

IMPACT OF WILDFIRE AND TARGETED GRAZING EFFECTS ON NORTHERN GREAT PLAINS
RANGE AND PASTURE LANDS

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Impact of Wildfire and Targeted Grazing Effects on Northern Great Plains
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ABSTRACT

The Grand River District in South Dakota has a high proportion of underutilized crested wheatgrass within native rangelands. Prescribed fire is rarely used for management; although, there is a strong evolutionary relationship with fire and grazing. Our project entailed two studies, the first looking at targeted grazing with cattle and second the effects of an April wildfire on plant species composition, basal cover, and phytomass production. The wildfire and targeted grazing did not affect ($p > 0.05$) plant species composition or phytomass production four and 16 months after treatment (MAT). Targeted grazing increased ($p \leq 0.05$) crested wheatgrass utilization, but did not reduce ($p > 0.05$) basal or canopy cover. Fire was ineffective ($p > 0.05$) at decreasing crested wheatgrass frequency. Fire increased ($p \leq 0.05$) bare ground and decreased ($p \leq 0.05$) litter cover four and 16 MAT. Multiple treatments may be more effective at controlling crested wheatgrass.

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DEDICATION

I dedicate this work to my brother, Chuck, and my grandpa, Charlie. No two people have encouraged me more to pursue academics and maintain a strong work ethic.

PREFACE

Chapter 2 and 3 are written as manuscripts to be submitted to a peer-reviewed journal. As of now, only Chapter 3, “Effects of early spring wildfire on crested wheatgrass invaded rangelands of the Northern Great Plains”, will be submitted to a journal. Both chapters are written following the style and formatting guidelines of *Rangeland Ecology and Management*.

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CHAPTER 1: INTRODUCTION AND LITERATURE REVIEW

Researchers imported crested wheatgrass (*Agropyron cristatum* [L.] Gaertn.) to the United States from Asia during the 1930's to establish permanent cover on marginal farmlands experiencing long periods of drought. It now occupies 6 - 11 million ha of pasturelands in North America (Hulet et al. 2010). Crested wheatgrass was planted because of its value as early spring forage, grazing resistance, and drought tolerance (Smoliak and Dormaar 1985). Although monocultures of crested wheatgrass are still productive, they lack biodiversity (Christian and Wilson 1999), create proper soil functioning issues (Liebig et al. 2006), and experience early senescence compared to native rangeland plants (Sedivec et al. 2010). Therefore, managers and producers are trying to balance crested wheatgrass plantings within rangelands by reducing the cover and survival of crested wheatgrass using tools such as prescribed fire, timing and intensity of grazing, disking, interseeding, and herbicides. Herbicides, plowing, and seeding are expensive and could cause other associated problems like erosion and weedy plant invasion (Sutter and Brigham 1998; Christian and Wilson 1999; Heidinga and Wilson 2002; Hulet et al. 2010; Fansler and Mangold 2011). A combination of techniques will be the key to reducing the cover of crested wheatgrass, preventing the spread of other introduced species, and increasing native plant cover (Hansen 2007).

Prescribed fire (Augustine and Milchunas 2009; Gittins et al. 2011; Vermeire et al. 2014) and grazing (Angell 1997; Wilson and Partel 2003; Vaness and Wilson 2008) are the most cost-effective ways of utilizing and controlling crested wheatgrass. Fire consumes standing dead vegetation often found in crested wheatgrass pastures and increases phytomass availability. Heavy grazing tramples non-grazed plants and increases grazing distribution. Both of these techniques can be used earlier in the season on crested wheatgrass invaded pastures compared to native rangelands without seriously impacting the native plant community. There are associated problems with heavy grazing like changes in species composition (Ingram et al. 2008; Vermeire et al. 2008) and compacted soils with low infiltration (Greenwood and McKenzie 2001), but others associated beneficial qualities with short-duration grazing, often involve hoof action (Manley et al. 1995). Prescribed fire is highly contested in the Northern Great Plains (NGP) due to societal concerns (White and Currie 1983) and inconsistent results on species compositional changes (Whisenant and Uresk 1989; Vermeire et al. 2011). The proper timing of fire and

grazing will be imperative to produce the most control and utilization of crested wheatgrass while encouraging the survival of native plant species. This timing will vary depending on region, climate, species composition, and desired results.

Published data is limited in the NGP studying prescribed fire effects on plant species composition, soil nutrients, phytomass production, and ground cover. The objective of this literature review is to 1) develop a timeline for the introduction and dispersal of crested wheatgrass; its uses, benefits, and negative impacts; and potential control methods; 2) determine positive and negative consequences of heavy stocking or targeted grazing on plant species, biomass, and soils; and 3) develop a timeline for the history and use of fire in the NGP and effects of fire on species composition, biomass, and soil.

Literature Review

History and Ecology of Crested Wheatgrass

Crested wheatgrass is an introduced perennial, cool-season, drought resistant, grazing tolerant bunchgrass found throughout the U.S. and Canada, originating from Asia (USDA-NRCS 2013). The first introduction to the U.S. occurred in 1898 in South Dakota, with a subsequent introduction to North Dakota in 1906. These seeds were obtained from the Valuiki Experiment Station in Russia. Early seedlings in the Dakotas were not as successful as the ones promoted in the 1930s (Rogler and Lorenz 1983).

Crested wheatgrass was planted extensively to control erosion that resulted from years of drought, poor farming practices, and abandoned farm fields. It was widely used because it established quickly, went dormant during droughts, survived winters, and provided early spring forage (Rogler and Lorenz 1983). Crested wheatgrass recovers from drought faster than native plant species, has increased longevity (Smoliak et al. 1967), persists when many native plants are unable to survive (Hulet et al. 2010), and produces more forage than healthy rangeland communities (Smoliak et al. 1967; Smoliak and Dormaar 1985). With earlier, increased forage outputs, livestock production increased on crested wheatgrass pasturelands compared to cattle grazing on native rangelands.

Crested wheatgrass possesses a variety of attributes that allow it to persist under a wide array of climatic conditions and disturbances. Survival increases and competition decreases as crested wheatgrass plants mature, creating large tussocks with wide interspaces between plants (Hyder and

Sneva 1963). Mortality in crested wheatgrass is very rare, but it is possible during periods of drought and inadequate soil moisture levels (McLean and Ryswyk 1973). Crested wheatgrass is a preferred forage for livestock and some wildlife in the spring and fall. It reaches peak productivity between the second and fourth growing season (Holechek 1981; USDA-NRCS 2013). Crested wheatgrass can be a very important grass in early grazing situations, but becomes undesirable later in the growing season due to numerous reproductive stems and culms (Hyder and Sneva 1963). Some ranchers prefer calving on crested wheatgrass pastures because it offers nutritious forage at an optimal time. On average, it will leaf out ten days ahead of northern plains bluegrass species (Kentucky bluegrass (*Poa prantensis* L.) and Canada bluegrass (*Poa compressa* L.)) and up to two weeks ahead of most native cool-season grass species; however, palatability decreases throughout the growing season, becoming worthless for winter forage (USDA-NRCS 2013).

