not significant across treatments.

Dominant exotic species were similar to native species in that they exhibited a decrease in total cover the year following burning in the burned treatments ($p \leq 0.0391$). Japanese brome exhibited an interaction of fire and year ($p = 0.0391$) through a decrease in mean cover in the burned treatments and an increase in mean cover in the unburned treatments between year one and year two. Japanese brome decreased in 2011 except in No Grazing treatment where no change was detected. Yellow sweetclover decreased from collection one to collection three ($p =$

0.0013) with a significant decrease in mean cover in the burned treatment ($p = 0.0019$). Grazing decreased yellow sweetclover in burned treatments following fire, with most abundant cover in 2011 in No Grazing/Burned treatment (Table 3). Overall, fire led to a short term decrease in both exotic species that are often considered invasive.

Cover of shrub species and total shrubs had a complex response to grazing, fire, time, and their interactions. Total shrub cover decreased from year one to year two across plots that were burned, most shrub species recovered to varying degrees in grazing and burning treatments in both the Daubenmire frame collection method and five meter diameter circle. Both methods showed an interaction of year by fire ($p \leq 0.0048$) with a significant decrease in cover only in burned treatments ($p \leq 0.0142$) for Wyoming big sagebrush, and no significant recovery of percent cover in early or late 2011 sampling periods for either data collection method ($p > 0.05$). Fringed sagebrush had a significant year by fire interaction evident in the Daubenmire frame data collection method ($p = 0.0019$), and a year by grazing treatment effect exhibited by a decrease in cover in both No Grazing and Wildlife Only grazing treatments in year two ($p \leq 0.0095$).

The five meter circle cover showed a decrease in average total shrub cover and big sagebrush cover in the unburned treatment over the winter as indicated by the differences between collection two and collection three ($p \leq 0.0487$). Gardner's saltbush exhibited a grazing treatment main effect with No Grazing having a significantly higher percent cover than Open Grazing or Wildlife Only grazing treatments ($p \leq 0.0083$). There were no significant effects of any treatment on prickly pear ($p > 0.05$).

Wyoming big sagebrush seedlings were counted in 2011 to determine the potential recovery of this species following fire. At Agate Ridge enclosure, five seedlings were present within one five meter circle in early 2011 in the Wildlife Only grazing treatment in a burned area previously dominated by dense juniper, increasing to eight seedlings in late 2011. Also at Agate

Ridge, in the burned, Open Grazing treatment, five seedlings were present in 2011 an area also previously dominated by juniper. Each seedling was approximately six inches tall, and no other seedlings were observed except in those two areas which underwent intense fire activity and at which location no big sagebrush previously existed.

Species Richness

Species richness was unaffected by burning or grazing treatment. Analysis of the Daubenmire frame data collection method exhibited change in species richness only affected by time for total species richness, grass species richness, forb richness, and total herbaceous richness ($p \leq 0.0142$) from year one to year two. No effects were significant for total shrub species richness from Daubenmire frame data collection method ($p > 0.05$), but analysis of five meter shrub cover data collection method exhibited an effect by time ($p = 0.0273$).

Tallest Species

Composition of species of dominant height shifted from grass, shrub, and forb species pre burn in June 2010 to primarily grass species post burn in June 2011. Ranking of species composition of five most abundant species in year one included: western wheatgrass, yellow sweetclover, needle-and-thread, Japanese brome, fringed sagebrush, and big sagebrush. Ranking of species composition of five most abundant species in year two included: western wheatgrass, Sandberg's bluegrass, needle-and-thread, Japanese brome, prairie junegrass, and big sagebrush. Only time was statistically significant with an increase in mean height from year one to year two ($p = 0.0028$).

CHAPTER IV

DISCUSSION AND CONCLUSIONS

Discussion

Vegetation of the northern Great Plains developed with fire and grazing as disturbances. Recently, much attention has been given to the unique interaction of fire and grazing as a dominant process that is prevalent throughout the world (Noy-Meir 1995; Vermeire *et al.* 2004; Archibald *et al.* 2005; Williams *et al.* 2006; Kerby, Fuhlendorf & Engle 2007; Fuhlendorf *et al.* 2009; Leonard, Kirkpatrick & Marsden-Smedley 2010; Mbatha & Ward 2010). Both grazing and fire disturbance have the ability to change plant communities on a patch and landscape level, and terms like “pyric herbivory” (Fuhlendorf *et al.* 2009) and fire as a “global herbivore” (Bond & Keeley 2005) illustrate the diversity of thought and research coupling these disturbances as one process. The results from our study demonstrate that the complex fire-grazing interaction is a dominant process in influencing ecosystems through the complex feedbacks where fire and grazing not only have direct effects but also produce critical feedbacks by influencing other disturbances.

Research on grazing effects on rangelands are abundant and largely focus on vegetation or ecosystem level responses that are independent of fire (Milchunas & Lauenroth 1993). An important aspect that is often overlooked is the influence grazing has on the effect and behavior

of wild and prescribed fires. One significant result of our study was the influence of grazing on fine fuel loading, fuel continuity, and the resulting fire behavior. A significant difference in area burned within the fire treatment existed between grazing treatments with 88% of No Grazing treatment burned, 75% of Wildlife Only treatment burned, and 55% of Open Grazing treatment burned (Fig. 1, Fig. 10). Fine fuel loading of grass and litter determines fire behavior in shrub land and grassland ecosystems where higher fine fuel loading burns the landscape more continuously while lesser fuel loads create discontinuous burn patterns across the landscape (Belsky & Blumenthal 1997; Davies, Svejcar & Bates 2009; Twidwell *et al.* 2009). Grazing impacts in forested ecosystems have been shown to promote stand-replacing fire by removing herbaceous material –the dominant competitor of tree seedlings – which promotes dense, even-aged understory communities which act as ladder fuels to the canopy (Zimmerman & Neuenschwander 1984). Alternatively, moderate livestock grazing in sagebrush communities has been shown to decrease fine fuel loading, fuel continuity, and heterogeneity of fuel, decreasing potential for catastrophic wildfire in sagebrush-bunchgrass communities (Archer, Schimel & Holland 1995; Derner & Whitman 2009; Davies *et al.* 2010).

Fine fuel removal by grazing can act as fuel management in shrublands and rangelands by decreasing the likelihood that fire will move throughout a landscape. Fuel heterogeneity increases variability of fire (Bond 2005; Bond & Keeley 2005) and the likelihood that some areas will remain unburned. Livestock grazing to remove fuel can be compared to fuels reduction in forests to reduce extent and severity of wildfires, but rangelands are rarely focused on for fuels management research (Davies *et al.* 2010). Results from our study indicated that grazing increases heterogeneity of burned treatments and variability of fire (Fig. 5). Significant differences in percent area burned occurred across all grazing treatments as grazing pressure increased. This suggests managed livestock grazing may be utilized as a fuels reduction agent to maximize landscape heterogeneity and to promote low intensity burns.

In rangeland and shrublands, risks are associated with catastrophic wildfire just as in forests. Risk of invasive colonization increases after stand-replacing fires in sagebrush communities, as seen in the Great Basin and invasion of sagebrush communities by cheatgrass (*Bromus tectorum*) (Condon, Weisberg & Chambers 2011; Davies *et al.* 2011), and Southwestern deserts by buffleggrass (*Pennisetum ciliare*) (McDonald & McPherson 2011b). Increased fire frequency and severity in shrublands can lead to propagation of the “grass/fire cycle” (D'Antonio & Vitousek 1992; Brooks & Chambers 2011; McDonald & McPherson 2011a) which increases the preponderance of fire in native shrublands as exotics increase fine fuels and increase frequency and size of fires from the previous fire regime. Generally, exotic species with adaptations differing from native species in resource use, resource acquisition, or phenology are likely to impact the ecosystem (D'Antonio & Vitousek 1992; Miller, Svejcar & West 1994). Dominant exotics in our study included Japanese brome and yellow sweetclover that are considered species of concern throughout North America. In our study, both exotic species decreased in response to fire.

Japanese brome has the potential to become invasive in rangelands (Ogle, Reiners & Gerow 2003; USDA-NRCS 2002). Japanese brome contributes to dense litter accumulation (Ogle, Reiners & Gerow 2003), and litter promotes germination of new seed (Whisenant & Uresk 1990; Harmoney 2007), suggesting that Japanese brome has the ability to increase its own abundance by generating more above ground litter (D'Antonio & Vitousek 1992) in the absence of disturbance. Early spring growth allows Japanese brome to take advantage of nutrients and soil water earlier than native species, which allows it to spread at an increased rate and often smothering other grasses (Ogle, Reiners & Gerow 2003; USDA-NRCS 2002). Japanese brome has been shown to decrease following fire and grazing, though persist for years under disturbance (Harmoney 2007). In our study, Japanese brome was most abundant in the No Grazing treatments pre-burn and in No Grazing/Unburned treatment post burn, averaging 18% in No

Grazing/Unburned and decreased in the grazed and burned treatments to between 5% and 10% (Table 3, Fig. 8), similar to other studies (Whisenant & Uresk 1990; Harmony 2007). Decrease in cover of Japanese brome following disturbance at our sites suggests that climate may have influenced response following grazing and burning. Climatic trends of Wyoming big sagebrush communities of eastern Montana are more mesic compared to other Wyoming big sagebrush regions and basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) communities where exotic brome species are of concern (Knick, Holmes & Miller 2005).

Yellow sweetclover is native to the Mediterranean and central Eurasia. It was reported in North America in 1664 as an impurity in forage seed, it now has been listed as an invasive in Canada, the US, Japan, Australia, and areas of Southeastern Europe (Pacanoski 2010). Yellow sweetclover was included in a seed mix seeded over much of the CMR refuge following burns prior to 1974, though at the time no change in species composition was detected (Eichhorn & Watts 1974). Yellow sweetclover is well adapted to the climate and soil conditions of central Montana (Ogle, St. John & Tilley 2008), and its symbiotic relationship with *Rhizobium* bacteria enables it to grow in nitrogen-depleted soils (NRCS 2010; Pacanoski 2010) and can positively influence native plant communities by facilitating growth of other species (Van Riper & Larson 2009). However, other studies have suggested that increases in yellow sweetclover led to a decrease in abundance of other species, species richness, and diversity (Dickson *et al.* 2010). As a forage species, sweetclover is used by domestic cattle, elk, and deer. Forage studies in the Missouri River breaks of north central Montana show yellow sweetclover comprises 75% of elk summer diet, 50% of deer summer diet, and up to 40% of cattle summer diet (Mackie 1970; Dusek 1975). In our study, cover of yellow sweetclover was less in grazed treatments, dropping from 8% cover in the Wildlife Only treatment pre-burn to 3% post burn, and from 3% to 1% in Open Grazing Burned treatments (Table 3, Fig. 9) and decreased in the burned treatments with

increased grazing pressure. This suggests that both fire and grazing may be used to control yellow sweetclover.

In rangelands that evolved as fire dependent ecosystems, fire is critical to maintaining community structure and function. Lack of appropriate fire disturbance in these communities leads to encroachment of woody species including sagebrush (Arno & Gruell 1983; Brooks & Chambers 2011) and juniper (Miller, Svejcar & West 1994). Arguably long term change in fire regimes across much of the West has shifted the community structure from grassland dominated prairies to sagebrush and juniper dominated shrublands (Miller & Rose 1999; Yanish 2002; Davies *et al.* 2010; Bates, Davies & Sharp 2011) and from sagebrush shrublands to pine and fir forests (Arno & Gruell 1983; Zimmerman & Neuenschwander 1984; Brown *et al.* 1994; Borman 2005; Heyerdahl, Miller & Parsons 2006). Wyoming big sagebrush was dominant at our sites, suggesting an alternate stable state in which fire has been absent for periods long enough to promote shrub dominance (Westoby, Walker & Noy-Meir 1989). As the only shrub species that did not resprout following fire, this suggests the historical fire return interval was once shorter and burned greater areas with higher frequency (Reid 2011).

Coupling of fire and grazing interactions necessitate discussion of fire effects on grazing behavior. Application of fire to the landscape, either as a natural process or by anthropogenic prescribed burning, influences changes in vegetation structure, composition in the long term, and forage quality and quantity in the short term. In our study, burning increased crude protein content in cool season forage, but the effect disappeared quickly. Studies have shown a 100 day window of protein increase in central Great Plains Tallgrass prairie (primarily warm season grasses) after which the effect begins to diminish (Allred *et al.* 2011). Similar responses were observed in our study in cool season grasses of central Montana (Fig. 2, Fig. 4).

Increased forage quality is one of the drivers that attract grazers to recently burned sites based on herbivore nutrient requirements and digestive physiology (Gwynne & Bell 1968; Jarman 1974; Belovsky 1997; Sensenig, Demment & Laca 2010). As plants reach maturity, proportion of crude protein to biomass decreases (Norton 1982). Fire increases dry matter digestibility as grass in burned areas is of different biological age than grass in unburned areas which makes foraging on young plants more productive for grazers (Mbatha & Ward 2010).

Forage quality and dry matter digestibility have been shown to decrease as the growing season advances (Mbatha & Ward 2010; Allred *et al.* 2011). Plant response to the removal of biomass following fire and stimulation of new growth increases forage quality and may be beneficial in regions where precipitation is often limiting to new plant growth at the end of the growing season. Historical fire regimes in central Montana suggest naturally occurring fires were most common in late summer due to dry lightning ignition of cured fuel (Arno & Gruell 1983; Higgins 1984; Brown *et al.* 1994; Westerling *et al.* 2003) as well as anthropogenic fire activity set by Native Americans (Boyd 2002). In our study, as well as others, fire occurrence at the end of the growing season has the potential to greatly benefit domestic and native grazers at a time of year when forage quality is low and nutrient reserves must be stored for winter (Mbatha & Ward 2010; Allred *et al.* 2011; Vermeire, Crowder & Wester 2011; Waterman & Vermeire 2011). Studies in the both the Northern and Southern Great Plains show livestock gain the majority of yearly weight in the first half of the grazing season extending from April –September (McCollum *et al.* 1999; Waterman & Vermeire 2011), during a time when forage quality is uniformly higher. Crude protein requirements for maintaining and gaining weight in stocker, replacement, lactating, and pregnant cows range from 7% to 11% crude protein (NRC 2000; Paterson, Funston & Cash 2009) while percent CP in our study averaged below 10% at the end of the growing season in the absence of fire, increasing to around 20% in the 100 days immediately following burning but disappeared by early summer 2011. This significant increase in CP following burning in August

would highly benefit nutritional requirements of range cattle and some wildlife and potentially contribute to end of season increased body condition score and reduced feed supplementation.

Though forage quality increased following fire, biomass in burned areas decreases but returned to pre-burn levels by early summer of 2011 in our study. An inverse relationship between forage quality and quantity exist (Fig. 4). The interaction of biomass and percent CP is an indicator of percent CP at our sites and has also been documented at other sites as well (Allred *et al.* 2011). The result of reduced biomass following fire forces grazers to choose between small areas of high quality, low quantity forage and large areas of low quality, high quantity forage. Additionally, recovery of biomass in our large ungulate grazing treatments following burning suggests focused grazing in the Wildlife Only and Open Grazing treatments by large ungulate grazers influenced biomass recovery following burning and demonstrates grazer attraction to burned areas (Fig. 2). Grazers display preference for recently burned areas, even with reduced biomass (Archibald *et al.* 2005; Fuhlendorf *et al.* 2009; Allred *et al.* 2011) and those areas with increased grazer access will experience higher grazing pressure following fire, as shown by lower rates of recovery in Open Grazing and Wildlife Only grazing treatment in our study.

Early studies of grazing effects following fire suggested grazing inhibits recovery of ecosystems following fire (Clarke, Tisdale & Skoglund 1943; Coupland 1973) which led to land management agencies implementing grazing deferment (typically 2-3 years) prior to allowing livestock grazing on rangelands (BLM 2007). More recently, vegetation response following fire has been shown to be a function of environmental factors (Vermeire, Crowder & Wester 2011). Immediate benefits exist to herbivores grazing on recently burned areas and ewes have been shown to benefit from increased forage quality longer in the growing season following summer burns (Waterman & Vermeire 2011). Grazing deferment following fire misses the positive forage quality effects present immediately following fire (Roselle, Seefeldt & Launchbaugh 2010;

Waterman & Vermeire 2011). Concerns regarding over-use of burned areas by livestock can be addressed by using fire as a tool to distribute grazers across the landscape.

In Australia and South Africa, grazing lawns are kept short following fire from increased grazing pressure as grazers are drawn to, and keep returning to, grazing lawns (Archibald *et al.* 2005; Cromsigt, Prins & Olff 2009; Leonard, Kirkpatrick & Marsden-Smedley 2010). We propose, and additional studies support, that grazers require some area of land burned every year to obtain the benefits from increased crude protein levels post burning (Vermeire *et al.* 2004; Archibald *et al.* 2005; Augustine & Milchunas 2009; Fuhlendorf *et al.* 2009; Leonard, Kirkpatrick & Marsden-Smedley 2010; Mbatha & Ward 2010). Patch burning has been shown to create and maintain habitat heterogeneity (Vermeire *et al.* 2004). In our study, the positive effect of increased crude protein disappeared in the time between late summer of year one and early summer of year two. The limited temporal benefits to grazers necessitate immediate utilization following fire to reap the nutritional benefits of increased forage quality. Patch burning creates built-in grazing deferment necessary for grazer distribution and habitat resource quality. In highly fragmented or small management areas, increased habitat and landscape heterogeneity has the ability to support more diverse populations of wildlife and maintain species diversity (Du Toit & Cumming 1999; Fuhlendorf & Engle 2001; Fryxell *et al.* 2005). Based on grazer physiology, habitat requirements, and vegetation response at our sites, our research suggests a shift toward a fire regime with smaller, more frequent fires would be beneficial to domestic and native ungulates in the northern mixed-grass prairie.

Foraging habits of ruminants are based on digestive physiologies which distribute grazers across the landscape based on foraging preferences (Mysterud 1998; Cromsigt, Prins & Olff 2009). Fire events have the potential to alter grazer distribution for management purposes, thereby increasing landscape heterogeneity. Deer and cattle foraging habits in the Missouri River Breaks in north central Montana suggest minimal overlap in habitat usage by deer and cattle

based on digestive physiologies (Dusek 1975; Mysterud 1998), while elk and cattle have been shown to utilize similar habitat types in the fall (Mackie 1970). In addition, sagebrush-bunchgrass upland ridges in the Missouri River breaks where our study sites were located have been determined to be key habitat for mule deer and elk (Mackie 1970). Results from a study of herbivore body mass in relation to distribution of habitat use found ruminants to be more evenly distributed over the landscape with increasing body mass (Cromsigt, Prins & Olff 2009), suggesting cattle and elk on CMR may utilize more diverse landscapes than deer. Based on digestive physiologies, North American grazers likely follow similar distribution patterns as African grazers based on size and diet selection. Burning patches of habitat has the potential to decrease competition between large grazers by promoting landscape mosaics and providing more diverse habitats with varying areas of time since disturbance. Patch burning can also promote overlap of foraging habitats between grazing species, resulting in rest of unburned areas.

Study of vegetation response in grazing exclosures helps to broaden understanding of ecological processes influencing plant communities. Our study included three levels of grazing pressure in an attempt to determine an effect by domestic cattle compared to native wildlife. Fleischner (1994) argued strongly against livestock grazing, citing alterations in ecosystem composition, function, and structure resulting in changes in vegetation stratification, hydrology, pedology, negative impacts on wildlife, and increased introductions of exotic plant species. Grazing treatment studies in the West are fairly abundant (Rice & Westoby 1978; Anderson & Holte 1981; Holechek & Stephenson 1983; Lauenroth 1994; Laycock 1994; Bork, West & Walker 1998; Hart 2001; Holechek *et al.* 2003; Holechek, Galt & Khumalo 2006), though most examine specific groups of organisms or closely related processes species (but see Kay & Bartos 2000; Augustine & McNaughton 2006; Veblen & Young 2010). We were unable to detect significant differences between Open Grazing and Wildlife Only grazing treatments Results from

our grazing exclosures are supported by one additional study excluding wildlife and livestock (Manier & Hobbs 2007).

Individual species and functional groups respond in varying ways to grazing and fire. At our sites the shrub species resprouted, with the exception of Wyoming big sagebrush, and regrowth of grass and forbs returned to pre-burn levels one growing season following fire. Response of individual species depends on evolutionary pressures, as well as site history. Native grasses are adapted to defoliation and rapidly replace photosynthetic tissues and resume growth, whether defoliation occurred from burning or grazing (Ralphs & Banks 2009). Western wheatgrass exhibited no significant difference between grazing treatments, as anticipated from a native bunch grass which evolved under grazing pressure, and was shown to benefit from early season grazing (Olson, Brethour & Launchbaugh 1993; Harms 2007) and increased following burning, decreased with burning and high intensity grazing, and increased with burning and moderate to low intensity grazing (Table 2; Table 3; Launchbaugh 1967; Olson, Brethour & Launchbaugh 1993; Harms 2007). Additional studies have also shown western wheatgrass to respond to grazing exclusion with increased cover in ungrazed areas (Hart 2001; Willms *et al.* 2002); though no grazing effect was present in our study. This is supported by Vermeire *et al.* (2008) who found grazing exclosures produced no effect on western wheatgrass cover one growing season following removal of grazing treatment pressure. Blue grama also exhibited no response to burning treatments, but was significantly more abundant in Open Grazing treatments (Fig. 7). This result is supported by additional studies (Milchunas *et al.* 1990; Hart & Ashby 1998; Willms *et al.* 2002; Vermeire, Heitschmidt & Haferkamp 2008), and is not surprising for a warm season grass that is known to tolerate grazing.

Wyoming big sagebrush exhibited a significant decrease in all burned treatments in our study (Fig. 6, Table 3) but no effect from grazing treatment with no recovery in the growing season following fire, as supported by literature on sagebrush recovery following burning (Cook,

Hershey & Irwin 1994; Miller & Rose 1999; Baker 2006; Bates *et al.* 2009). Sagebrush can become dominant in the absence of fire (Miller, Svejcar & West 1994; Keane *et al.* 2008) and changes in grazing regime slowly alter vegetation composition from grassland to sagebrush dominated sites (Turner 1971; Anderson & Holte 1981; West *et al.* 1984; Seefeldt & McCoy 2003). Considering sagebrush encroachment is a long and slow process, it is not surprising that our sites exhibited no significant effect from grazing treatment following 40 years of grazing treatment. Additionally, grazing exclosures have not generally been shown to reverse sagebrush-dominated sites by excluding grazing (Anderson & Holte 1981; Bork, West & Walker 1998; Manier & Hobbs 2007). Grazing exclosure studies in sagebrush communities have shown little change following grazing exclosure, and this is supported by studies reporting slow rangeland recovery in dry climates (McLean & Tisdale 1972; Smeins, Taylor & Merrill 1976; Rice & Westoby 1978). In addition, condition of range prior to exclosure from grazing would tend to lengthen the time before changes were seen.

An unexpected result was the presence of sagebrush seedlings in burned areas previously dominated by juniper at Agate Ridge exclosure (Fig. 11). Considering the absence of mature plants in those areas prior to burning, and the prevalence of literature citing non-viability of sagebrush seedbanks following fire (West & Yorks 2002; Allen, Chambers & Nowak 2008; Bates *et al.* 2009; Davies *et al.* 2009), the seeds must have arrived on site in the time since burning. Seed may have been dispersed through wind, water, or animal movements to the area. There is an absence of literature on sagebrush seed survival following digestion, but studies have been done on weed, shrub, and grass species processed through the rumen with limited but measurable survival rates (Blackshaw & Rode 1991; Doucette, Wittenberg & McCaughey 2001; Haidar, Gharib & Sleiman 2010). A second dispersal option somewhat unique to this region of Montana would be dispersal through tracking of hoofed species. Bentonite clay soil structure on CMR becomes sticky with any measurable precipitation and forms tacky balls of clay on the soles of

shoes and hooves of ungulates (Fig. 12). This tracking through mud following snow melt in the spring may have transferred sagebrush seedlings into burned areas as herbivores traveled in and out of these areas in search of new forage, highlighting again the interdependent nature of fire and grazing, here, as a possible method for seed dispersal. Climatic conditions during this two year study should not be overlooked as eastern Montana received above average precipitation during the two years of our study, in combination with the more mesic

Response of total shrub cover following fire showed partial recovery of total cover one growing season following fire due to resprouting of all shrub species excluding Wyoming big sagebrush. Alteration of fire regime of the northern Great Plains is illustrated by the presence of sagebrush as the dominant, non-sprouting shrub as part of a community of resprouting shrubs, grasses, and forbs. Sensitivity of sagebrush to fire suggests that levels of cover observed in this study would not have been historically sustainable in an environment with increased fire (Arno & Gruell 1983; Boyd 2002).

Litter accumulation on rangelands influences biological processes by trapping heat and moisture at the soil surface, promoting germination of seed, and decomposing into soil organic matter (Anderson & Holte 1981; Harmoney 2007). Litter and standing dead have been shown to decrease in Northern mixed-grass prairie in the first growing season following fire and recover in subsequent growing seasons (Grant *et al.* 2010). Litter has been shown to accumulate in post-fire communities with rapid regrowth of forb and grass species (Vermeire, Crowder & Wester 2011). Litter also influences plant height, tiller weight, and herbage yield of cool season grasses, which may be influenced by the increased water availability, while decreasing height and cover of blue grama (Willms, McGinn & Dormaar 1993). In the absence of litter, plant shoots benefit from increased solar radiation and photosynthesis, and excessive litter has been shown to limit productivity by reducing photosynthetic rates and establishment of woody seedlings when litter levels exceed than 100 g/m² (Brown & Archer 1989; McCarron & Knapp 2003). Grazing is a

primary remover of prairie biomass in the Northern Great Plains (Milchunas, Lauenroth & Chapman 1992; Lauenroth 1994; Belsky & Blumenthal 1997; Bork, West & Walker 1998; Derner *et al.* 2009) which directly affects litter accumulation. Our research exhibited a decrease in cover of litter, with a corresponding increase in percent bare ground. This decrease in litter and increase in bare ground has been shown to have a positive effect on plant growth as fire removes standing dead biomass which promotes plant growth due to increased solar radiation, and increases the nutrient availability in plants (Hulbert 1969; Old 1969; Schacht & Stubbendieck 1985; Hulbert 1988; Willms, McGinn & Dormaar 1993; Shay, Kunec & Dyck 2001). Litter accumulation also contributes to carbon storage and has been shown to accumulate rapidly following fire and remain relatively constant (Cleary, Pendall & Ewers 2010; Vermeire, Crowder & Wester 2011).

Summer burning of Northern mixed-grass prairie has shown plant community resistance to summer fire (Vermeire, Crowder & Wester 2011) exhibited by similarities in root biomass in burned and unburned sites, and rapid recovery in forb biomass in the year following fire (Milchunas & Lauenroth 1993). Deeply rooted, native bunch grasses have been shown to persist in mature sagebrush stands and recover rapidly following fire which may contribute to resilience to grazing and fire (Bates *et al.* 2009; Cleary, Pendall & Ewers 2010) This supports the observed increase in grass species in our study which responded favorably or showed no response following burning. Persistence of graminoid root structure and below ground biomass also supports rapid increase of above ground biomass following fire and resilience to grazing following spring and summer burning (Bates *et al.* 2009; Cleary, Pendall & Ewers 2010).

Grassland resilience to disturbance may be due to deep rooted bunch grasses capitalizing on nutrient availability following fire (Cleary, Pendall & Ewers 2010). Growing season grazing has been shown to promote species richness, reduce light limitations, and sustain plant productivity and forage quality (Frank & McNaughton 1993). Following burning, exposed soil

absorbs solar radiation which increases soil temperature and stimulates soil microbial activity.

This can lead to increased availability of soil nutrients and enables rapid root growth that can exceed root growth of unburned plants (Ramundo & Seastedt 1990; Limb *et al.* 2011).

Belowground biomass also affects resprouting shrub response following fire. A study in Kansas documented increased post-fire productivity of shrub shoots was greater than at unburned sites (McCarron & Knapp 2003).

Several factors may have contributed to results seen in our study. The two years encompassing this study saw record precipitation in central Montana. For this reason, August burns in 2010 may have resembled spring burns rather than late summer burns with respect to herbaceous and woody moisture, relative humidity, temperature, and curing of fuel. Responses of plants in 2011 may also have benefited from precipitation levels up to 224% above normal, however similar studies report similar vegetation response to ours (Vermeire, Crowder & Wester 2011) in normal precipitation years. In addition, measurable precipitation immediately following burns in August 2010 exhibited plant response under ideal conditions, which are not always present. Furthermore, the Wyoming big sagebrush community in eastern Montana exists in a region more similar to the relatively mesic mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) communities which exist in areas of greater precipitation and are therefore more resilient to disturbance (Knick, Holmes & Miller 2005).

Grasshopper cage biomass may have been influenced by microclimates created within grasshopper grazing exclosures. Cages used in our study were similar to cages used in studies to enclose grasshoppers for predation studies (Belovsky & Slade 1993; Laws *et al.* 2009), though this is the first use of grasshopper cages to identify grasshopper herbivory influence on vegetation biomass. Cages may have created a vegetation microclimate including shading, increased humidity, and decreased day to night temperature fluctuations, though the very nature of excluding grasshoppers precluded monitoring cage effects in the presence of grasshoppers in our

study. At this time, additional research utilizing grasshopper exclosure cages on vegetation biomass is recommended.

This two year study only allowed us to sample one year post fire which could be argued to be too short to establish realistic vegetation response to fire. Our goal, however, was to determine immediate effects of a fire event on vegetation in historical grazing treatments. Our results showing an immediate increase in forage quality and decrease in biomass following fire, with a disappearance in fire effects nine months after, which supports arguments for grazing access to recently burned areas to reap the most nutritional benefits. In addition, lack of change in species richness in burned plots suggests a rapid recovery from burning to establish pre-burn richness grass and forb species, though long-term change in fire regime would most likely change the plant community. Variation in vegetation between exclosures may have contributed to above or below expected response of some species to treatment or over all cover, but variation in vegetation types allowed the results to be applied to a broader range of vegetation types and increase applicability to landscape-level management.

Community dynamics change throughout seasons and across yearly and decadal scales as a result of grazing, vegetation response, fire, climate, and abiotic processes. Many changes are simply due to the advancement of time as ecosystems remain in stable states due to the absence of major disturbances. Influence of time as a main effect, or the interaction by time with each treatment, was present in analysis of every variable measured in our treatments. Accepting time as an ever present effect allowed us to discuss interactions and main effects of fire and grazing, understanding the constant influence of time on our study. Biological processes of growth, reproduction, and senescence continue in time, regardless of disturbance so it is necessary to recognize time and the progression of growth of individual plants and communities as a function of time.

Exclosure design of our study with each fence type removing successively different large grazers has produced some quantifiable change in community structure. However, the large amount of sagebrush still present in the exclosures can be explained by these ecological models. Sagebrush dominated sites represent an alternative state at which ecosystems are stable, and removal of grazing often does not return sites to their pre-grazing condition (Turner 1971; Anderson & Holte 1981; West *et al.* 1984). Data collected at our sites one growing season post fire suggests the impact of fire as an input of energy to the site was enough to shift our sites across a threshold of sagebrush dominated state to one with increased grass and forb cover. Succession of plant communities at our sites will be dependent on soil type, climate, and future management strategies.

Conclusions

Evolution of fire and grazing as one ecological process becomes increasingly evident as more research concludes the coupling of these processes are responsible for much of the vegetation and animal responses observed in this type of research. The influence of herbivory on fuel loading, fire spread, and fire behavior highlight the management implications involving grazing as a tool to manipulate both fire sensitive and fire dependent landscapes. Recovery of ungrazed treatments following fire and slower recovery of grazed treatments suggests that fire studies conducted without exclosures may be confounded by grazing. The application of fire as a grazing management tool and herbivore response to fire benefits vegetation response as well as fulfilling or exceeding nutrition requirements in livestock and wildlife.

Our study supports theories that vegetation responds in similar ways across varying ecosystems. Studies in varying climates across continents exhibit increased nutritive content of vegetation and increased attraction of grazers following fire, and altered fire behavior between

grazed and ungrazed plant communities. Fire has been shown to support landscape heterogeneity and to support increased numbers of herbivore species in South Africa, Australia and North America, and our research supports this.