Crested wheatgrass reproduces vegetatively and through seeds. Seeds remain near parent plants and offer the majority of the plant recruitment, rather than aggressively spreading by rooting material (Hulet et al. 2010; USDA-NRCS 2013). Crested wheatgrass stands maintain constant densities and are replaced by seeds established within the interspaces between plants (Looman and Heinrichs 1973). Seeds have the most germination success during the first year after production, but they remain viable up to five years in the soil (Holechek 1981; Marlette and Anderson 1986). After five years, there is a decrease in seed survivorship (USDA-NRCS 2013). Some varieties of crested wheatgrass are more likely to reproduce by with synchronous tillers. This growth occurs through stem development from the previous year's growth (Cook and Stoddart 1953). Tillers normally grow from lower axillary buds and can be found more frequently on damaged stems where reproductive stems have failed earlier in the year (Hyder and Sneva 1963). Both types of reproduction are used by crested wheatgrass to maintain its dominance in original plantings. These characteristics have led to 6 - 11 million ha of marginal farmland being planted with crested wheatgrass in North America (Hulet et al. 2010). Crested wheatgrass has proven to be a valuable plant species, but it has some detrimental impacts to long-term range health.

Long-Term Impacts of Crested Wheatgrass

Crested wheatgrass was planted on marginal lands that were used for annual crop farming practices for many years. These historic farming practices also contributed to the negative impacts seen on crested wheatgrass pastures, especially in regards to soil properties. Presumably, the historic farming practices lead to compacted soils, reduced infiltration rates, and removed organic matter. Some of the most common impacts of long-term crested wheatgrass pastures are exposed soil, increased erosion, higher bulk density, less soil organic matter, decreased pH, and less root matter compared to native rangelands (Smoliak and Dormaar 1985; Christian and Wilson 1999; Liebig et al. 2006). Many of these crested wheatgrass fields maintain row-crop soil characteristics that are exacerbated by the growth of crested wheatgrass plants (Lesica and DeLuca 1996).

Crested wheatgrass fields affect belowground nutrients by allocating more nutrients aboveground than native species. Crested wheatgrass also has higher carbon: nitrogen ratios than rangelands (Christian and Wilson 1999). For this reason, crested wheatgrass cannot maintain the “biogeochemical” quality that rangelands produce (Dormaar et al. 1995). Crested wheatgrass is highly competitive with other species because it decreases soil resource availability, emerges early, and is taller than most vegetation in the mixed-grass prairie, shading out other species. Species with similar life forms and season of growth may be particularly at risk, since crested wheatgrass outcompetes other cool-season plants (Heidinga and Wilson 2002). The space found between rows has less litter and creates an inhospitable environment for new seeds of other species (Marlette and Anderson 1986). Most cattle will avoid large bunches of crested wheatgrass, using the interspaces between plants, compacting the soil even more (Balph and Malechek 1985).

One of the largest concerns with crested wheatgrass pasturelands is a lack of both plant and animal species diversity (Christian and Wilson 1999; Heidinga and Wilson 2002; Hulet et al. 2010; Fansler and Mangold 2011). Research reports both positive and negative effects associated with wildlife, especially songbirds, and crested wheatgrass, as different species rely on different conditions for survival and reproduction. Crested wheatgrass stands do not offer as much structural diversity as native rangelands. There is much more bare ground due to tussock spacing, which may be related to bird

success or failure depending on the species (Sutter and Bringham 1998). One example of success is burrowing owls. They appear to utilize bunch interspaces for capturing prey (Restani et al. 2008).

Crested wheatgrass, in most instances, does not appear to be aggressive. It should be possible to have a combination of crested wheatgrass and native vegetation on rangelands. In some cases where crested wheatgrass appeared to be invasive, planted fields almost always act as the source population and spread via seed (Heidinga and Wilson 2002). Species richness and species turnover will be lower on crested wheatgrass fields. Although the biodiversity will be lower on crested wheatgrass pastures, it is speculated that it may be a better option compared to other introduced grasses like Kentucky bluegrass, cheatgrass (*Bromus tectorum* L.), or smooth brome (*Bromus inermis* Leyss.). Range management strategies should carefully consider a course of action which decreases the cover of crested wheatgrass while simultaneously sustaining or enhancing native plant species. Management should focused on containing crested wheatgrass pastures that border native rangelands to stop encroachment. Then it may be possible to consider options for diversifying or restoring entire pastures. Controlling disturbance is the most effective technique to decrease the likelihood of other invasive species moving into crested wheatgrass pastures.

Crested wheatgrass can have some beneficial purposes in rangeland ecology. It is highly useful in newly disturbed areas for suppressing weed invasion, reducing runoff, and creating high value forage for livestock (Rogler and Lorenz 1983; Pellant and Lysne 2005). Some have even proposed using crested wheatgrass to assist restoration of native habitats invaded by cheatgrass (Francis and Pyke 1996; Cox and Anderson 2004). Crested wheatgrass controls stands of cheatgrass by germinating even earlier and at colder temperatures (Cox and Anderson 2004; USDA-NRCS 2013). Crested wheatgrass will not always be able to compete with cheatgrass, but native seeds planted into crested wheatgrass are significantly more successful than native seeds planted into cheatgrass stands (Cox and Anderson 2004). A deliberately introduced species with some beneficial attributes can be considered problematic, mostly due to monocultures lacking species diversity and early senescence. The characteristics which made crested wheatgrass so successful in the early 1900s now contribute to current managers looking to control its spread and longevity. Crested wheatgrass pastures are rarely replaced, since they do offer

valuable forage in the spring, but concerns for biodiversity and habitat will be major factors for controlling stands.

Management Options

Management sought to strengthen crested wheatgrass pastures to increase production and improve livestock forage in the past (Hyder and Sneva 1963; Trlica and Cook 1971; Looman and Heinrichs 1973; Leyshon and Campbell 1992). These studies acknowledged the instability of crested wheatgrass pastures, and research has shown when to apply other management techniques to encourage crested wheatgrass. In contrast, these studies can be used in reverse to determine how to limit monocultures, since there is focus on improving native forb and grass diversity on areas previously planted to crested wheatgrass pastures.

The persistence of crested wheatgrass may be dependent on management techniques that reduce the cover and vigor of stands, water availability, and climate. Clipping or grazing may be most effective when plants are stressed from lack of water. Under water and defoliation stress, crested wheatgrass plants reduce the amount of seed production. Since most introduced species have higher seed production and increased seedling survival, controlling large tussocks using herbicides or multiple defoliations may reduce the production of seeds. Persistent management will be important, since mature plants did not see any reduction in tussock size and seed production until the second year of management. A non-selected herbicide (glyphosate) was effective in reducing crested wheatgrass in the short-term, but it should not be a long-term solution due to costs. Control will be extremely important during wet years when plants are likely to produce more seeds (Hansen and Wilson 2006).

Early spring grazing is another management option, although it might not be effective on large acreages (Angell 1997; Wilson and Partel 2003; Vaness and Wilson 2008). Grazing should be timed with phenological stages which reduce the vigor and diminish survival of mature plants. Hoof action at higher stocking rates should help reduce seedling survival in plant interspaces, and grazing during drought could be even more detrimental. Grazing should be at heavier stocking densities because light, short-duration and continuous grazing have the potential to increase tiller density depending on climatic conditions

(Angell 1997). Moderate to heavy short-duration grazing should be suitable for most areas. Exact stocking rates will depend on site conditions, range health, and soil moisture availability.

Previous research investigating the use of fire on crested wheatgrass has not shown any changes in cover. Crested wheatgrass is fire resistant with little to no damage on individual plants. The purpose of using prescribe fire on crested wheatgrass would be to increase palatability for grazing animals. Fire decreases the amount of standing dead (Augustine and Milchunas 2009; Vermeire et al. 2014), which decreases forage quality (Gittins et al. 2011), and it can temporarily improve the nutritional value of many forage species (Waterman and Vermeire 2011). Another reason to use fire in the NGP on crested wheatgrass is to draw cattle into unused pasturelands as fire resets the growth cycle and provides new green material (Fuhlendorf and Engle 2001).

Few studies focus on the impacts of fire on crested wheatgrass because of its tolerance to fire. However, the impact of fire on other large caespitose grasses can allow comparisons with the relative impact of fire. A study (Snyman 2003) in South Africa found that bunchgrasses experience more mortality than other grasses. Larger clumps have more fuel to burn, so plants experience more consumption and heat stress. These same patterns were shown in the NGP when large tussocks showed less regrowth and more damage than rhizomatous grasses to a spring wildfire (Antos et al. 1983). There was also reduced herbage production of *Agropyron* species on lowlands after a summer wildfire (Erichsen-Arychuck et al. 2002). In the Southern Great Plains, selected bunch grasses benefitted the most on non-burned treatments (Pfeiffer and Steuter 1994).

Secondary treatments will be critical in converting crested wheatgrass stands to native vegetation and diversifying pasturelands (Hansen 2007; Hulet et al. 2010). The seedbed will have to be cleared of crested wheatgrass to have successful native plant species recovery. Restoration efforts will be highly dependent on climatic factors (Bakker et al. 2003), since this is a main driver of species composition (Biondini et al. 1998). Reseeded natives may need assistance if drought occurs, as other invasive species will have an opening if crested wheatgrass is removed. If there is an extensive drought, residual plants will be valuable to block out other invasive plant species. Herbicide will be a necessary treatment to remove large plants (Hansen 2007).

The best management option for crested wheatgrass stands appears to be reducing the seedbank, reducing and removing most of the mature plants, providing suitable habitat for native plant species, and reseeding (Marlette and Anderson 1986). For native seedlings to be successful, they need available space and be able to survive in that space. The space between planted tussocks can offer available space, but some type of technique will be needed to loosen the soil, reduce bulk density, and create a better seedbed. The seeding composition should be a mixture of warm- and cool-season grasses and forbs. Since crested wheatgrass outcompetes similar species, it will be beneficial to use warm-season plants like blue grama (*Bouteloua gracilis* [Willd. Ex Kunth] Lag. Ex Griffiths) (Heidinga and Wilson 2002; Wilson and Partel 2003) to avoid competition. However, one study found green needlegrass (*Nassella viridula* [Trin.] Barkworth), a cool-season bunch grass, to be highly successful in competing with crested wheatgrass (McWilliams and Van Cleave 1960). Rapidly established plants such as bottlebrush squirreltail (*Elymus elymoides* [Raf.] Swezey subsp. *californicus* [J.G. Sm.] Barkworth) would also be competitive with crested wheatgrass (Gunell et al. 2010), while others suggest using cryptogams and dicots (Christian and Wilson 1999). Small-scale, pasture treatments may not be as effective as large-scale, landscape treatments (Krueger-Mangold et al. 2006), but heavy grazing could provide a cost-effective way to potentially reduce the cover of crested wheatgrass. One study found suppression techniques not helpful in diversifying monocultures, but this study only used mechanical disking and herbicide (Fansler and Mangold 2011).

Targeted Grazing

Targeted grazing aims to improve resources, mainly vegetation, by using livestock or wildlife as a tool for management. Targeted grazing accomplishes an established goal by manipulating how animals graze with respect to time, location, and intensity. This is different from most management practices, since traditional management aims to increase livestock production. Targeted grazing is growing in popularity and utility. A manual was developed to define targeted grazing, present benefits, and help managers and landowners successfully implement this tool on a mixture of landscapes (Launchbaugh et al. 2006).

Targeted grazing, also referred to as prescribed grazing, focuses on control over forage availability, stocking rate, and grazing time. Complete or targeted biomass removal early in the season can reduce invasive cover and increase native species. These changes are not immediate and happen as invasive plants are continually defoliated and stressed for carbon reserves. In contrast, grazing late in the season can cause invasive species to increase and native species to decrease (Rinella and Hileman 2009).

Targeted grazing has been used successfully on various forbs throughout the western United States, including leafy spurge (*Euphorbia esula* L.), spotted knapweed (*Centaurea stoebe* L. ssp. *micranthos* [Gugler] Hayek), and yellow starthistle (*Centaurea solstitialis* L.) (Popay and Field 1996; Launchbaugh et al. 2006; Wallace et al. 2008). Targeted grazing usually involves sheep and goats consuming forage not utilized by cattle, but cattle can also be effective in some circumstances. Sheep consume a broad mixture of plant types (forbs and graminoids) and are commonly used because they eat a wider range of forages compared to cattle (Beck and Peek 2005). Goats can also be used if landscapes are dominated by shrubs, since they will consume more shrubs than either sheep or cattle (Squires 1982).

Targeted grazing is rarely used on graminoid species, since these are freely grazed and willingly consumed by cattle and sheep. However, targeted grazing has been recently utilized on some introduced graminoids. In these cases, management strategies could increase the stocking rate to eliminate foraging choices. When a grass species dominates a landscape, cattle have no choice but to consume that species, especially if they are enclosed on areas with high coverage. Targeted grazing by cattle has been used on cheatgrass to reduce the fire load and decrease fire risks (Diamond et al. 2009).

Targeted Grazing of Crested Wheatgrass

Stocking rates, not grazing system, play the largest role in determining individual animal gains. With increasing stock density, individual animal gains will go down, but the gains per unit land area will increase. The efficiency of harvest will also increase but solar capture will decrease with increasing stocking densities (Hart et al. 1988; Derner et al. 2008; Briske et al. 2008). At high stocking rates, there may be some rangeland deterioration, so 50 percent removal is widely recommended (Biondini et al.

1998). A problem commonly faced on public grasslands is overuse of native rangelands and low to no use of crested wheatgrass. Cattle overuse native plants adjacent to crested wheatgrass pastures because crested wheatgrass cures early. Cattle are normally turned out to pasture too late on intermixed native and introduced grass pastures, primarily because native plants have not produced sufficient biomass during the early season and have a slower growth than introduced cool-season grasses (Campbell 1952; Sedivec et al. 2010).