The shifting focus to the use of rangelands as carbon sinks and recognizing the importance of complex interactions driving ecological processes is shifting management. Effects of anthropogenic activity on productivity and carbon storage necessitate the importance of understanding of complex biotic and abiotic interactions. Past research decoupling grazing and fire on the landscape has provided preliminary data of vegetation response to disturbance, but further research is needed to understand interactions multiple of disturbances on rangeland. This project is one of the first to examine the varied vegetation response to fire with native and domestic ungulates, but further research is needed to fully understand the complex interactions of grazer influence on and attraction to fire.

Management Implications

Grazing preferences of domestic and native ungulates vary greatly, as do differences between native species. Applying fire as a forage enhancement tool on rangelands in central Montana has been shown to increase crude protein levels in late August following late summer burns. Benefits to livestock managers are numerous, the most important of which is livestock diet selection of high quality forage when other late season forage quality is declining. Herbivore attraction to burned patches reduces grazing pressure on unburned vegetation, creating built –in rest periods for rangelands in the absence of fencing. Though this study was not conducted in a drought period, potential management implications for maintaining stocking rates through unfavorable conditions are possible, though more research is needed.

The effect of grazing on fire behavior and burn patterns as a fuel reduction method in rangelands has important implications for large and small scale grazing operations. Understanding the effect of grazing on fire behavior has applications to both public land management entities and private ranches planning to incorporate fire as a management tool. Our study exhibited significant interactions of grazing intensity by herbivores of differing foraging habits and the effect on fire behavior and area burned. By manipulating stocking rates, managers may be able to utilize grazing as a cost effective way to safely manage fuel loads to meet burn objectives and reduce the occurrence of stand replacing fires. The impact of increased grazing pressure in combination with fire in sagebrush may have application to managing habitat of sensitive wildlife species to reduce intensity and continuity of fire and increase patch heterogeneity. By coupling fire and grazing and applying them as a combined management tool, benefits to vegetation and to grazers are greatly enhanced.

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Evidence	Species	2010		2011			
		Late Summer		June		Late Summer	
		Burned	Unburned	Burned	Unburned	Burned	Unburned
Fecal	Deer	6 ± 0.44	5 ± 0.44	20 ± 0.62	13 ± 0.53	21 ± 0.67	8 ± 0.54
	Elk	0	3 ± 0.24	30 ± 0.99	20 ± 0.61	27 ± 0.80	20 ± 0.61
	Cattle	2 ± 0.22	10 ± 0.89	3 ± 0.17	7 ± 0.40	13 ± 0.84	10 ± 0.40
	Lagomorph	2 ± 0.22	2 ± 0.15	11 ± 0.52	4 ± 0.29	6 ± 0.47	4 ± 0.24
Trampling	Deer	34 ± 1.27	28 ± 0.95	26 ± 1.06	29 ± 1.09	40 ± 1.56	39 ± 1.48
	Elk	18 ± 1.00	14 ± 0.96	29 ± 1.23	21 ± 1.12	39 ± 1.65	40 ± 1.65
	Cattle	9 ± 0.67	8 ± 0.77	13 ± 1.00	11 ± 1.10	30 ± 1.64	21 ± 1.45

Table 1 Ungulate usage of burned areas following fire. Descriptive statistics for fecal and trampling evidence of deer, elk, cattle, and lagomorphs usage following burning in late summer 2010, and June and late summer 2011. Burned and Unburned treatments are averaged across all grazing treatments. Burned areas appear to be preferred over unburned areas, though not consistent across all species

Group	Species	2010		
		No Grazing	Wildlife Only	Open Grazing
GRASS	<i>Bouteloua gracilis</i>	4.24 ± 0.57	5.71 ± 0.76	9.22 ± 0.98
	<i>Bromus japonicus</i>	13.44 ± 0.41	8.00 ± 0.54	8.67 ± 0.69
	<i>Pascopyrum smithii</i>	15.21 ± 0.78	13.72 ± 0.83	10.91 ± 0.76
	<i>Stipa viridula</i>	2.87 ± 0.43	5.28 ± 0.61	3.29 ± 0.49
FORB	<i>Melilotus officinalis</i>	8.93 ± 0.80	8.39 ± 0.78	3.41 ± 0.53
SHRUB	<i>Artemisia frigida</i>	4.28 ± 0.48	3.66 ± 0.44	4.24 ± 0.48
	<i>Artemisia tridentata</i>	8.87 ± 0.98	7.37 ± 0.82	9.73 ± 1.00
	<i>Opuntia polyacantha</i>	1.86 ± 0.31	1.50 ± 0.30	1.66 ± 0.28

Table 2 Average percent cover estimates for species of interest in year one (2010) in Daubenmire frame collection method. Burned and unburned treatments are pre-burn data. Grazing treatments are averaged across all enclosure sites.

		2011					
		No Grazing		Wildlife Only		Open Grazing	
		Unburned	Burned	Unburned	Burned	Unburned	Burned
GRASS	<i>Bouteloua gracilis</i>	1.76 ± 0.38	3.13 ± 0.49	3.41 ± 0.57	4.03 ± 0.59	7.64 ± 0.91	6.37 ± 0.73
	<i>Bromus japonicus</i>	18.53 ± 1.39	8.01 ± 0.93	9.58 ± 0.93	3.66 ± 0.53	10.38 ± 1.20	9.42 ± 1.05
	<i>Pascopyrum smithii</i>	13.15 ± 0.79	12.71 ± 0.94	11.34 ± 0.89	11.26 ± 0.90	12.08 ± 0.92	9.38 ± 0.71
	<i>Stipa viridula</i>	2.00 ± 0.36	1.67 ± 0.38	5.65 ± 0.72	3.86 ± 0.68	3.12 ± 0.61	2.81 ± 0.48
FORB	<i>Melilotus officinalis</i>	2.97 ± 0.66	5.75 ± 1.13	2.28 ± 0.46	2.92 ± 0.56	2.15 ± 0.61	0.99 ± 0.26
SHRUB	<i>Artemisia frigida</i>	1.66 ± 0.33	0.47 ± 0.20	2.13 ± 0.37	1.04 ± 0.27	4.51 ± 0.63	2.22 ± 0.51
	<i>Artemisia tridentata</i>	9.02 ± 1.35	0.65 ± 0.42	7.02 ± 1.06	2.81 ± 0.63	10.42 ± 1.32	4.18 ± 0.92
	<i>Opuntia polyacantha</i>	1.51 ± 0.36	0.96 ± 0.31	0.95 ± 0.26	0.64 ± 0.23	1.28 ± 0.32	1.26 ± 0.36

Table 3 Average percent cover estimates for species of interest in year two (2011) in Daubenmire frame collection method. Grazing treatments are averaged across all enclosure sites.

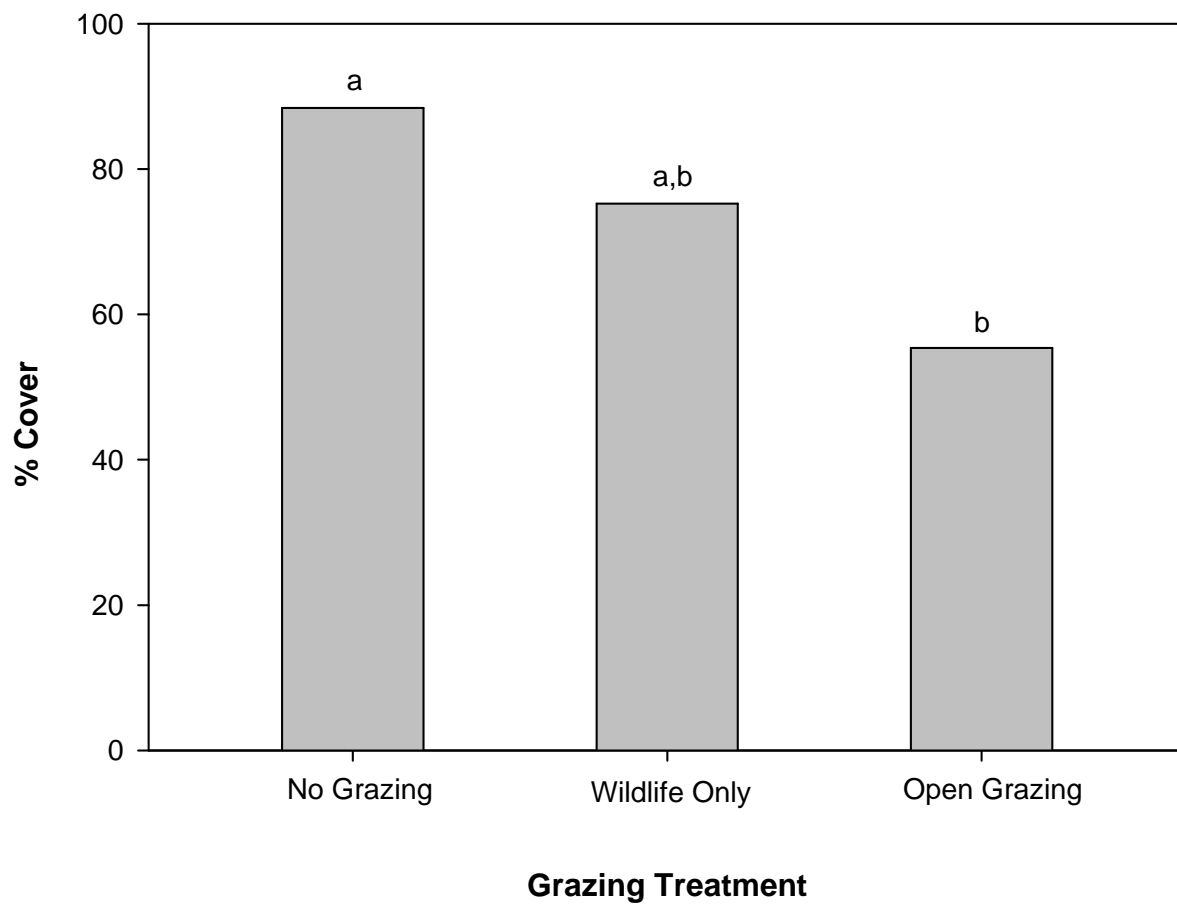


Fig. 1 Grazing treatment effect on Percent Area Burned. Means marked with the same letter are similar, No Grazing and Open Grazing are significantly different from each other, and Wildlife Only is significantly different from both. No Grazing treatment burned 88% of cover, Wildlife Only grazing treatment burned 75% of cover, and Open Grazing burned only 55% of cover.

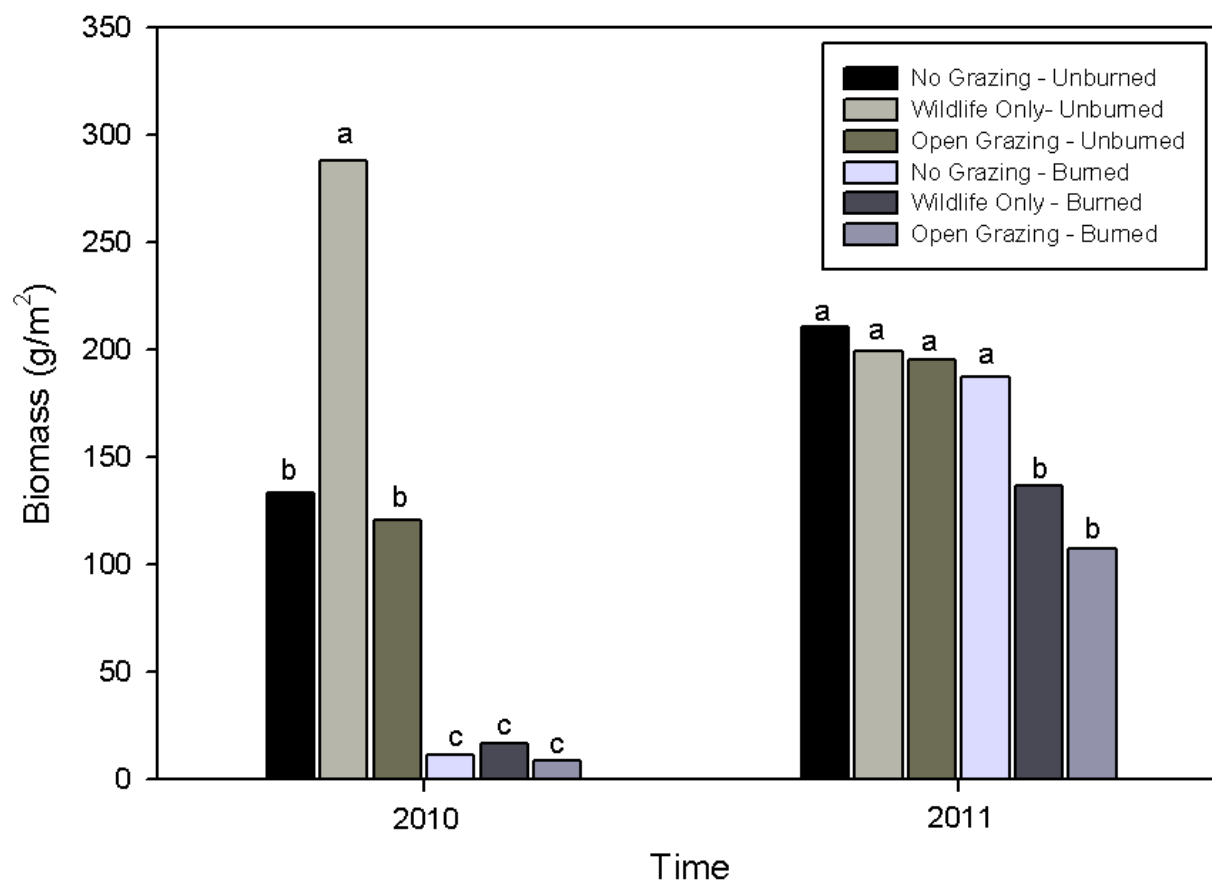


Fig. 2 Biomass measurements from 2010 and 2011 for biomass collected from grasshopper, rabbit, and control cages. Biomass was averaged across all cages and all enclosure sites for each Burning and Grazing treatment. Letters represent differences in means within years. 2010 means are grouped by burn treatment (b, c) with Wildlife Only, Unburned treatment (a) exhibiting greatest biomass. 2011 means are grouped by burn treatment, except No Grazing, Burned treatment recovering to unburned levels.

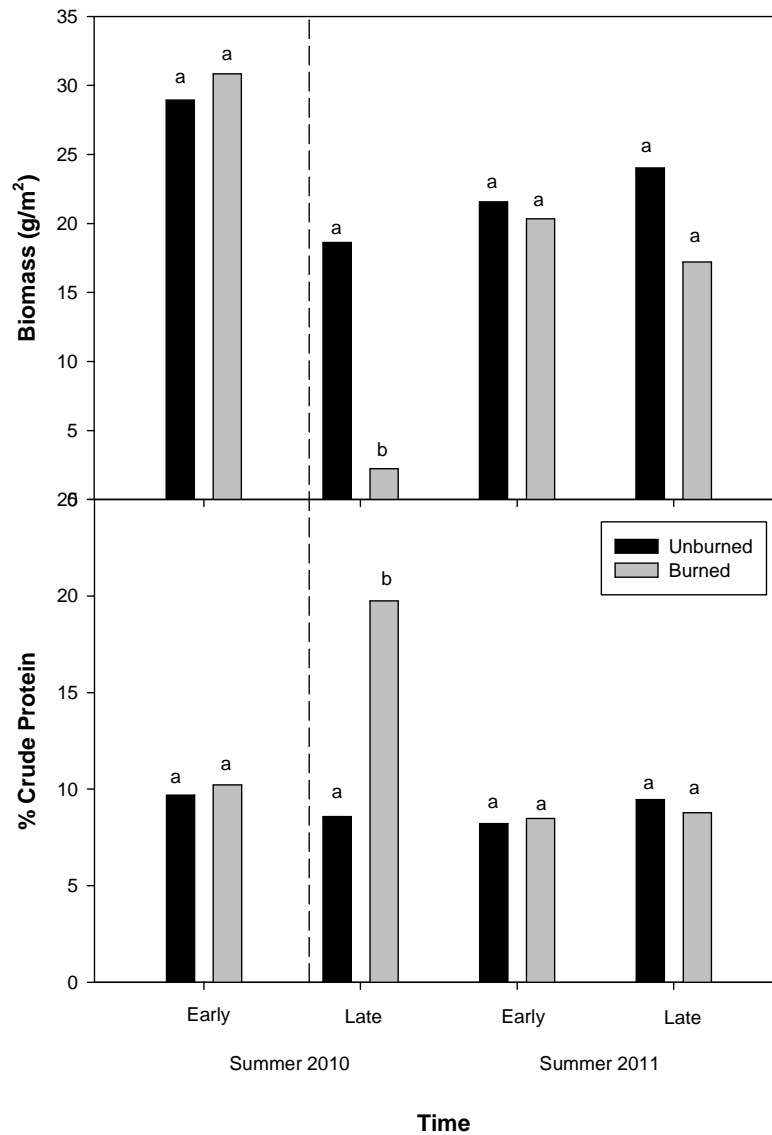


Fig. 3 Forage quantity (biomass, top graph) and forage quality (crude protein, bottom graph). Dashed line represents burn event. Letters indicate differences in means within graphs. Biomass decreased immediately following burning in the Burned treatment in late summer, 2010, but recovered by early summer 2011. Crude protein increased in the Burned treatment in late summer following burning, but positive effects disappeared by early summer, 2011.