Various studies have looked at the grazing impacts to rangeland health, crested wheatgrass cover, and animal performance on crested wheatgrass pastures (Frischknecht et al. 1953; Willms et al. 1990; Vaness and Wilson 2008). Many of these studies increased the stocking rate or used short-duration grazing in an attempt to utilize 70–80 percent of the crested wheatgrass to eventually decrease the cover of crested wheatgrass and increase grazing distribution. Cultivation and fire have been used to draw herbivores to crested wheatgrass plantings found within rangelands. Sheep increased grazing on crested wheatgrass pastures burned in the fall. Grazing and fire reduced the frequency of crested wheatgrass, but only cultivation reduced the basal cover (Lodge 1960). Grazing alone cannot reduce the amount of crested wheatgrass. Fall grazing can keep crested wheatgrass from increasing more compared to spring grazing and spring and fall grazing (Currie 1970).

Grazing can also be used to decrease the number of residual plants with standing dead accumulating within bunches, also known as wolf plants. As few as two cured stems can affect cattle preference, since these tussocks will be avoided (Ganskopp et al. 1993). Light grazing crested wheatgrass increased the occurrence of wolf plants and encourage patchy grazing distributions in following seasons (Frischknecht et al. 1953). High stocking rates reduces livestock choice and increases trampling which prevents standing dead from accumulating (Balph and Malechek 1985).

Grazing crested wheatgrass at the right time can also help prevent spreading. Crested wheatgrass should be grazed during wet years, since it will put more energy into reproductive shoots. Grazing will help decrease the amount of new seed added to the seedbed. There can also be increases in soil water depletions when crested wheatgrass plants are grazed early in the season in either a wet or dry year (Miller et al. 1990).

Targeted grazing of crested wheatgrass in the spring benefits native rangelands by keeping cattle from grazing these areas for approximately one month (Frischknecht et al. 1953). Keeping cattle off of native range in the spring is very advantageous for the NGP and is often called “delayed use spring grazing systems”. In this type of deferment, native rangelands are allowed to produce more forage before herbivores are allowed to graze (Lodge 1970). This type of grazing is also called complimentary grazing (Smoliak 1968; Hart et al. 1988). There are associated costs of heavy grazing during targeted grazing, but longer rest periods and supplements can help mitigate these costs. In general, crested wheatgrass was not reduced by grazing alone. Targeted grazing can help reduce standing dead, increase quality, increase trampling, and improve livestock gains (Currie 1970; Olson and Malechek 1988).

Heavy Grazing Effects

Management goals, inputs, and desired outputs for grazing rangelands and pasturelands dictate grazing systems and intensity. There are several important issues when considering different grazing management strategies. Rotational grazing may benefit producers and allow them to use higher stocking rates, but most weight should be placed on climate, specifically precipitation (Biondini and Manske 1996). Research has pointed out the similarities between season long grazing and rotational grazing; there are not any differences between season long grazing and rotational grazing at moderate levels of defoliation in regards to species composition, production, and utilization (Hart et al. 1993; Derner and Hart 2007; Derner et al. 2008; Briske et al. 2008). For the purpose of this project, focus will be placed on the effects of heavy grazing, including studies with heavy long- and short-term use in the northern mixed-grass prairie.

Biomass. There is much controversy over the role of grazing and plant production. The Grazing Optimization Hypothesis predicts a certain level of grazing severity, potentially with a specific season of grazing, is able to increase the production of an individual plant. Grazing increases plant production in some cases (McNaughton 1979), but other research shows plant production increases with rest (Lacey and Van Poollen 1981). Others argue that plant responses to grazing are dependent on other factors, and defoliation leads to under-compensation, overcompensation, or compensation (Hilbert et al. 1981; Williamson et al. 1989; Belsky 1993). High amounts of natural variability, grazing severity, timing,

adjacent plants, species composition, competition, nutrient availability, and climate all play a large role in the amount of biomass produced in a year. Quality and quantity of forage will also change based on these factors. These differences make strict definitions of annual net primary productivity, standing biomass, and production necessary (Belsky 1993).

Standing crop was linked to precipitation and stocking rate, not grazing system (Hart et al. 1993; Derner and Hart 2007; Vermeire et al. 2008). Drought reduces the amount of annual net primary production (Biondini and Manske 1996), and heavy grazing can decrease the height of standing crop and reduce both basal litter and standing dead (Naeth et al. 1991). Peak standing biomass will decrease (Schuman et al. 1999; Derner and Hart 2007; Vermeire et al. 2008) and utilization will increase (Hart et al. 1993) with increasing stocking rate. If stocking rate was decreased, peak standing biomass would increase and utilization would decrease. For belowground biomass, grazed pastures compared to exclosures had less root biomass. However, heavy and light continuous grazing showed similar root biomass results (Schuman et al. 1999). Grazing reduced the amount of available energy to the root system. Without a continuous supply of carbohydrates, root respiration decreases, the entire system cannot be maintained, and the plant experiences root mortality (Briske and Heitschmidt 1991).

Species Composition. Species composition can be modified with early spring grazing and changing stocking rates (Vermeire et al. 2008), but species composition is more impacted by drought and grazing pressure than grazing systems (Biondini and Manske 1996). Most studies focus on blue grama and western wheatgrass when comparing composition (Whisenant and Uresk 1989; Hart et al. 1993; Biondini and Manske 1996). There are similar results in regards to specific species like western wheatgrass and blue grama (Hart et al. 1993). In some cases, blue grama basal cover increased (Biondini and Manske 1996; Schuman et al. 1999) due to competitor release and lower grazing stress (Hart et al. 1993). Another study found blue grama to decrease (Biondini et al. 1998); this is only expected when grazing pressures are exceptionally high (Hart et al. 1993). Species composition shifted in a 12 yr study when western wheatgrass decreased on non-grazed treatments. Western wheatgrass became the dominant species in continuous, moderately grazed plots (Schuman et al. 1999).

Forbs have been found to increase under heavy grazing (Biondini and Manske 1996; Vermeire et al. 2008) and drought (Biondini and Manske 1996), but forbs can also increase under grazing release

(Schuman et al. 1999). Other species that are not normally reported also experience changes from heavy grazing. Grazing after May produced pastures with fewer cactus and lower standing crop of annual cool-season grasses. Even after several very dry years, functional groups did not change significantly due to stocking rate or season of grazing.

In general, grazing in the mixed-grass prairie encourages warm-seasons to become dominant. These same results can be seen under drought conditions, but sedge species, a cool-season graminoid, tend to remain stable instead of declining (Biondini and Manske 1996). Warm-season grasses increase on non-grazed sites and cool-season grasses decrease (Ingram et al. 2008). Cool-season grasses are maintained with light stocking rates and early spring precipitation (Derner and Hart 2007). In the tallgrass prairie, heavy clipping increased native warm-season grass cover and reduced both introduced and native cool-season grass cover (Smart et al. 2013). Heavy grazing can lead to a decrease in live vegetative basal cover (Naeth et al. 1991), which can increase bare ground and potential for soil health problems. Biomass and species composition are one of the largest concerns when discussing heavy grazing, but attention should additionally be spent on belowground properties which contribute to aboveground characteristics.