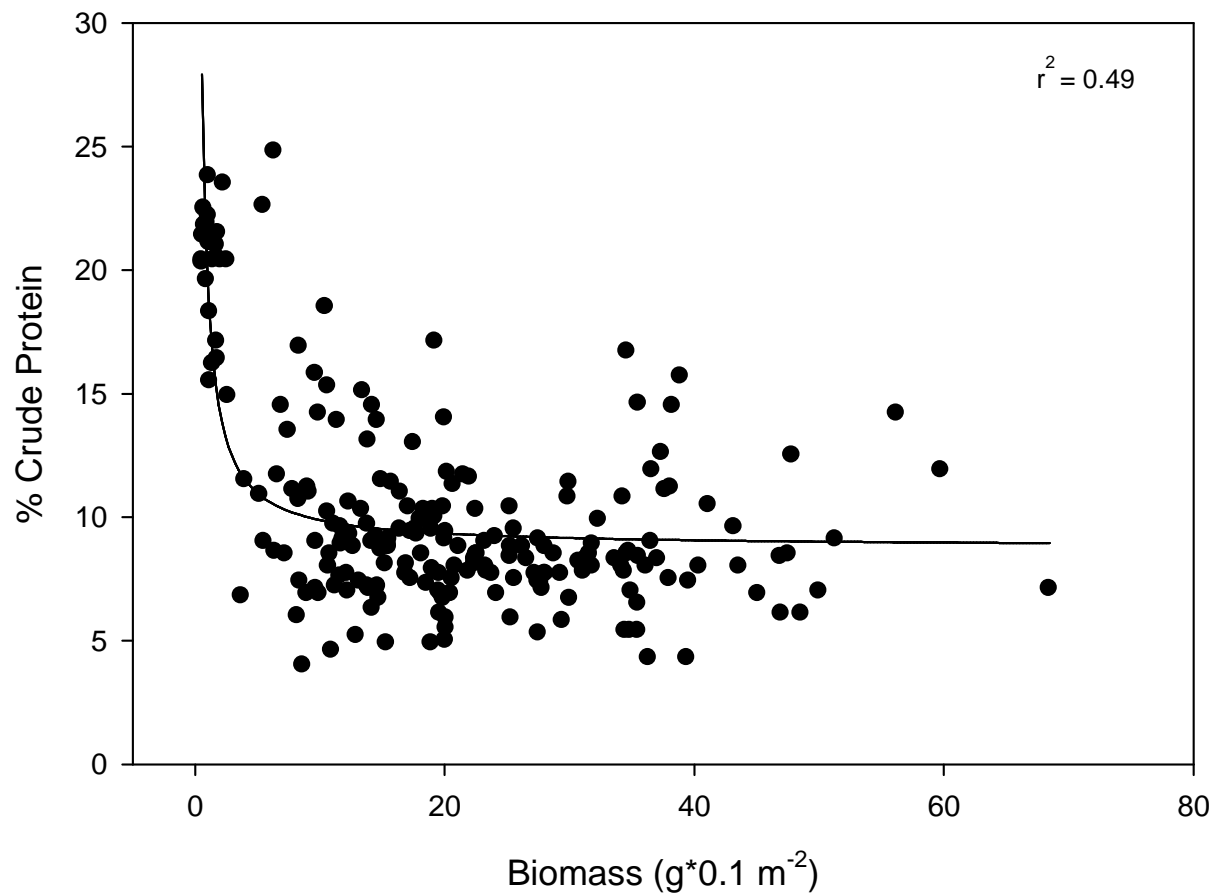


Figure 4 Relationship of crude protein to biomass. Data points represent analysis of biomass and percent crude protein for one pre-burn and three post-burn data collections. An inverse relationship exists between biomass and percent crude protein where biomass is a good indicator of crude protein.

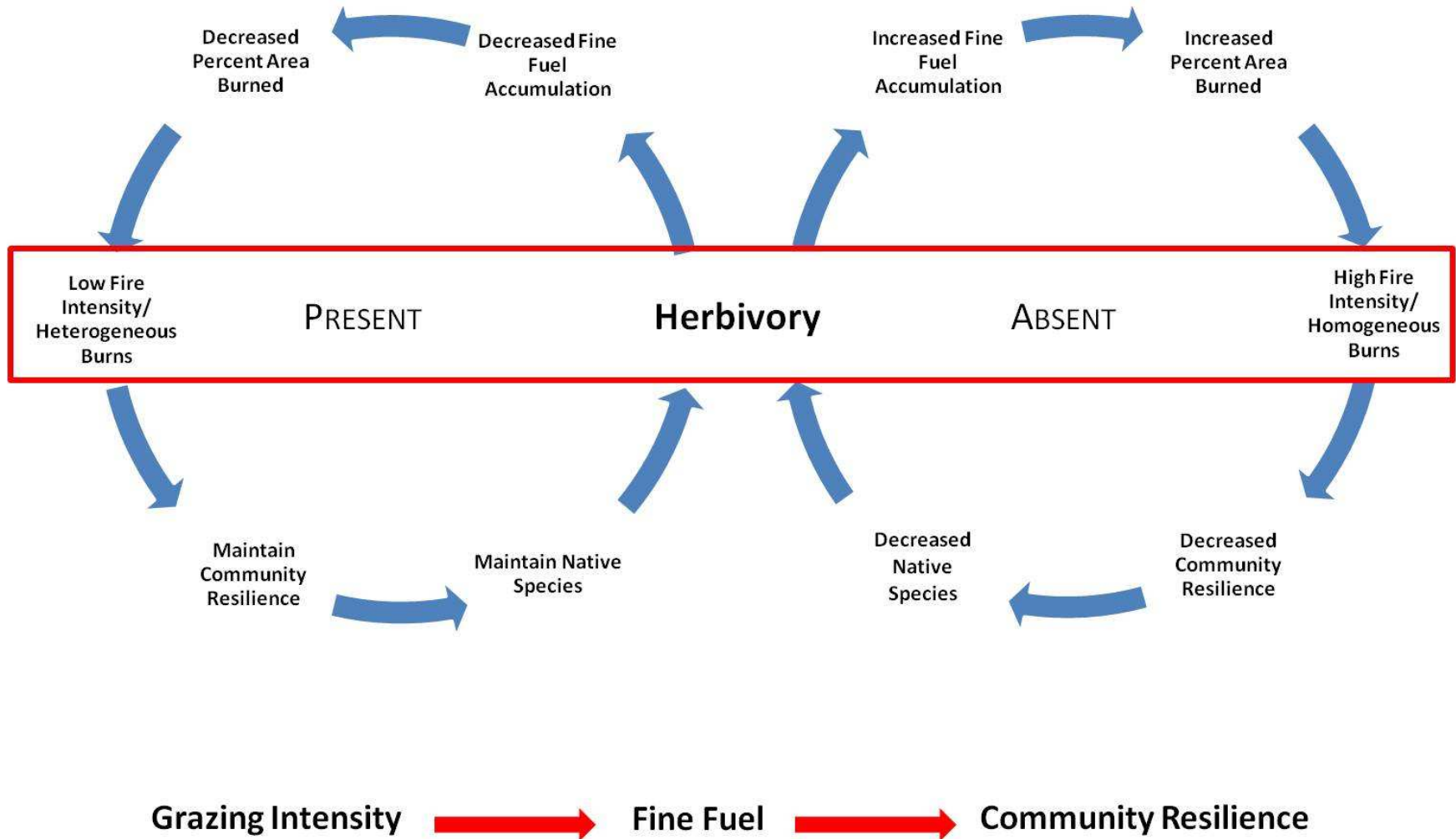


Figure 5 Influence of herbivory on fire behavior in a plant community developed with grazing and fire. Presence of herbivory decreases fine fuel which influences burn patterns and promotes native plant species. Absence of herbivory from the plant community increases fine fuel and promotes homogeneous burns which decrease native plant community. Overall, grazing intensity influences fine fuel and the resiliency of native plant community to invasive species.

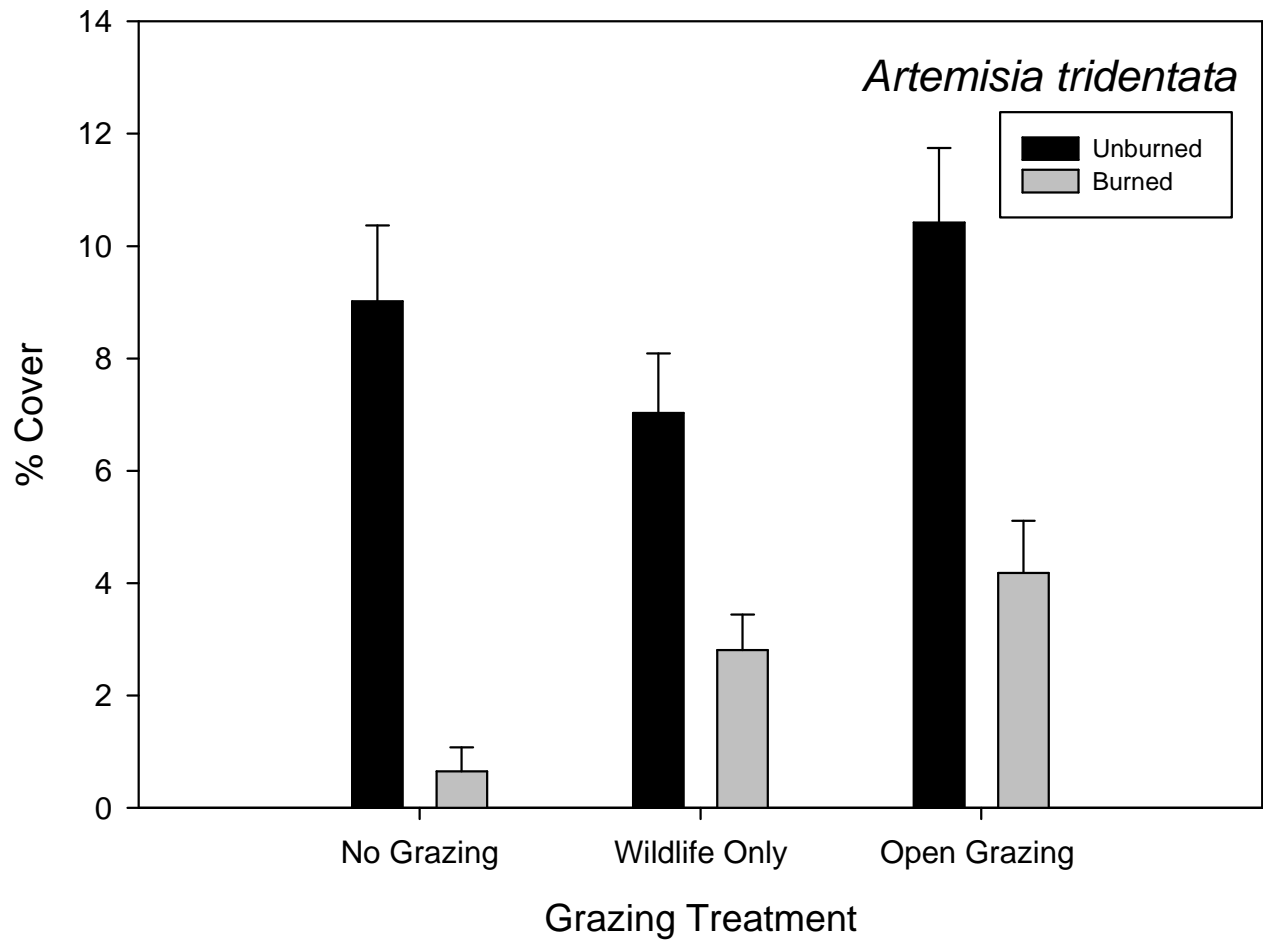


Figure 6 Percent cover of *Artemisia tridentata* spp. *wyomingensis* in year two (2011). Response to grazing and burning treatment with grazing treatment averaged across all exclosure sites. No significant grazing treatment effect was present, but trends are visible in burned treatment representing influence of herbivory on fine fuel and percent area burned. Fuel continuity was greatest and largest area burned in No Grazing treatment, with fuel continuity decreasing as herbivory increased.

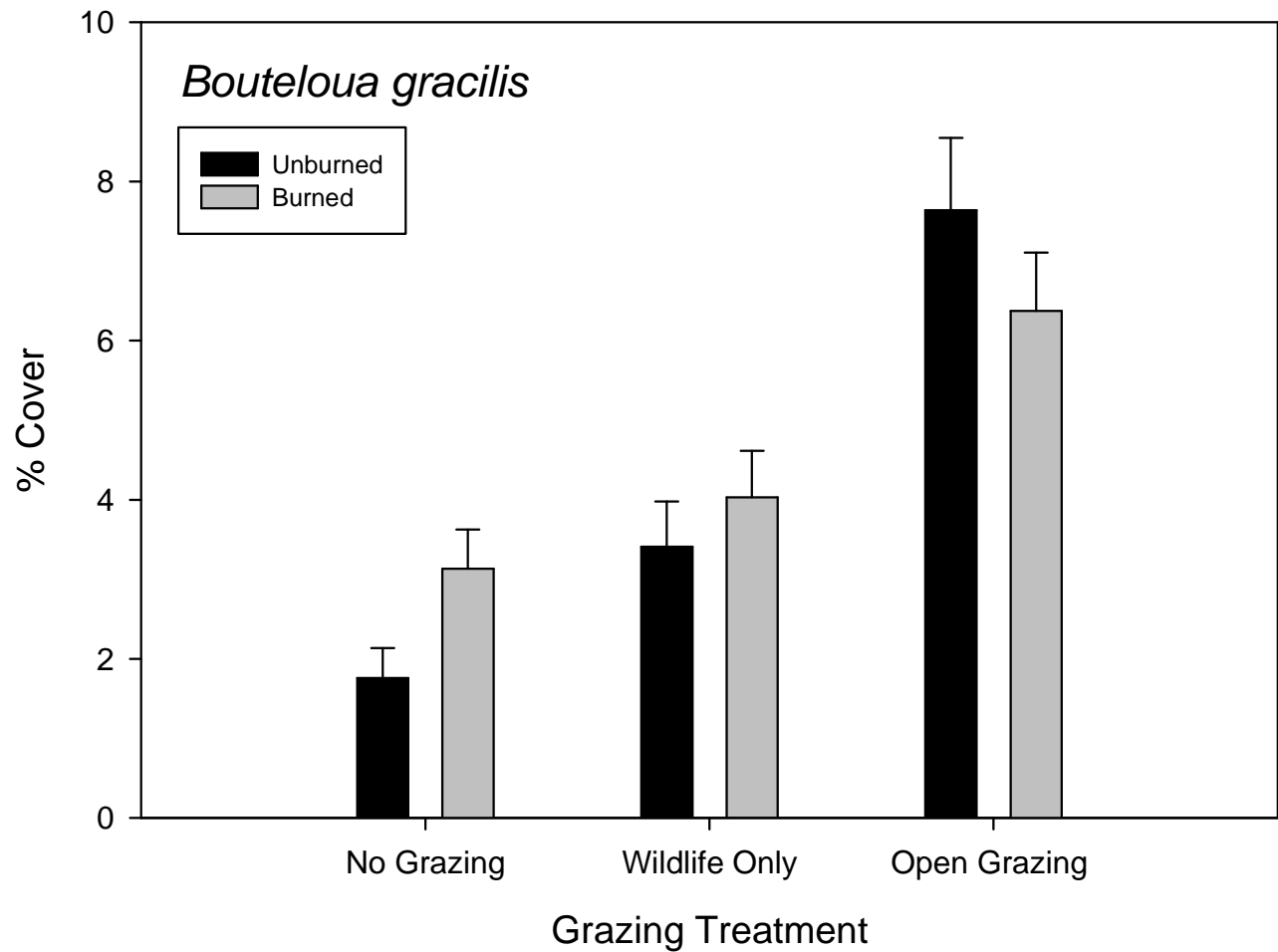


Figure 7 Percent cover of *Bouteloua gracilis* in year two (2011). Response to grazing and burning treatment with grazing treatment averaged across all exclosure sites. No significant grazing treatment effect was present, but trends are visible in burned treatment representing influence of herbivory on fine fuel and percent area burned. Fuel continuity was greatest and largest area burned in No Grazing treatment, with fuel continuity decreasing as herbivory increased. *B. gracilis* tolerates grazing well, represented by increased cover in Open Grazing treatment.

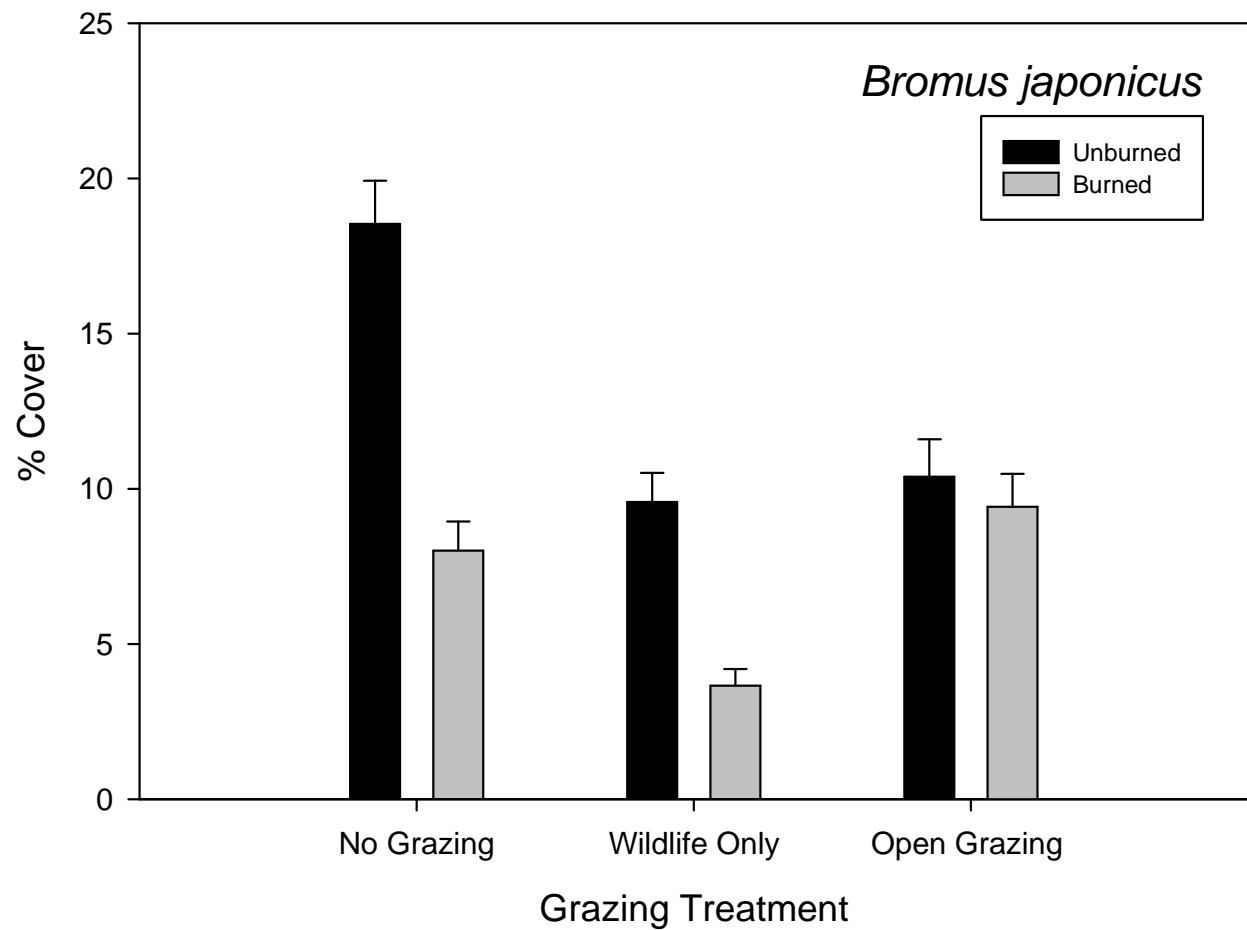


Figure 8 Percent cover of *Bromus japonicus* in year two (2011). Response to grazing and burning treatment with grazing treatment averaged across all exclosure sites. Increased cover in No Grazing, Unburned treatment is supported by studies suggesting *B. japonicus* thrives in environments with increased litter production and declines under grazing and burning treatments.

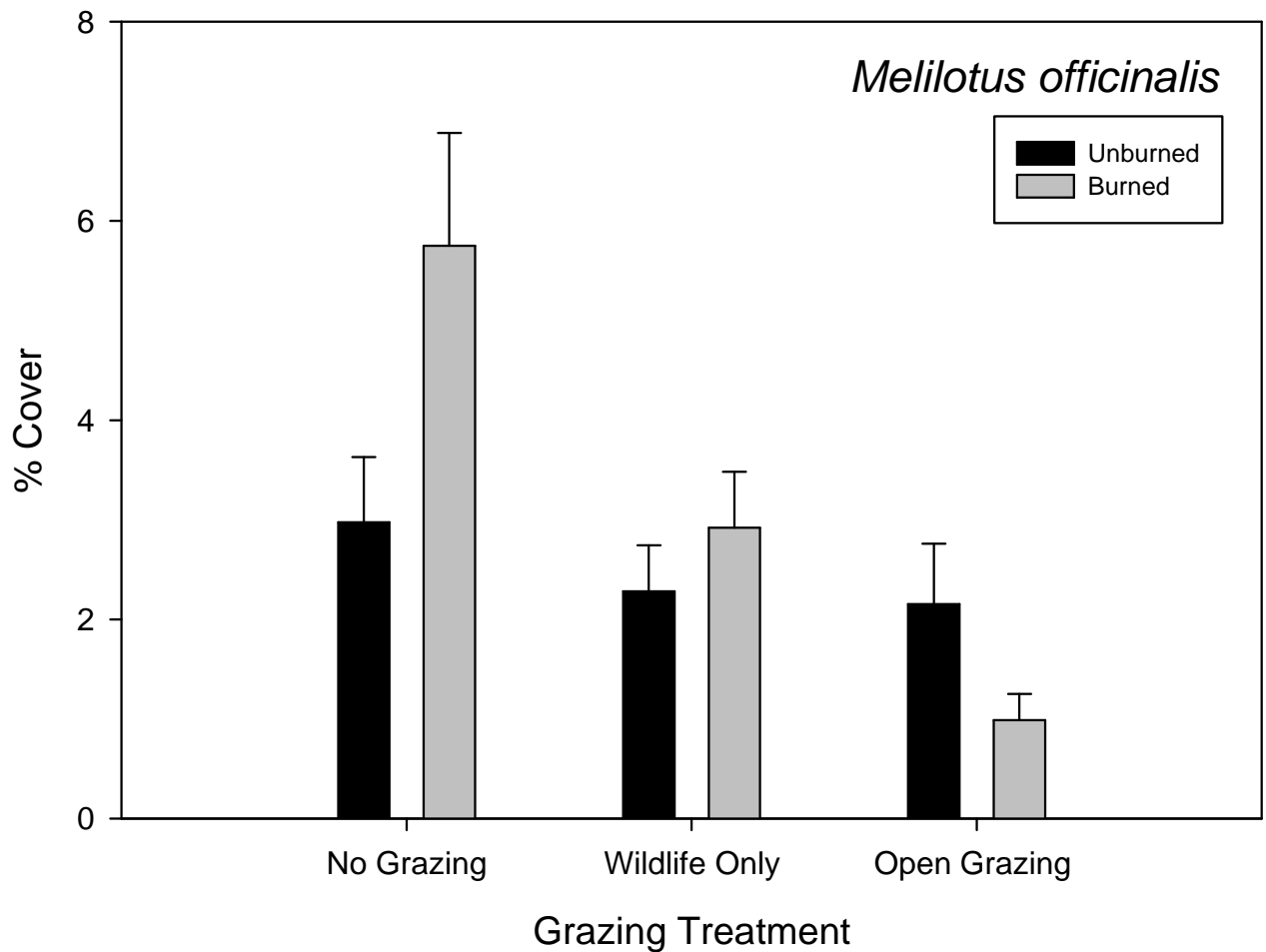


Figure 9 Percent cover of *Melilotus officinalis* in year two (2011). Response to grazing and burning treatment with grazing treatment averaged across all exclosure sites. This biannual plant is highly palatable when young and an important forage species to wildlife and livestock. Decrease in percent cover of *M. officinalis* with increasing grazing pressure is likely due to increased herbivory and is supported by research



Figure 10 Grazing treatments highly influenced percent area burned. Open Grazing (a) averaged 55% of total area burned, Wildlife Only (b) averaged 75% of area burned, and No Grazing (c) averaged 88% of area burned.

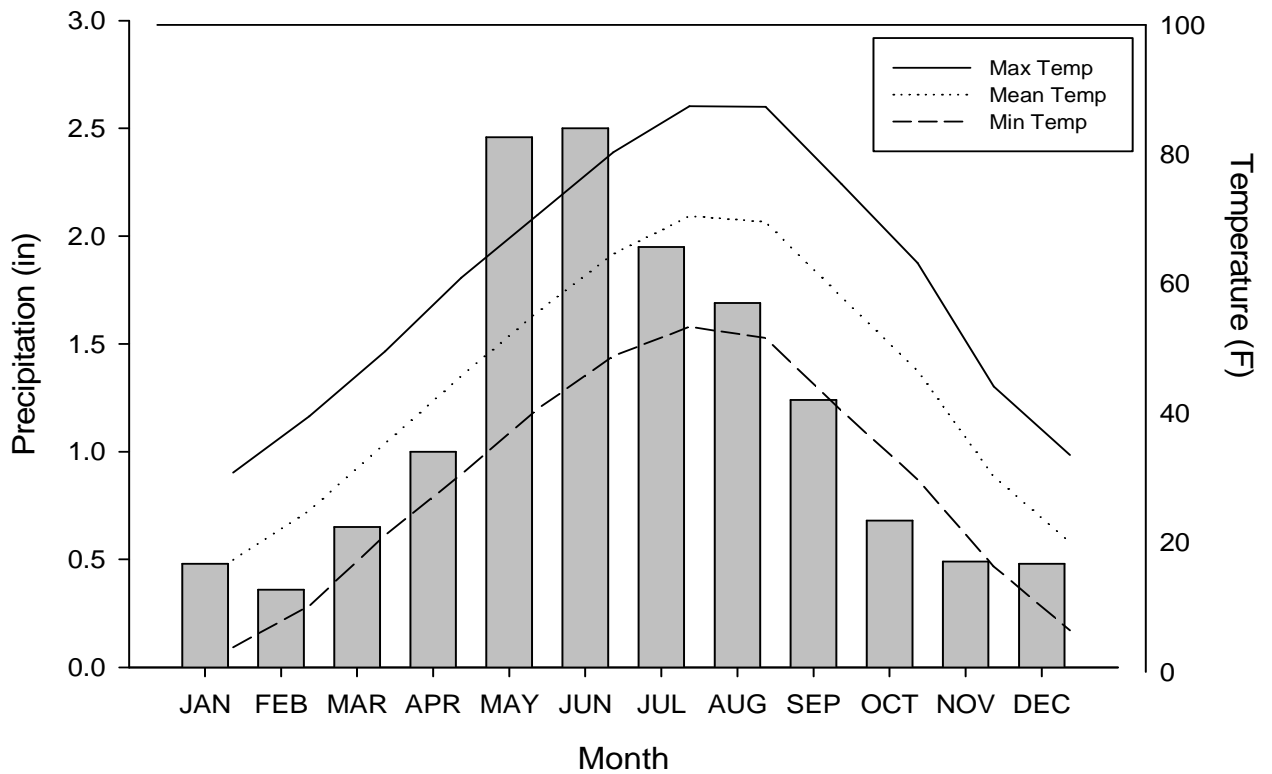


Figure 11 *Artemisia tridentata* spp. *wyomingensis* seedling one growing season following August 4, 2010 burn. Seedlings were present only in areas previously dominated by dense juniper stands where intense fire behavior occurred. No *A. tridentata* shrubs were present near seedlings prior to burning.



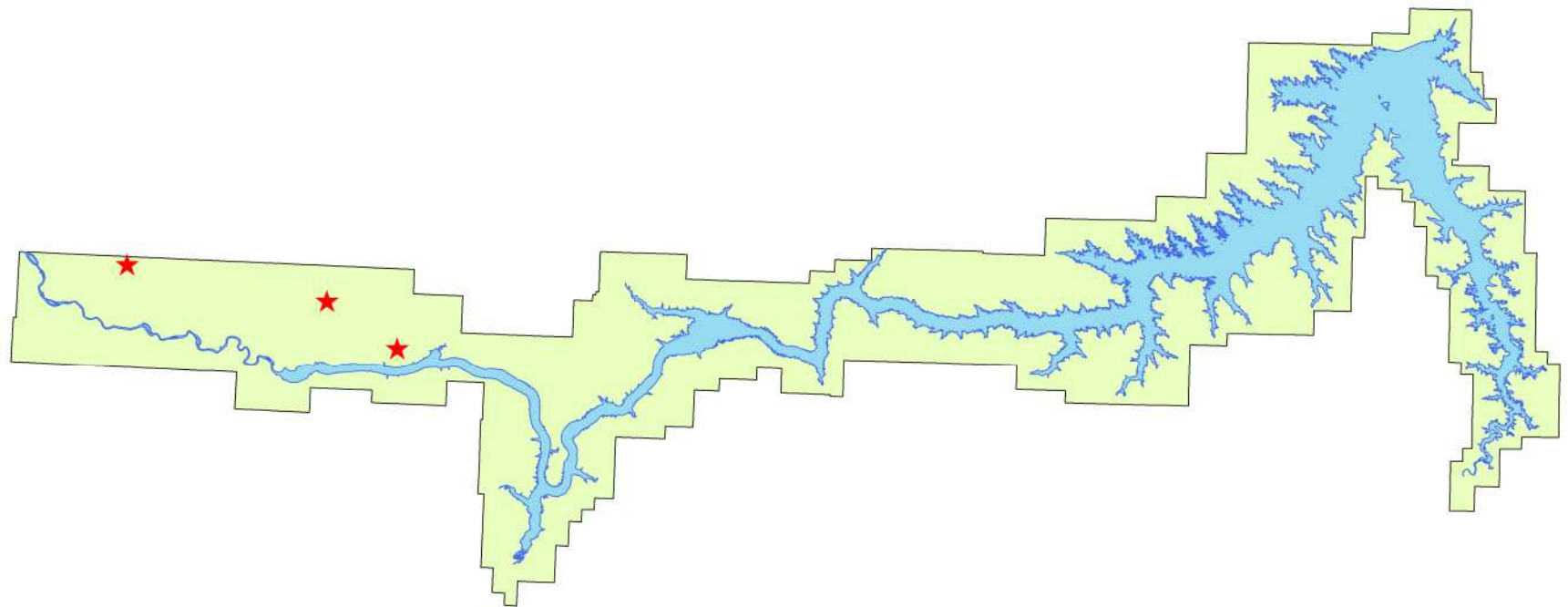
Figure 12 Bentonite clay soil ball with imprint of deer hoof. Clay soil structure binds to shoes and hooves, forming dense balls. Movement of mammals across the landscape may act as a seed dispersal agent.

APPENDIX A



29 year precipitation and temperature averages for Roy, MT (36 km SW of the Charles M. Russell National Wildlife Refuge). Months with highest precipitation are May and June, months with highest temperatures are July and August. Temperatures range from -12 °C to +27 °C and precipitation ranges from 22mm to 32mm per year with the majority of precipitation falling between April and September in the form of rain (Department of Commerce 2000).