Soil Properties. Research found grazing to both help and hinder soil health and nutrient cycling (Dormaar et al. 1988; Manley et al. 1995; Ingram et al. 2008). Most of the positive effects have been from light to moderate grazing and include: increased infiltration, increased soil organic matter, and increased soil nutrient cycling. As stocking rates increase, these soil health qualities diminish (Biondini et al. 1998). Without a strong, stable soil, severe consequences can be seen on rangelands and pasturelands including: loss of production, decreased water infiltration, increased run-off, decreased nutrient cycling of carbon and nitrogen, increased bulk densities, and decreased soil microbial activity. Moderate hoof action can help promote soil health by increasing infiltration and promoting microbial activity (Greenwood and McKenzie 2001). Some have argued that associated rests with short, heavily grazed pastures, not hoof action, may also be responsible for increased hydrologic function (Warren et al. 1986). Non-grazed soils had more moisture and were darker in color (Dormaar et al. 1988). Most of the soil responses to grazing take place in the upper portion of the profile. Like species composition, effects from grazing may be

dependent on climate, since soils are at higher risk of degradation when they are saturated (Greenwood and McKenzie 2001).

Carbon and nitrogen can be tied up in biomass not in contact with the soil (Biondini and Manske 1996), and grazing often leads to trampling that facilitates more contact between litter, soil surface, and soil microbes to increase soil organic carbon and nitrogen (Manley et al. 1995). Litter is broken down faster when in contact with soil, but this does not always mean the soil organic matter will increase (Naeth et al. 1991). It might not always increase litter incorporation and result in less soil organic matter and lower carbon and nitrogen on grazed soils. (Dormaar et al. 1988). Although some found no differences between treatments of continuous or rotational grazing and nutrient amounts (Manley et al. 1995; Biondini et al. 1998), others found differences between low continuous, high continuous, and no-grazing treatments. The most soil organic carbon and nitrogen were found on low continuous grazing treatments. Trends showed more nitrogen loss on grazed sites, and increased nitrogen mineralization on non-grazed treatments (Ingram et al. 2008). This was also true for crested wheatgrass pastures. Nitrogen decreased when stocking rates doubled, but phosphorus did increase (Willms et al. 1990).

Bulk density generally increases with grazing (Dormaar et al. 1988). Two studies (Abdel-Magrib et al. 1987; Mapfumo et al. 1999) found bulk density to increase during the fall after heavy grazing, but the freeze-thaw process, along with biotic activity, decreased bulk density by spring. Infiltration rates, commonly associated with bulk density, were higher on continuous grazing treatments compared to heavy, short-term grazing, but this relationship can change between years. Infiltration rates are also driven by yearly changes (Abdel-Magrib et al. 1987). Climate, soil type, soil texture (Van Haveren 1980), and stocking rate are most likely responsible for differences seen in studies across various landscapes under different stocking rates and grazing systems.

History of Fire in the Northern Great Plains

The historical fire regime in the NGP has been established using historical accounts and carbon dating of core sediments and soils. There are several reviews that show sufficient support for previous fire history (Sauer 1950; Higgins 1984; Axelrod 1985; Umbanhowar 1996; Brown 2005; and Anderson 2006). There does not seem to be any strict pattern or interval, but fires are both lightning-caused and man-

caused in the NGP. There can be differences in fire habit from climatic factors, plant species composition, topography, and fuel loads (Brooks et al. 2004) to affect changes in plant diversity, biomass production, habitat availability, and soil nutrients.

Grasslands became dominant during the end of the Cenozoic (Retallack 2001). The climate during the Pleistocene (2.6-.01 million years ago) provided the changes needed to promote grasslands, fire, and herbivores (Sauer 1950). During the middle Miocene (23-5.3 million years ago) to early Pliocene (5.3-2.6 million years ago) the central grasslands of North America was highly dominated by woody vegetation. Aridity increased during the Miocene and Pliocene, but moist areas continued to support trees and shrubs. Grasslands started to spread from areas with less woody canopy cover, and they started to rapidly expand 7-5 million years ago. They were adapted to high temperatures and low moisture levels compared to woody species. Climate, along with man-made and natural fires, was highly responsible for the continued success of grass species, and large herbivores helped to promote their continued expansion by trampling seedlings and pushing older trees down (Axelrod 1985). Tree species also declined when humans used remaining areas of woody vegetation for permanent structures. Grasslands are still very young in geologic time compared to other plant communities, and many species can be found in both grass- and woodland habitats (Retallack 2001).

Fires have been present in the NGP for at least 4 500 years (Brown et al. 2005). The fire frequency was formulated by looking at soil and lake cores. Soil horizons corresponding to the last 5 000 years in Manitoba show fire residue. There was an extreme decrease in the amount of fire residue in the upper soil horizons that paralleled with European settlement. After settlement, the frequency of fire dramatically decreased due to fire suppression techniques and land fragmentation. The soil profile that corresponded with 2 500 years ago saw much higher fire residue. This correlates with the presence of woodland tribes in the area and promotes the deliberate use of fire by Indigenous Peoples (Boyd 2002). Lake sediment cores from North Dakota, South Dakota, and Montana showed similar results (Umbanhowar 1996). There was a quick decline in the amount of charcoal sediments in lake cores after settlement, but there was a varying amount of charcoal sediments leading up to settlement. There appears to be a sharp increase in charcoal sediments directly before European settlement. This was attributed to trains throwing off sparks, as well as patchy European colonization. Suppression tactics were

not immediately implemented with settlement, since some settlers used fire to clear farmland and homestead areas (Sauer 1950).

Indigenous tribes across North America used fire for various applications, but none were found to use fire suppression techniques (Higgins 1984). Fire was commonly used to increase quality of forage production in order to manipulate prey species' patterns and distributions. Freshly burned areas would draw in large herbivores, and active fires could be used to move prey into optimal hunting areas. It was also used to change or maintain the plant community. Some tribes used fire to reduce fuel loads and maintain travel routes, while others used fire as a warfare tactic (Levy 2005; Diekmann et al. 2007). Californian tribes, like the Pomo, have a well-documented use of fire to create specific vegetative growth forms used for tools and decorative materials like baskets (Anderson 1999). Colonization of the western United States drastically decreased the ability of American Indians to use fire (Higgins et al. 1986).

Grassland fires are expected in areas that have adequate ground cover and a period of time when the cover dries (Sauer 1950). Even with fragmentation and human suppression, lightning-caused fires still occur in the NGP. Lightning caused fires were monitored from 1940 to 1981. On average, 24.7 fires occurred per year per 10 000 km² on western grasslands of North and South Dakota. Fires were common from April to September, but a majority took place in July and August. Fire frequency and size peaked in July corresponding to the highest thunderstorm activity, average temperatures, and dry fine fuels. Fires happening from October to March were probably caused by another catalyst besides lightning, most likely human (Higgins 1984). As of 2014, Moderate-resolution Imaging Spectroradiometer (MODIS) found most of the fires occurring in the NGP to happen from August to September. Fires taking place outside of this range influenced much smaller areas (Smith and McDermid 2014).

Currently, fire is limited by habitat fragmentation including roads, cultivated fields, heavy grazing, suppression techniques, and other developments and infrastructure (Higgins 1984; Higgins et al. 1986; Anderson 2006). The loss in connectivity makes it harder for prescribed fires to carry over larger landscapes. Fire regimes are variable in the NGP (Umbanhowar 1996; Brown et al. 2005), but they did occur with some frequency. Naturally occurring fire normally changes with climate and can have a direct relationship with the amount of precipitation. Increased precipitation can support larger fuel loads and more fires; decreased precipitation reduces the fuel load and diminishes the chances of fire (Brown et al.

2005). The climate must have enough precipitation to produce litter but not such an excessive amount that encourages rapid rates of decomposition. The same climate must also have periods of aridity to dry fuels and expose them to combustion (Boerner 1982). Compositional changes of cool- and warm-season species can also cause changes to the fuel load (Umbanhowar 2004). The NGP can see different changes from fire because the plant community is dominated by cool-season grasses (Steuter 1987); therefore, the timing of fire can be critical in this region (Redmann et al. 1993). Effects from both prescribed fires and wildfires have been documented extensively, but only several (Biondini et al. 1989; Vermeire et al. 2014) discuss fire effects in the NGP.

Effects of Prescribed Burning and Wildfires

The grasslands found in the NGP would not be present without fire associated with dry, hot climates, as areas with enough moisture would have sustained trees and shrubs. Fire is partially controlled by grazers and climate (Anderson 2006), but fire also produces unique changes to plant composition and community functions. Fire is used for a host of reasons, including: increasing phytomass yields, forage quality, forage availability, and utilization. It can also improve wildlife habitat for some animals by decreasing the cover of undesirable species, clearing habitat, and increasing the amount of palatable plants. Fire can also have positive effects on the seedbed by releasing nutrients and clearing excessive litter build-up. However, there are some negative impacts on plant communities and soils from fire. Notably, prescribed fires should never be conducted during a drought. Plant stress will inevitably increase if stress is placed on an already strained plant. Therefore, fires should be conducted during the dormant season or when soils are moist to saturated (Wright 1974).

Many of these benefits are still used today for the justification of prescribed fire. Burning is still used as a means of attracting large herbivores to certain areas (Ericksen-Arychuk et al. 2002; Vermeire et al. 2004). Fire and grazing can help control herbivore movements and grazing areas without additional fencing and restore historical disturbance regimes (Augustine and Milchunas 2009) with inexpensive management and labor (Vermeire et al. 2004). Fire, and additional grazing, can help manage wildfire and decrease undesirable species (Augustine and Milchunas 2009). One of the largest problems with using prescribed fire is societal perceptions of fire (White and Currie 1983). It is seen as a dangerous tool

without many benefits because prescribed fires and wildfires can have varying results that are dependent upon climatic factors in the NGP (Wright 1974; Gartner et al. 1986; Biondini et al. 1989; Whisenant and Uresk 1989; White and Loftin 2000; Vermeire and Rinella 2009; Vermeire et al. 2011; Vermeire et al. 2014).

Biomass. The amount of biomass, phytomass, standing dead, litter, and annual production all can be affected by fire. Spring and fall burns slow the early growth of graminoids in the growing season following fire in Saskatchewan (Redmann et al, 1993). They also found spring burns to have a greater effect than fall burns on these changes in growth, but fall burns saw the highest reduction in graminoid phytomass production. During a summer burn in Montana, standing crop was reduced 60 percent during drought but was unaffected when the growing season started with a wet spring, but the biomass between treatments was fairly similar (Vermeire et al. 2011). They did not find the above- or below-ground productivity to be reduced by their treatment. There were some differences seen in the standing crop of cool-season grasses based on water conditions, but the productivity was unaffected by this characteristic. Dissimilarly, summer fires in South Dakota produced the least amount of annual phytomass compared to spring and fall burns (Steuter 1987). Responses can also be dependent on site conditions. Standing crop was higher in wet areas, but there were no significant differences between burned and control sites in regard to standing crop on wet or dry sites, and biomass was only unaffected by treatment on wet sites in Manitoba (Shay et al. 2001). One of the first studies done on fire effects in the NGP in North Dakota found phytomass production to be significantly less on late May burned plots (Dix 1960). In South Dakota, fire did not increase or improve production because it consumed early growth during a spring burn (Engle and Bultsma 1984; Gartner et al. 1986).

Production response to fire may be more subtle and only distinguished with more sampling (White and Currie 1983). A study in Montana looked at fire effects on different plant communities (blue grama, western wheatgrass, and threadleaf sedge) in combination with clipping. The results showed many different patterns because biweekly vegetation sampling was used compared to only one sampling. Spring burning was useful for increasing the production of western wheatgrass and blue grama up until mid- and late June. During the mid-June sampling, burned treatments had significantly more phytomass than the control, but results varied for sampling dates after June. Fall burns stimulated production slightly

below that of the spring burns. Threadleaf sedge production was unchanged by spring burning and reduced by fall burning, but all the treatments produced more after being burned than the control plots (White and Currie 1983).

Some production loss has been connected to litter decreases following a fire. Annual litter removal can be very detrimental; although, some litter removal may not affect overall production. Characteristics of plant growth can be affected including height, tiller weight, and individual species phytomass production (Willms et al. 1993; Erichsen-Arychuk et al. 2002). Litter decreases, standing dead decreases, and bare ground increases after a prescribed fire or wildfire in the NGP (White and Currie 1983; Engle and Bultsma 1984; Redmann et al. 1993; Erichsen-Arychuk 2002; Shay et al. 2001; Grant et al. 2010; Vermeire et al. 2011; Vermeire et al. 2014). Many studies talk about litter changes because it plays an important role in plant-soil-water interactions.

A review on plant litter interactions with plant and soil communities found that litter reduces light infiltration. Shading can be harmful or helpful to seeds and seedlings depending on the species. By blocking the sunlight, the litter acts as insulation and keeps the ground temperature more constant than bare ground, reducing “thermal amplitude” (Facelli and Pickett 1991). The right amount of litter and stable surface temperature help reduce the amount of soil water evaporation and diffusion (Wright 1974; Facelli and Pickett 1991; Willms et al. 1993; Shay et al. 2001). Without litter, bare ground heats up faster and reduces soil water content. Disadvantages of losing litter be drought or are accentuated during a drought (Engle and Bultsma 1984; Erichsen-Arychuk et al. 2002). If drought is extremely severe or precipitation is very high, litter may be ineffective at producing beneficial results (Willms et al. 1993). Too much litter can also intercept all of the precipitation and eliminate infiltration into the soil causing plant stress (Facelli and Pickett 1991). In one case, there were no changes in soil moisture or temperature following litter removal from a summer fire (Vermeire et al. 2011). This is likely to occur in some cases, since not all of the litter is consumed during a fire (Redmann et al. 1993). Some litter may remain that prevents large amounts of water and wind erosion (Dix 1960).

The loss of litter and standing dead is not only detrimental to plant species, but it can also affect some bird species that nest in high structured areas or build their nests from litter and standing dead. Fire can drastically change the residual vegetation and structure, and bird species richness was found to be

lowered after a fire in North Dakota (Grant et al. 2010). Effects may be less severe if grassland birds use more standing dead compared to litter, since standing dead will recover faster than litter (Redmann et al. 1993). The loss of litter and shorter vegetation height associated with burns can also be beneficial for grassland birds that need short structure (Fuhlendorf and Engle 2001). A matrix of different species and structural heights is highly encouraged to improve habitat. The changes in biomass, litter, and basal cover can lead to the changes seen in species composition.