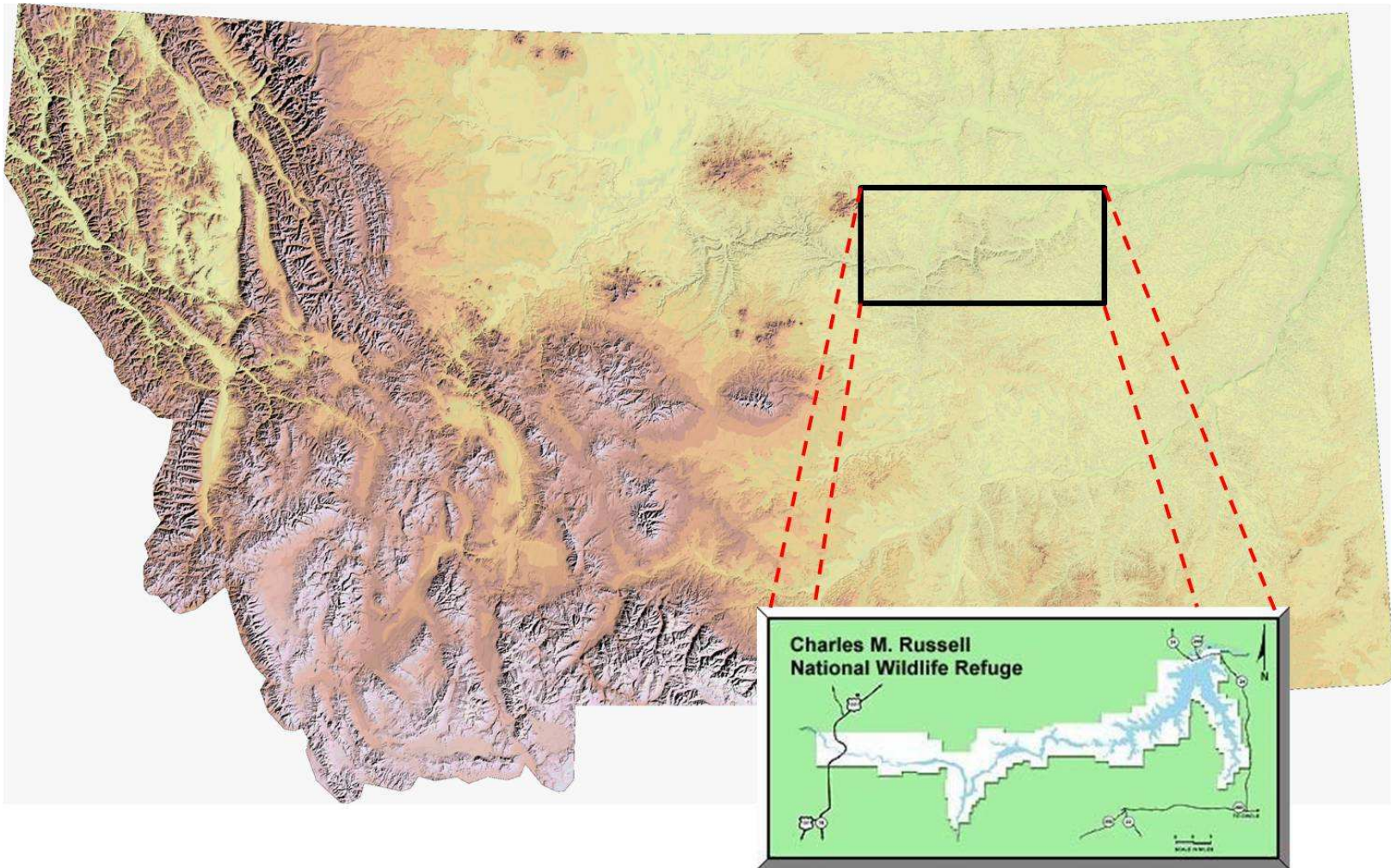
Charles M. Russell National Wildlife Refuge and Fort Peck Reservoir Central Montana



0 5 10 20 30 40 Miles

Legend

- ★ Exclosure Site
- Fort Peck Reservoir
- CMR Refuge Boundary



The 454,000 ha Charles M. Russell National Wildlife Refuge is located in central Montana, 105 km northeast of Lewistown, MT along the Missouri River. The Fort Peck dam and Fort Peck reservoir and Montana state highway 24 mark the eastern boundary, and the Upper Missouri River Wild and Scenic River and US highway 191 boarder the refuge on the west.

APPENDIX B

Comprehensive list of all species encountered during vegetation sampling. In addition to this list, there were 14 unknown species. Total cover of unknowns made up less than 10% of total cover averaged across all sites. Plant taxonomy follows USDA Plants Database. Verification of species identification was provided by NRCS Biologist, Peter Husby, Bozeman, Montana, USA.

	Latin Name	Common Name
Grass	<i>Bouteloua gracilis</i>	Blue grama
	<i>Bromus japonicus</i>	Japanese brome
	<i>Carex brevior</i>	Short beaked sedge
	<i>Elymus lanceolatus</i>	Thickspike wheatgrass
	<i>Hordeum jubatum</i>	Foxtail barley
	<i>Koeleria pyramidata</i>	Prairie junegrass
	<i>Pascopyrum smithii</i>	Western wheatgrass
	<i>Poa secunda</i>	Sandburg bluegrass
	<i>Pseudorogneria spicata</i>	Bluebunch wheatgrass
	<i>Schedonnardus paniculatus</i>	Tumblegrass
	<i>Stipa comata</i>	Needle-and-thread
	<i>Stipa viridula</i>	Green needlegrass
Forb	<i>Achillea millefolium</i>	Common yarrow
	<i>Achillea millefolium</i>	Western yarrow
	<i>Agoseris glauca</i>	Short-beaked agoseris
	<i>Allium textile</i>	Wild onion
	<i>Androsace occidentalis</i>	Western rock jasmine
	<i>Anteneria neglecta</i>	Field pussytoes
	<i>Antennaria parvifolia</i>	Small-leaf pussytoes
	<i>Astragalus agrestis</i>	Purple milkvetch
	<i>Astragalus bisulcatus</i> var. <i>bisulcatus</i>	Twogrooved milkvetch
	<i>Astragalus missouriensis</i>	Missouri milkvetch
	<i>Calochortus nuttallii</i>	Sego lily
	<i>Camelina microcarpa</i>	Littlepod false flax
	<i>Collomia linearis</i>	Slenderleaf collomia
	<i>Comandra umbellata</i>	Bastard toadflax
	<i>Conringia orientalis</i>	Hares ear mustard
	<i>Crepis</i> spp.	Crepis
	<i>Dalea purpurea</i>	Purple prairie clover
	<i>Descurainia pinnata</i>	Tansey mustard
	<i>Descurainia pinnata</i>	Western tansymustard
	<i>Draba aurea</i>	Golden draba
	<i>Erigeron pumulus</i>	Buff fleabane
	<i>Eriogonum ovalifolium</i>	Cushion buckwheat
	<i>Erysimum inconspicuum</i>	Shy wallflower
	<i>Filago arvensis</i>	Field cottonrose
	<i>Galium aparine</i>	Stickywilly
	<i>Gaura coccinea</i>	Scarlet guara

<i>Glycyrrhiza lepidota</i>	American licorice
<i>Grindelia squarrosa</i>	Curlycup gumweed
<i>Hackelia floribunda</i>	Manyflower stickseed
<i>Helianthus</i> spp.	Sunflower
<i>Hymenoxys richardsonii</i>	Pingue rubberweed
<i>Lactuca pulchella</i>	Blue lettuce
<i>Lactuca serriola</i>	Prickly lettuce
<i>Lactuca tatarica</i>	Purple milkweed
<i>Lappula redowski</i>	Sticky annual
<i>Latriplex argentic</i>	Lambsquarters
<i>Lepidium densiflorum</i>	Branched peppergrass
<i>Lesquerella</i> spp.	Bladderpod
<i>Lomatium foeniculaceum</i>	Desert biscuitroot
<i>Machaeranthera canescens</i>	Hoary tansyaster
<i>Maianthemum stellatum</i>	False lily of the valley
<i>Melilotus officinalis</i>	Yellow sweetclover
<i>Monolepis nuttalliana</i>	Nuttall's povertyweed
<i>Musineon divaricatum</i>	Wild parsley
<i>Nothralais</i> spp.	Slender milkweed
<i>Oenothera caespitosa</i>	Tufted evening primrose
<i>Opuntia polyacantha</i>	Prickly pear
<i>Orobancha fasciculata</i>	Clustered broomrape
<i>Pedimelum</i> spp.	Breadroot
<i>Penstemon angustifolius</i>	Narrowleaf beardtongue
<i>Penstemon nitidus</i>	Waxleaf penstemon
<i>Phacelia linearis</i>	Threadleaf phacelia
<i>Phlox hoodii</i>	Hoods phlox
<i>Plantago elongata</i>	Prairie plantain
<i>Plantago patagonica</i>	Indian wheat
<i>Ratibida columifera</i>	Prairie coneflower
<i>Scenecio integerrimus</i>	Lambstongue groundsel
<i>Solidago missouriensis</i>	Missouri goldenrod
<i>Sphaeralcea coccinea</i>	Scarlet globemallow
<i>Taraxacum officinalis</i>	Common dandelion
<i>Thermopsis rhombifolia</i>	Golden pea
<i>Thlaspi arvense</i>	Field pennycress
<i>Tragopogon dubius</i>	Salsify
<i>Vicia americana</i>	American vetch
<i>Viola nuttallii</i>	Nuttall's violet
<i>Yucca glauca</i>	Soapweed yucca

Shrub

<i>Artemesia ludoviciana</i>	Western mugwort
<i>Artemisia cana</i>	Silver sagebrush
<i>Artemisia frigida</i>	Fringed sagebrush
<i>Artemisia tridentatea</i>	Wyoming big sagebrush

<i>Atriplex gardneri</i>	Gardners saltbush
<i>Ericameria nauseosa</i>	Green rubber rabbitbrush
<i>Ericameria nauseosa</i> var. <i>nauseosa</i>	Grey rubber rabbitbrush
<i>Escobaria vivipara</i>	Spinystar
<i>Gutierrezia sarothrae</i>	Broom snakeweed
<i>Juniperus</i> spp.	Juniper
<i>Krascheninnikovia lanata</i>	Winterfat
<i>Opuntia polyacantha</i>	Plains pricklypear
<i>Prunus virginiana</i>	Chokecherry
<i>Rhus aromatica</i>	Skunk brush sumac
<i>Ribes aureum</i>	Golden currant
<i>Ribes</i> spp.	Currant
<i>Rosa arkansana</i>	Prairie rose
<i>Sarcobatus vermiculatus</i>	Greasewood
<i>Symphoricarpos occidentalis</i>	Common snowberry

Average percent cover estimates for species by functional group for year one (2010) in Daubenmire frame collection method. Burned and unburned treatments are pre-burn data. Grazing treatments are averaged across all enclosure sites. Average Total rows are sums of averages. For the following tables: species denoted by asterisk (*) were present in amounts too small to be recognized by two decimal places. There were 14 unknown forb species out of 108 total species. The unknowns are not listed here and make up less than or equal to 10 percent of the foliar cover for each treatment

		2010					
Group	Species	Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
GRASSES	<i>Bouteloua gracilis</i>	5.05	7.25	8.23	3.53	4.27	10.22
	<i>Bromus japonicus</i>	10.62	6.46	8.62	15.92	9.45	8.75
	<i>Carex brevior</i>	0.00	0.33	0.73	0.11	0.23	0.00
	<i>Elymus lanceolatus</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Hordeum jubatum</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Keoheria pyramidata</i>	0.65	1.46	0.28	0.23	1.44	0.89
	<i>Pascopyrum smithii</i>	13.72	12.00	7.91	16.51	15.32	13.91
	<i>Poa secunda</i>	1.50	1.68	2.48	2.15	3.03	2.56
	<i>Poa spp.</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Pseudorogneria spicata</i>	1.66	2.44	1.88	1.09	1.46	0.07
	<i>Schedonnardus paniculatus</i> *	0.00	0.00	0.06	0.00	0.05	0.09
	<i>Stipa comata</i>	0.08	0.19	0.48	0.00	0.46	0.01
	<i>Stipa viridula</i>	4.01	3.98	4.04	1.88	6.50	2.55
	Average Total Grass	2.87	2.75	2.67	3.19	3.25	3.00
FORBS	<i>Achillea millefolium</i>	0.03	0.47	0.10	0.39	0.24	0.45
	<i>Agoseris glauca</i> *	0.00	0.00	0.01	0.01	0.15	0.00
	<i>Allium textile</i>	0.05	0.26	0.01	0.15	0.02	0.56
	<i>Androsace occidentalis</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Antennaria parvifolia</i> *	0.01	0.00	0.00	0.00	0.00	0.00
	<i>Astragalus agrestis</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Astragalus bisculatus</i>	0.16	0.08	0.15	0.00	0.55	0.00
	<i>Astragalus missouriensis</i>	0.00	0.00	0.00	0.00	0.00	0.00

Group	Species	Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
	<i>Calochortus nuttallii</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Collomia linearis</i>	0.15	0.03	0.01	0.02	0.09	0.06
	<i>Comandra umbellatum</i>	2.37	1.08	1.83	2.27	1.09	0.28
	<i>Conringia orientalis</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Crepis spp.</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Dalea purpurea</i>	0.00	0.18	0.00	0.00	0.00	0.00
	<i>Descurainia pinnata</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Erigeron pumulus*</i>	0.03	0.00	0.00	0.00	0.00	0.06
	<i>Eriogonum ovalifolium</i>	0.01	0.04	0.00	0.00	0.00	0.00
	<i>Filago arvensis</i>	0.62	0.98	0.54	2.59	0.84	0.77
	<i>Galium aparine</i>	0.00	0.01	0.00	0.00	0.00	0.03
	<i>Gaura coccinea</i>	0.22	0.09	0.05	0.25	0.37	0.12
	<i>Glycyrrhiza lepidota</i>	0.07	0.16	0.07	0.00	0.00	0.00
	<i>Grindelia squarrosa</i>	0.00	0.03	0.09	0.15	0.40	0.00
	<i>Hackelia floribunda*</i>	0.01	0.00	0.00	0.00	0.00	0.00
	<i>Helianthus spp.*</i>	0.00	0.00	0.00	0.07	0.02	0.00
	<i>Hymenoxys richardsonii</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Lactuca serriola</i>	0.10	0.05	0.18	0.91	0.41	0.35
	<i>Lactuca tatarica*</i>	0.08	0.04	0.00	0.02	0.00	0.00
	<i>Lappula redowski*</i>	0.03	0.41	0.00	0.00	0.00	0.13
	<i>Latriplex argentia</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Lepidium densiflorum*</i>	0.00	0.01	0.58	0.00	0.00	0.01
	<i>Lesquerella spp.</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Lomatium foeniculaceum</i>	0.07	0.01	0.22	0.02	0.01	0.15
	<i>Machaeranthera canescens</i>	0.00	0.01	0.00	0.00	0.00	0.01
	<i>Maianthemum stellatum</i>	0.00	0.76	0.00	0.05	0.57	0.00
	<i>Melilotus officinalis</i>	9.05	10.50	5.47	8.83	6.42	1.36
	<i>Monolepus nuttaliana</i>	0.00	0.00	0.00	0.01	0.00	0.00
	<i>Musineon divaricatum*</i>	0.03	0.00	0.00	0.02	0.04	0.01

Group	Species	Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
	<i>Nothralais spp.</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Pedimelum spp.</i>	0.08	0.71	0.33	0.01	0.44	0.00
	<i>Penstemon nitidus*</i>	0.00	0.00	0.00	0.13	0.00	0.00
	<i>Phacelia linearis*</i>	0.22	0.07	0.00	0.65	0.07	0.14
	<i>Phlox hoodii</i>	0.35	0.12	1.10	0.13	0.34	0.06
	<i>Plantago elongata</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Plantago patagonica</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Ratibida columnifera</i>	0.07	0.21	0.17	0.00	0.02	0.70
	<i>Scenecio integerrimus</i>	0.00	0.00	0.10	0.00	0.00	0.01
	<i>Solidago missouriensis</i>	0.00	0.00	0.00	0.06	0.02	0.00
	<i>Solidago spp.</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Sphaeralcea coccinea</i>	0.09	0.12	0.26	0.28	0.86	0.70
	<i>Taraxacum officinalis</i>	0.81	0.09	0.08	0.04	0.60	0.21
	<i>Thlaspi arvense</i>	0.19	0.00	0.01	1.12	0.51	0.04
	<i>Thermopsis rhombifolia</i>	0.20	0.97	0.53	0.29	0.10	0.00
	<i>Tragopogon dubius</i>	0.08	0.05	0.28	0.07	0.36	0.06
	<i>Vicia americana</i>	1.75	1.77	1.38	0.84	5.03	1.47
	<i>Viola nuttallii</i>	0.00	0.00	0.00	0.00	0.02	0.00
	Average Total Forb	0.28	0.33	0.24	0.32	0.33	0.13
SHRUBS	<i>Artemisia cana</i>	0.00	0.08	0.00	0.00	0.00	0.00
	<i>Artemisia frigida</i>	5.06	3.97	4.29	3.60	3.38	4.20
	<i>Artemisia ludoviciana</i>	0.27	0.08	0.00	0.06	0.00	0.00
	<i>Artemisia tridentata</i>	8.57	6.88	7.18	9.13	7.82	12.27
	<i>Atriplex gardneri</i>	0.47	0.41	0.08	1.50	0.13	0.25
	<i>Ericameria nauseosa</i> var. <i>nauseosa</i>	0.17	0.34	0.08	0.18	0.08	0.00
	<i>Ericameria nauseosa</i>	0.00	0.00	0.00	0.00	0.17	0.00
	<i>Escobaria vivipara*</i>	0.03	0.00	0.01	0.00	0.01	0.00
	<i>Gutierrezia sarothrae</i>	0.00	0.19	0.35	0.06	0.17	0.01

Group	Species	Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
	<i>Juniperus spp.</i>	4.99	8.10	2.75	0.91	1.96	0.89
	<i>Krascheninnikovia lanata</i> *	0.00	0.00	0.08	0.00	0.00	0.17
	<i>Opuntia polyacantha</i>	1.48	1.16	1.90	2.20	1.82	1.41
	<i>Prunus virginiana</i>	0.00	0.00	0.00	0.01	0.00	0.38
	<i>Rhus aromatica</i>	1.04	0.23	0.27	0.15	0.40	0.23
	<i>Ribes aureum</i> *	0.66	0.16	0.00	0.00	0.00	0.16
	<i>Ribes spp.</i>	0.00	0.01	0.00	0.00	0.00	0.16
	<i>Rosa arkansana</i>	0.84	0.40	0.38	0.07	0.32	0.00
	<i>Sarcobatus vermiculatus</i>	0.17	0.27	0.00	2.01	0.22	0.57
	<i>Symphoricarpos occidentalis</i>	0.83	0.77	0.54	0.53	0.22	0.33
	Average Total Shrub	1.23	1.14	0.88	1.02	0.83	1.05

Average percent cover estimates for species by functional group for year two (2011) in Daubenmire frame collection method. Grazing treatments are averaged across all exclosure sites. Average Total rows are sums of averages.

Group	Species	2011					
		Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
GRASSES	<i>Bouteloua gracilis</i>	3.13	4.03	6.37	1.76	3.41	7.64
	<i>Bromus japonicus</i>	8.01	3.66	9.42	18.53	9.58	10.38
	<i>Carex brevior</i>	0.07	0.54	1.37	0.08	1.23	0.06
	<i>Elymus lanceolatus</i>	0.00	0.00	0.00	0.00	0.00	3.01
	<i>Hordeum jubatum</i> *	0.00	0.00	0.04	0.00	0.00	0.01
	<i>Keoheria pyramidata</i>	0.62	2.14	1.27	0.15	2.15	0.70
	<i>Pascopyrum smithii</i>	12.71	11.26	9.38	13.15	11.34	12.08
	<i>Poa secunda</i>	3.87	5.38	5.65	4.83	4.06	5.97
	<i>Poa spp.</i>	0.00	0.01	0.00	0.00	0.00	0.00
	<i>Pseudorogneria spicata</i>	0.00	0.04	0.07	1.10	0.32	0.00
	<i>Schedonnardus paniculatus</i>	0.00	0.01	0.10	0.00	0.05	0.09
	<i>Stipa comata</i>	0.00	0.00	0.69	0.00	0.27	0.00
	<i>Stipa viridula</i>	1.70	3.86	2.81	2.00	5.65	3.12
	Average Total Grass	2.32	2.38	2.86	3.20	2.93	3.08
FORBS	<i>Achillea millefolium</i>	0.07	1.01	0.37	0.68	0.69	0.37
	<i>Agoseris glauca</i>	0.03	0.07	0.04	0.06	0.00	0.04
	<i>Allium textile</i> *	0.01	0.08	0.10	0.00	0.04	0.14
	<i>Androsace occidentalis</i>	0.49	0.23	0.17	0.42	0.02	0.01
	<i>Antennaria parvifolia</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Astragalus agrestis</i> *	0.00	0.40	0.00	0.00	0.13	0.00
	<i>Astragalus bisculatus</i>	0.00	0.00	0.01	0.00	0.00	0.00
	<i>Astragalus missouriensis</i> *	0.00	0.00	0.13	0.02	0.23	0.00
	<i>Calochortus nuttallii</i>	0.00	0.00	0.00	0.00	0.01	0.00
	<i>Collomia linearis</i>	0.22	0.23	0.03	0.95	0.38	0.30
	<i>Comandra umbellatum</i>	1.33	0.88	1.17	0.92	0.94	0.32

Group	Species	Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
	<i>Conringia orientalis</i>	0.00	0.00	0.00	0.00	0.06	0.00
	<i>Crepis spp.</i>	0.18	0.03	0.01	0.10	0.01	0.82
	<i>Dalea purpurea</i>	0.01	0.03	0.09	0.00	0.00	0.00
	<i>Descurainia pinnata</i>	0.60	0.49	0.35	0.01	0.01	0.19
	<i>Erigeron pumulus</i>	0.01	0.00	0.01	0.00	0.01	0.23
	<i>Eriogonum ovalifolium</i>	0.00	0.07	0.00	0.02	0.00	0.01
	<i>Filago arvensis</i>	4.08	3.62	2.97	1.39	1.53	0.40
	<i>Galium aparine</i>	0.00	0.19	0.00	0.01	0.00	0.26
	<i>Gaura coccinea</i>	0.35	0.32	0.29	0.24	0.80	0.22
	<i>Glycyrrhiza lepidota*</i>	0.00	0.00	0.03	0.00	0.00	0.00
	<i>Grindelia squarrosa</i>	0.01	0.03	0.04	0.00	0.02	0.00
	<i>Hackelia floribunda*</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Helianthus annuus*</i>	0.10	0.00	0.00	0.09	0.01	0.00
	<i>Hymenoxys richardsonii</i>	0.00	0.00	0.00	0.00	0.06	0.00
	<i>Lactuca serriola</i>	0.37	0.03	0.08	0.85	0.27	0.22
	<i>Lactuca tatarica</i>	0.00	0.00	0.01	0.00	0.01	0.00
	<i>Lappula redowski</i>	0.53	0.12	0.54	0.34	1.03	0.05
	<i>Latriplex argentia*</i>	0.00	0.00	0.00	0.01	0.00	0.00
	<i>Lepidium densiflorum</i>	1.40	0.10	0.19	0.06	0.63	0.16
	<i>Lesquerella spp.*</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Lomatium foeniculaceum</i>	0.92	0.59	0.32	1.29	0.53	0.52
	<i>Machaeranthera canescens</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Maianthemum stellatum</i>	0.00	0.16	0.00	0.16	0.29	0.00
	<i>Melilotus officinalis</i>	5.75	2.92	0.99	2.97	2.28	2.15
	<i>Monolepus nuttaliana*</i>	0.00	0.00	0.00	0.01	0.00	0.00
	<i>Musineon divaricatum</i>	0.03	0.01	0.01	0.01	0.01	0.01
	<i>Nothralais spp.*</i>	0.00	0.11	0.23	0.00	0.40	0.00
	<i>Pediomelum spp.</i>	0.08	0.24	0.71	0.02	0.31	0.00
	<i>Penstemon nitidus*</i>	0.07	0.00	0.00	0.00	0.00	0.00

Group	Species	Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
	<i>Phacelia linearis</i>	1.54	0.44	1.30	0.43	0.09	0.03
	<i>Phlox hoodii</i>	0.15	0.15	0.68	0.06	0.52	0.62
	<i>Plantago elongata</i>	0.60	0.28	0.63	0.04	0.11	0.22
	<i>Plantago patagonica</i>	0.62	0.12	0.40	0.14	0.19	0.37
	<i>Ratibida columnifera</i>	0.23	0.16	0.20	0.04	0.11	0.43
	<i>Scenecio integerrimus</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Solidago missouriensis*</i>	0.00	0.00	0.00	0.06	0.00	0.00
	<i>Solidago spp.</i>	0.03	0.01	0.00	0.15	0.00	0.00
	<i>Sphaeralcea coccinea</i>	0.29	0.06	0.27	0.47	0.29	0.29
	<i>Taraxacum officinalis</i>	1.56	1.01	1.20	0.84	0.74	0.68
	<i>Thlaspi arvense</i>	0.17	0.38	0.43	1.80	0.41	0.01
	<i>Thermopsis rhombifolia</i>	0.48	1.13	0.73	0.28	0.01	0.07
	<i>Tragopogon dubius</i>	1.38	0.60	0.48	0.30	0.18	0.25
	<i>Vicia americana</i>	2.73	3.47	3.41	1.79	4.65	2.25
	<i>Viola nuttallii*</i>	0.00	0.01	0.00	0.00	0.09	0.00
	Average Total Forb	0.44	0.33	0.31	0.29	0.30	0.19
SHRUBS	<i>Artemisia cana*</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Artemisia frigida</i>	0.47	1.04	2.22	1.66	2.13	4.51
	<i>Artemisia ludoviciana</i>	0.08	0.40	0.00	0.01	0.08	0.01
	<i>Artemisia tridentata</i>	0.65	2.81	4.18	9.02	7.02	10.42
	<i>Atriplex gardneri</i>	0.08	0.01	0.26	0.99	0.06	0.36
	<i>Ericameria nauseosa</i> var. <i>nauseosa</i>	0.17	0.03	0.00	0.07	0.19	0.00
	<i>Ericameria nauseosa*</i>	0.00	0.00	0.00	0.00	0.00	0.00
	<i>Escobaria vivipara*</i>	0.08	0.00	0.00	0.00	0.01	0.01
	<i>Gutierrezia sarothrae</i>	0.00	0.01	0.16	0.00	0.13	0.07
	<i>Juniperus spp.*</i>	0.00	0.97	0.00	0.55	2.83	1.47
	<i>Krascheninnikovia lanata*</i>	0.00	0.00	0.00	0.00	0.00	0.07
	<i>Opuntia polyacantha</i>	0.96	0.64	1.26	1.51	0.95	1.28

Group	Species	Burned			Unburned		
		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
	<i>Prunus virginiana</i>	0.00	0.00	0.00	0.00	0.00	0.58
	<i>Rhus aromatica</i>	0.07	0.27	0.16	0.52	0.00	0.07
	<i>Ribes aureum</i>	0.07	0.00	0.00	0.00	0.00	0.01
	<i>Ribes spp.*</i>	0.00	0.00	0.00	0.00	0.00	0.07
	<i>Rosa arkansana</i>	0.67	0.47	0.61	0.50	0.09	0.01
	<i>Sarcobatus vermiculatus</i>	0.00	0.00	0.00	3.10	0.00	0.58
	<i>Symphoricarpos occidentalis</i>	0.70	0.81	0.18	0.44	0.08	0.05
	Average Total Shrub	0.20	0.37	0.44	0.92	0.68	0.98

Average percent cover estimates for shrub species for June year one (2010) in five meter collection method. Grazing treatments are averaged across all exclosure sites. Average Total rows are sums of averages. Burned and Unburned treatments are pre-burn data.

		Early 2010					
		Burned			Unburned		
Species		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
SHRUBS	<i>Artemisia cana</i>	0.00	0.10	0.00	0.09	0.52	0.10
	<i>Artemisia ludoviciana</i> *	0.58	1.21	0.10	0.50	0.00	0.00
	<i>Artemisia tridentata</i>	20.23	16.88	19.30	17.84	16.91	26.74
	<i>Atriplex gardneri</i>	2.47	0.45	0.23	1.99	0.51	0.63
	<i>Ericameria nauseosa</i> var. <i>nauseosa</i>	0.23	0.78	0.41	0.61	0.21	0.54
	<i>Ericameria nauseosa</i> *	0.00	0.10	0.00	0.00	0.01	0.00
	<i>Gutierrezia sarothrae</i>	0.01	0.13	1.60	0.01	0.02	0.22
	<i>Juniperus</i> spp.	5.79	10.59	6.05	2.13	4.20	2.87
	<i>Krascheninnikovia lanata</i> *	0.58	0.00	0.01	0.00	0.01	0.12
	<i>Opuntia polyacantha</i>	6.14	5.38	8.12	5.97	4.72	10.37
	<i>Prunus virginiana</i>	0.00	0.00	0.10	0.50	0.00	1.27
	<i>Rhus aromatica</i>	4.08	0.53	1.04	2.29	0.64	1.48
	<i>Ribes aureum</i> *	0.57	0.10	0.00	0.00	0.00	0.00
	<i>Ribes</i> spp.*	0.00	0.10	0.00	0.00	0.00	0.10
	<i>Rosa arkansana</i>	0.90	2.09	1.91	0.10	0.10	0.01
	<i>Sarcobatus vermiculatus</i>	0.74	0.11	0.11	5.64	0.66	1.66
	<i>Symphoricarpos occidentalis</i>	2.25	4.04	2.90	2.13	0.01	0.53
	Average Total Shrub	2.62	2.51	2.46	2.34	1.68	2.74

Average percent cover estimates for shrub species for late summer year one (2010) in five meter collection method. Grazing treatments are averaged across all exclosure sites. Average Total rows are sums of averages.

		Late 2010					
		Burned			Unburned		
Species		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
SHRUBS	<i>Artemisia cana</i> *	0.00	0.00	1.27	0.09	0.61	0.00
	<i>Artemisia ludoviciana</i> *	0.00	0.21	0.00	0.00	0.00	0.10
	<i>Artemisia tridentata</i>	1.03	8.01	9.84	20.78	20.63	26.74
	<i>Atriplex gardneri</i>	0.44	0.98	0.22	1.58	0.51	0.72
	<i>Ericameria nauseosa</i> var. <i>nauseosa</i>	0.01	0.11	0.31	0.79	0.30	0.11
	<i>Ericameria nauseosa</i> *	0.00	0.00	0.00	0.00	0.10	0.00
	<i>Gutierrezia sarothrae</i>	0.12	0.22	0.32	0.19	1.33	0.42
	<i>Juniperus</i> spp.	0.00	0.55	0.01	1.94	4.71	2.87
	<i>Krascheninnikovia lanata</i> *	0.11	0.00	0.01	0.00	0.01	0.12
	<i>Opuntia polyacantha</i>	1.60	3.53	9.49	9.95	7.76	12.68
	<i>Prunus virginiana</i>	0.00	0.00	0.00	0.50	0.00	1.27
	<i>Rhus aromatica</i>	0.03	0.57	0.31	2.76	1.75	1.38
	<i>Ribes aureum</i> *	0.00	0.00	0.00	0.00	0.00	0.10
	<i>Ribes</i> spp.	0.00	0.00	0.00	0.00	0.00	0.10
	<i>Rosa arkansana</i>	0.12	0.31	0.31	0.10	0.39	0.01
	<i>Sarcobatus vermiculatus</i>	0.11	0.01	0.21	5.85	0.62	1.61
	<i>Symphoricarpos occidentalis</i> *	0.01	0.21	0.00	1.94	0.52	0.53
	Average Total Shrub	0.21	0.87	1.31	2.73	2.31	2.87

Average percent cover estimates for shrub species for June year two (2011) in five meter collection method. Grazing treatments are averaged across all enclosure sites. Average Total rows are sums of averages.