Species Composition. Fire can have positive, negative, or neutral effects on plant groups and species. The climatic conditions of the NGP play into the effects seen from a prescribed fire or wildfire, and all of the following studies described below occurred in Montana, North Dakota, South Dakota, Alberta, Manitoba, and Saskatchewan. Fire effects can be directly related to different species or have more general applications to functional/structural groups and physiological groups. Some of the most common groups, growth forms, and species will be examined to discover overarching effects from fire in the NGP.

Studies in the past have found warm-season grasses to respond well to spring fires and cool-season grasses to fall fires, since the respective species are mostly dormant during burning (White and Currie 1983; Redmann et al. 1993; Howe 1995; Shay et al. 2001). Spring fires are meant to improve range condition when sites are dominated by cool-season grasses to increase forage availability (Gartner et al. 1986). The productivity of warm-season plants increased after a spring fire (Whisenant and Uresk 1990), but a summer fire did not significantly change the composition of warm- and cool-season grasses (Vermeire et al. 2011). Fire did not affect warm- and cool-season ratios on sandy sites, but shorter fire return intervals on loamy and blowout sites had increased warm-season plants (Smith and McDermid 2014).

Spring fires will not increase the amount and cover of warm-season grasses if they do not respond quickly to increased light availability and warmer conditions facilitated by more bare ground. Overall, the cool- to warm-season ratio will remain similar between treatments and the control. Warm-season grasses are not as well-adapted to fire in the NGP compared to cool-season grasses, but production is unaffected. The current ratio of cool- to warm-season grasses observed is from a long

evolutionary history, not from short-term responses to fire and other disturbances provided in experiments (Steuter 1987).

Species composition can also be addressed in terms of diversity and plant community composition. Higher alpha and beta diversities were found on unburned and summer burn treatments compared to fall and spring burns. Landscape mosaic diversity was highest on fall burned treatments (Biondini et al. 1989). Native plant community composition did not change following a fire, but there were decreases in native grasses and forbs when the fire return interval was higher than four years (Grant et al. 2010). Species compositional changes may lag behind fire disturbance (Dix 1960), or it may change directly (Antos et al. 1983).

Forbs also show varying responses to fire. Depending on species, forbs will decrease (Biondini et al. 1989; Shay et al. 2001), increase (Antos et al. 1983; Biondini et al. 1989), or remain neutral (Erichsen-Arychuk et al. 2002). Forb responses can vary due to fire timing, since more forbs were found on spring and fall burns compared to summer burns. Environmental interactions could also be causing changes (Biondini et al. 1989). Forb species will have discrete responses to fire (Antos et al. 1983). Most trends appear to be upward, but other forbs can decline or remain stagnant.

Specifically, western wheatgrass showed stimulated production following a spring burn, and the cover increased from fall burns (White and Currie 1983). In some cases, western wheatgrass decreased after fire (Antos et al. 1983; White and Currie 1983; Whisenant and Uresk 1989), but its tiller density and standing biomass increased in the year following a fire (Whisenant and Uresk 1990). Weight was significantly greater but heights were much shorter following a burn (Gartner et al. 1986). Western wheatgrass had similar covers between burned and non-burned treatments during a drought. There was an increase of cover of western wheatgrass on burned sites with a wet spring. This allowed western wheatgrass to become the dominant species onsite (Vermeire et al. 2011). Western wheatgrass showed rapid recovery following fire (Dix 1960).

Blue grama production was encouraged by a spring burn, unlike fall burns. During most spring burns, blue grama is dormant during the disturbance. It is not expected to be affected by spring burns, but one study showed yields were slightly higher than the control (White and Currie 1983). Others have shown reduced blue grama on burned plots (Whisenant and Uresk 1989). Significant increases occurred

in blue grama after two and three years of burning on wet and dry sites (Shay et al. 2001), but summer wildfires had neutral effects on blue grama cover (Erichsen-Arychuk et al. 2002).

Threadleaf sedge and other sedges have had multiple responses to fire. Production has been unaffected (Whisenant and Uresk 1990), increased cover (White and Currie 1983), and decreased (Vermeire et al. 2011) following fire. These differences may be due to precipitation. Threadleaf sedge production increased on burned sites with increased precipitation, but production was reduced when precipitation was below normal (Whisenant and Uresk 1989). Sedges were also affected by the fire interval. Sedges decreased after two and three years of spring burning on dry sites but increased on wet sites (Shay et al. 2001).

Green needlegrass and needle-and-thread were both negatively affected by fire, especially during a drought. They can be adversely impacted if they are actively growing during a fire (Whisenant and Uresk 1989; Whisenant and Uresk 1990). Needle-and-thread can recover quickly after a fire (Dix 1960) or even increase after a fire (Garnter et al. 1986), but it has been found to show decreasing trends and less total standing crop even when it was not actively growing (Vermeire et al. 2011). Green needlegrass has only been found to decrease following a fire (Engle and Bultsma 1984; Gartner et al. 1986).

Fire is also commonly used to control invasive and introduced species. Controlling cheatgrass and field brome (*Bromus arvensis* L.) are effectively controlled by fall burns (White and Currie 1983). Excessive litter can help the proliferation of field brome, so it is important to clear litter on infested sites. After burning, the tiller density and standing crop of field brome was reduced, and the fire also helped to kill field brome seedlings. This result was apparent for one year (Whisenant and Uresk 1990). Leafy spurge, field brome, spotted knapweed, and Russian knapweed could all be reduced with fire at any fuel load level. Fire promotes leafy spurge establishment at any fuel load (Vermeire et al. 2009). Fire reduced nonnative annual grasses (Vermeire et al. 2011). Kentucky bluegrass was reduced after fire that was either conducted during a drought (Engle and Bultsma 1984) or followed by dry conditions (Gartner et al. 1986), but Kentucky bluegrass may increase after fire (Grant et al. 2010).

There is a pattern of fire having multiple effects on plants species composition, cover, biomass, and site diversity. In some cases, forbs and shrub biomass are unaffected by spring and fall fires (Redmann et al. 1993), but they can also decline in relatively similar ecosystems (Vermeire et al. 2011).

There is more to be discovered on the timing and climatic conditions that influence the changes in species composition. Plant species' responses to fire vary based on their phenology and growth form (Antos et al. 1983).

Soil Properties. Many aboveground processes and effects can be linked to edaphic characteristics. Most studies focus on soil moisture and soil nutrients, since these are vital for plant growth, reproduction, and recovery. Several studies have looked at the effects of fire on soil properties in the NGP.

Fire has the potential to increase the drought stress of a plant by reducing the soil moisture level. Drought stress can be increased during an ongoing drought if a fire occurs, since soil moisture was significantly lower on burned plots (Wright 1974; Antos et al. 1983; Snyman 2003; Vermeire et al. 2011). Less litter cover can cause soil and surface temperatures to increase, along with evaporation rates. This decreases the amount of available soil moisture for plants and other organisms (Antos et al. 1983; Willms et al. 1993; Shay et al. 2001). Water is one of the most limiting factors for plant growth and reproduction (Chapin et al. 1987; Reece et al. 2007), but other vital components are nitrogen, carbon, and other essential macro- and micronutrients.