		Early 2011					
		Burned			Unburned		
Species		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
SHRUBS	<i>Artemisia cana</i> *	0.00	0.00	1.27	0.00	0.52	0.00
	<i>Artemisia ludoviciana</i>	1.15	1.52	0.11	0.50	0.61	0.10
	<i>Artemisia tridentata</i>	2.06	7.25	11.84	14.88	16.82	19.77
	<i>Atriplex gardneri</i>	1.01	0.74	0.32	2.08	0.92	1.69
	<i>Ericameria nauseosa</i> var. <i>nauseosa</i>	0.58	0.11	0.31	0.29	0.30	0.11
	<i>Ericameria nauseosa</i>	0.00	0.00	0.10	0.00	0.10	0.00
	<i>Gutierrezia sarothrae</i> *	0.00	0.01	0.13	0.09	0.21	0.03
	<i>Juniperus</i> spp.	0.00	1.31	0.01	1.94	3.06	2.07
	<i>Krascheninnikovia lanata</i> *	0.21	0.11	0.10	0.00	0.01	0.21
	<i>Opuntia polyacantha</i>	4.38	5.31	11.10	7.15	5.01	8.68
	<i>Prunus virginiana</i>	0.00	0.00	0.00	0.09	0.00	1.27
	<i>Rhus aromatica</i>	0.81	0.12	0.21	0.89	0.73	0.41
	<i>Ribes aureum</i> *	0.57	0.00	0.00	0.00	0.00	0.53
	<i>Ribes</i> spp.	0.00	0.00	0.00	0.09	0.00	0.10
	<i>Rosa arkansana</i>	0.90	1.21	2.11	0.51	1.42	0.01
	<i>Sarcobatus vermiculatus</i>	0.58	0.11	0.11	5.76	0.53	1.60
	<i>Symphoricarpos occidentalis</i>	2.61	3.38	1.37	1.94	0.62	1.80
	Average Total Shrub	0.87	1.25	1.71	2.13	1.81	2.26

Average percent cover estimates for shrub species for late summer year two (2011) in five meter collection method. Grazing treatments are averaged across all exclosure sites. Totals are sums of averages.

		Late 2011					
		Burned			Unburned		
Species		No Grazing	Wildlife Only	Open Grazing	No Grazing	Wildlife Only	Open Grazing
SHRUBS							
	<i>Artemisia cana</i> *	0.00	0.00	0.00	0.00	0.10	0.00
	<i>Artemisia ludoviciana</i>	0.79	1.72	0.10	0.50	0.61	0.63
	<i>Artemisia tridentata</i>	1.70	6.60	10.70	15.91	15.85	22.17
	<i>Atriplex gardneri</i>	1.12	0.74	0.32	2.58	1.20	0.97
	<i>Ericameria nauseosa</i> var. <i>nauseosa</i>	0.33	0.32	0.41	0.88	1.42	0.11
	<i>Ericameria nauseosa</i> *	0.00	0.00	0.00	0.00	0.10	0.00
	<i>Gutierrezia sarothrae</i> *	0.00	0.11	0.41	0.09	0.72	0.22
	<i>Juniperus</i> spp.	0.00	1.31	0.10	1.94	3.87	2.07
	<i>Krascheninnikovia lanata</i>	0.22	0.01	0.10	0.00	0.01	0.85
	<i>Opuntia polyacantha</i>	4.79	5.23	7.52	13.16	8.68	11.24
	<i>Prunus virginiana</i>	0.00	0.00	0.53	0.00	0.00	1.27
	<i>Rhus aromatica</i>	2.36	1.87	0.84	2.79	1.25	0.31
	<i>Ribes aureum</i> *	0.57	0.10	0.00	0.00	0.00	0.53
	<i>Ribes</i> spp.	0.00	0.00	0.00	0.00	0.00	0.53
	<i>Rosa arkansana</i>	2.25	2.62	2.97	1.09	0.72	0.64
	<i>Sarcobatus vermiculatus</i>	0.68	0.01	0.11	5.69	0.81	1.80
	<i>Symphoricarpos occidentalis</i>	3.50	3.83	2.67	2.03	0.61	1.80
	Average Total Shrub	1.08	1.44	1.58	2.75	2.11	2.66

APPENDIX C



Agate Ridge Exclosure, No Grazing on left, Wildlife Only on right. Photo taken 3 December 1965



Agate Ridge Exclosure, No Grazing on left, Wildlife Only on right. Photo taken 2 July 2011



Agate Ridge Exclosure, Wildlife Only on left, Open Grazing on right. Photo taken 3 December 1965



Agate Ridge Exclosure, Wildlife Only on left, Open Grazing on right. Photo taken 16 April 1985



Agate Ridge Exclosure, Wildlife Only on left, Open Grazing on right. Photo taken 2 July 2011



Agate Ridge Exclosure, No Grazing on left, Open Grazing on right. Photo taken 3 December 1965



Agate Ridge Exclosure, No Grazing on left, Open Grazing on right. Photo taken 2 July 2011



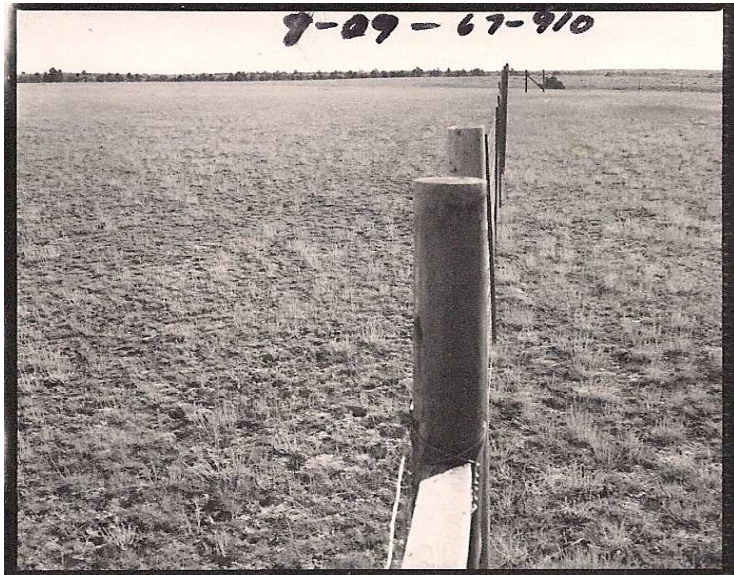
Agate Ridge Exclosure, Open Grazing on left, No Grazing on right. Photo taken 27 July 1968



Agate Ridge Exclosure, Open Grazing on left, No Grazing on right. Photo taken 16 April 1985



Agate Ridge Exclosure, Open Grazing on left, No Grazing on right. Photo taken 2 July 2011



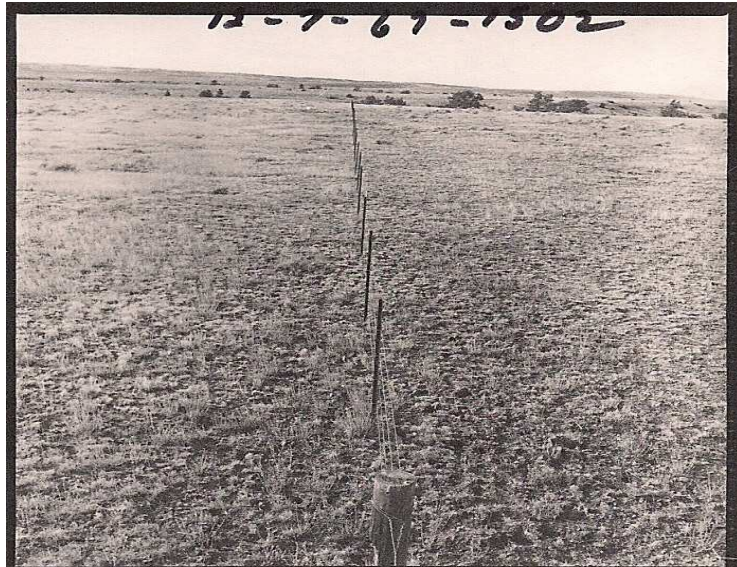
Opuntia Exclosure, Open Grazing on left, No Grazing on right. Photo taken 9 September 1965



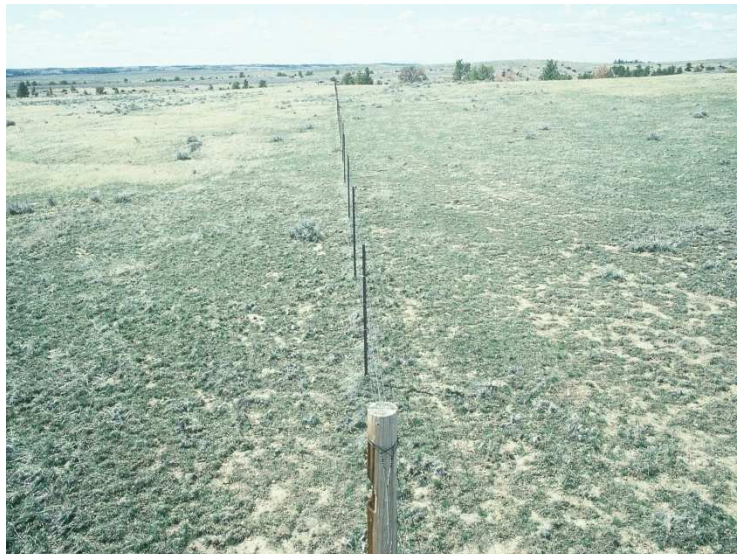
Opuntia Exclosure, Open Grazing on left, No Grazing on right. Photo taken 16 April 1965



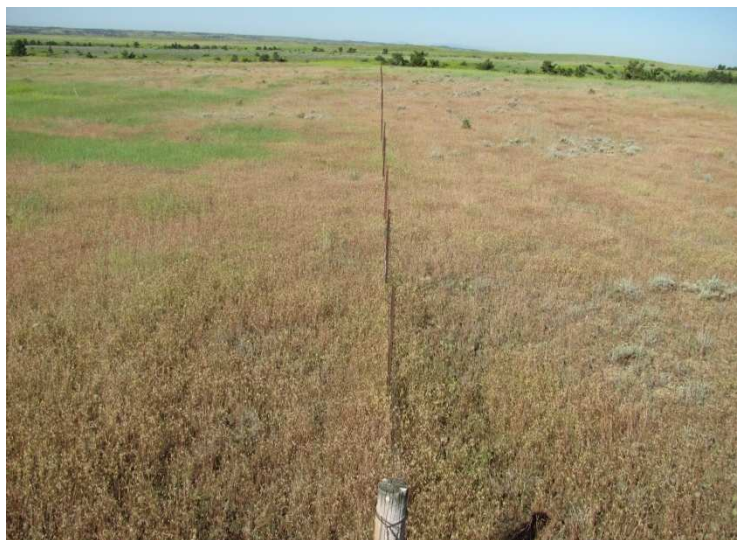
Opuntia Exclosure, Open Grazing on left, No Grazing on right. Photo taken 3 July 2011



Opuntia Exclosure, Wildlife Only on left, Open Grazing on right. Photo taken 7 December 1965



Opuntia Exclosure, Wildlife Only on left, Open Grazing on right. Photo taken 16 April 1985



Opuntia Exclosure, Wildlife Only on left, Open Grazing on right. Photo taken 3 July 2011



Opuntia Exclosure, Photo Point 1, East of exclosure. Photo taken 7 December 1965



Opuntia Exclosure, Photo Point 1, East of exclosure. Photo taken 3 July 2011



Opuntia Exclosure, Photo Point 2, inside No Grazing, looking toward ladder. Photo taken 8 September 1965



Opuntia Exclosure, Photo Point 2, inside No Grazing, looking toward ladder. Photo taken 3 July 2011



Opuntia Exclosure, Photo Point 3, inside No Grazing, looking toward ladder. Photo taken 12 July 1967



Opuntia Exclosure, Photo Point 3, inside No Grazing, looking toward ladder. Photo taken 3 July 2011



Opuntia Exclosure, Photo Point 4, inside Wildlife Only, looking toward ladder. Photo taken 12 July 1967



Opuntia Exclosure, Photo Point 4, inside Wildlife Only, looking toward ladder. Photo taken 3 July 2011



Opuntia Exclosure, Photo Point 5, inside Wildlife Only, looking East along fence line toward No Grazing. Photo taken 12 July 1967



Opuntia Exclosure, Photo Point 5, inside Wildlife Only, looking East along fence line toward No Grazing. Photo taken 3 July 2011



Opuntia Exclosure, Photo Point 6, inside Wildlife Only exclosure, looking toward ladder. Photo taken 8 September 1967. Note aircraft.



Opuntia Exclosure, Photo Point 6, inside Wildlife Only exclosure, looking toward ladder. Photo taken 3 July 2011

VITA

Ilana V. Bloom-Cornelius

Candidate for the Degree of

Master of Science

Thesis: VEGETATION RESPONSE TO FIRE AND DOMESTIC AND NATIVE
UNGULATE HERBIVORY IN WYOMING BIG SAGEBRUSH ECOSYSTEM

Major Field: Natural Resource Ecology and Management

Biographical:

Education: Graduated from Mendocino High School in Mendocino, CA in 2003; received a Bachelor of Science majoring in Animal Science from California Polytechnic State University, San Luis Obispo, CA in June 2008; completed the requirements for the Master of Science/Arts in Natural Resource Ecology and Management at Oklahoma State University, Stillwater, Oklahoma in December, 2011.

Experience: Worked as a livestock intern at California Polytechnic State University, Swanton Pacific Ranch, Davenport, CA 2007; worked as a Biological Technician for the USDA Forest Service in Big Piney, WY on the Bridger-Teton National Forest 2008-2009; worked as a Burned Area Emergency Response crew member on the New Fork Fire, Pinedale, WY 2008; conducted research on the effects of fire and domestic and native ungulate herbivory on the Charles M. Russell National Wildlife Refuge, Montana 2010-2011; conducted prescribed fires as part of the Oklahoma State University Prescribed Fire Crew 2010-2011 and with the Charles M. Russell National Wildlife Refuge Fire Crew 2010

Professional Memberships: The Society for Range Management

Name: Ilana V. Bloom-Cornelius

Date of Degree: December, 2011

Institution: Oklahoma State University

Location: Stillwater, Oklahoma

Title of Study: VEGETATION RESPONSE TO FIRE AND DOMESTIC AND NATIVE
UNGULATE HERBIVORY IN A WYOMING BIG SAGEBRUSH
ECOSYSTEM

Pages in Study: 108

Candidate for the Degree of Master of Science

Major Field: Natural Resource Ecology and Management

Scope and Method of Study:

Coupling of fire and grazing as a landscape management tool is a global phenomenon in many diverse landscapes. While grazing exclosure studies examining the effects of livestock on rangelands have provided insight into the effects of livestock grazing, wildlife and livestock have not commonly been studied in the presence of fire. Adding the influence of fire on herbivory and the effects of herbivory on fire make this a unique study. This study uses pre-established grazing exclosures on the Charles M. Russell National Wildlife Refuge in central Montana, USA that included three grazing treatments to exclude cattle and wildlife: No Grazing, Wildlife Only, and Open Grazing. A randomized complete block design with three by two factorial arrangement of grazing treatments split by fire was repeated at three sites. Vegetation cover data was collected using a modified Daubenmire method in 2010 and 2011 with percent area burned following fire. Forage quality and biomass data, herbivore attraction, and grasshopper and rabbit herbivory data were also collected.

Findings and Conclusions:

Large ungulate herbivory affected percent area burned ($p = 0.0255$), and crude protein increased and biomass decreased following burning ($p \leq 0.0008$). Fire decreased herbaceous and woody cover in burned treatments in the year following burning ($p \leq 0.0162$), though no differences were detected between Wildlife Only and Open Grazing treatments. Excluding big sagebrush, vegetation resprouted by one growing season post fire. Differences in percent area burned by grazing treatment suggest ungulate herbivory may be a driving factor in managing fine fuels in shrubland and grassland ecosystems and can potentially be important in managing fires to create a landscape mosaic. Increased forage quality immediately following fire, and the influence of grazing on fire behavior, suggests fire and grazing evolved as coupled disturbances. Dominance of sagebrush in a fire dependent community suggests historical fire return intervals may have been shorter and current levels of sagebrush were historically unsustainable. Additional studies on the interactions of native and domestic ungulates are recommended to understand potential competitive land use influences in combination with fire.

ADVISER'S APPROVAL: Dr. Samuel D. Fuhlendorf