Many of the nutrients used by plants are obtained from decomposing litter (Facelli and Pickett 1991), but litter can also hold nutrients and make them unobtainable without proper microorganism activity (Wright 1974). Fire can help increase microorganism activity by heating soils (Higgins et al 1986). Prescribed fire and wildfires consume some litter, and this consumption can either reduce the amount of nutrient availability or release nutrients being tied up in litter. Heat from fire can volatilize some nutrients like oxygen, carbon, hydrogen (Raison 1979), sulfur, and phosphorus (Raison 1979; Boerner 1982), but volatilization will all be dependent on the intensity, severity, duration, and other conditions of fire (Raison 1979; Higgins et al. 1986). Nitrogen concentrations were higher, but not significantly higher, on burned plots compared to non-burned plots (Redmann et al. 1993). In a review, most studies found total soil nitrogen pools unaffected by fire. Looking at various forms of nitrogen did show more patterns caused by fire in surface layers. Changes in various forms of nitrogen did not occur as the depth from surface increased. Ammonium (NH_4^+) and nitrate (NO_3^-) both increased following fire. Ammonium increased immediately after fire and had a higher peak with wildfire, 1 071 percent increase compared to 125

percent increase. Nitrate lagged behind the peak ammonium peak by 7-12 months and also increased more with wildfire, 929 percent compared to 129 percent. Both nutrients then decreased down to pre-fire conditions (Wan et al. 2001).

Two theories have been discussed to explain nutrient availability following fire. The Transient Maxima Hypothesis was found to be superior to the Enhanced Mineralization Hypothesis to explain nutrient changes. The Transient Maxima Hypothesis attributes increased annual production following a fire to less light competition and a reduction in nitrogen competition between plants. There is a strong pulse of nitrogen following a fire occurring only once, but this pool is quickly depleted. In most cases, areas burned only once respond similarly as areas never burned compared to areas burned twice or annually (Blair 1997).

Macronutrients like calcium (Ca), magnesium (Mg), sodium (Na), and potassium (K) increased following a fire in South Africa (Snyman 2003). This study also found decreases in the amount of organic matter and extractable phosphorus (P). Fire also increased the soil compaction and reduced the amount of infiltration. Generally, cations increase following a fire (Christensen 1976; Higgins et al. 1986), but other nutrients like nitrogen and carbon have various responses (Higgins et al. 1986). Mowing did not change the responses of nutrients to fire (Christensen 1976).

Most sites affected by fire return to pre-fire conditions in two to three years (Dix 1960; Antos et al. 1983; Whisenant and Uresk 1989; Wisenand and Uresk 1990; Redmann et al. 1993; White and Loftin 2000; Ericksen-Arychuk et al. 2002; Grant et al. 2010; Gittins et al. 2011; Vermeire et al. 2011). Some range qualities were improved from pre-fire conditions following fire. This can be valuable in two ways. If negative impacts are seen from a fire, it will only take a short time for conditions to go back to their starting point. If positive effects are seen, it can lead to a better understanding of the correct fire interval needed to maintain those results. Many studies connect major changes in ecological properties to yearly climatic conditions (Wright 1974; Biondini et al. 1989; Whisenant and Uresk 1989; White and Loftin 2000; Vermeire and Rinella 2009; Vermeire et al. 2011). Management after a fire can also cause different changes to species composition and phytomass.

Grazing and Fire

Fire and grazing is commonly managed together in mesic rangelands, but semi-arid rangelands normally discretely compartmentalize grazing and fire (Vermeire et al. 2014). In any ecosystem, management after fire is an important factor in determining the effects of prescribed fire (Wright 1974). Livestock grazing at the right time is imperative to maintain rangeland health (White and Currie 1983). Some agencies, like the Bureau of Land Management, have policy recommending the delay of livestock grazing after a wildfire or prescribed fire for at least two years (BLM 2007), but they do not specifically prohibit grazing after a fire. This has led many to believe that grazing is not permitted for two years following this type of disturbance. The current policy of the U.S. Forest Service in the Dakota Prairie Grasslands of South Dakota does not mention any type of probationary period after fire for livestock grazing (USFS 2005; USDOJ 2006). Grazing permits may be revoked if management believes the resource will be harmed by livestock grazing, but revocation is not necessary for every fire. Only a small portion of research has combined fire and grazing in the NGP, so it is yet to be determined the appropriate amount of time and conditions needed to resume grazing on burned rangelands and pasturelands. Cattle seem to show no preference to the season of burn (Vermeire et al. 2004), so research could solely focus on resource effects.

Standing biomass can be reduced by burning and clipping compared to control treatments (White and Currie 1983; Clark 2006). Western wheatgrass that was clipped and burned in the spring had less production during the beginning of the growing season. Plots that were clipped also maintained a shorter growth form compared to burned and control treatments. Blue grama also showed interesting patterns with clipping and fire. There was more production on the control plots in May and June, but blue grama yields were greater on burned and clipped compared to the control by the end of June. The burned only blue grama plots produced more than the control in June by $250 \text{ kg} \cdot \text{ha}^{-1}$, and clipped only sites produced $150 \text{ kg} \cdot \text{ha}^{-1}$ more than control plots. The only plots where clipping and fire separately produced similar results were during fall burns on threadleaf sedge plots. In conclusion, burning initially reduced the amount of blue grama, but it rebounded by the end of the season (White and Currie 1983).

There can be some dissimilarity between clipping and fire as management tools. Future studies should be discouraged from using clipping to simulate fire like other research has used clipping to simulate grazing (Stout et al. 1980; Hicks and Reader 1995; Smart et al. 2013; White et al. 2013).

Fire did not cause compositional changes in the transitional part of the NGP in Montana when the rangeland was non-grazed 80 years prior to wildfire (Antos et al. 1983). Fire and grazing can also cause other changes to the ecosystem besides compositional changes and production differences. Litter is generally reduced by fire, but slight changes can also occur from management after fire. Light grazing after fire increased the amount of humic mulch but decreased the amount of fresh mulch in North Dakota (Dix 1960). This same study found moderate grazing to decrease both types of mulch (Dix 1960).

Biomass collected after grazing a burned area decreased, but overall first year production was not affected. Fire and grazing were determined to increase grass productivity, while removing grazers after fire did not increase productivity. Grazing and fire did decrease the amount of forbs, annual grasses, and litter. Warm-season grasses also decreased as the fire interval and grazing both increased. The only problem associated with burning before grazing is loss of standing dead material. Grazing was not detrimental to rangelands burned in the summer at utilization rates of 17-50 percent. Plants have individual responses to fire instead of responses to climate (Vermeire et al. 2014).

Domestic sheep have been used to graze in the spring following a summer burn in Montana. Unlike the control, pastures grazed and burned did not have any standing dead. Animals did not show adverse signs to grazing previously burned areas, but stocking rates should be watched carefully. Stocking rate will be dependent upon regrowth. Until regrowth is known, grazing should be conservative or even postponed (Waterman and Vermeire 2011).

Patagonian rangelands have similar responses to fire and grazing. In most cases, fire can be followed by grazing, but grazing can also have detrimental effects on plant regrowth and vitality. Competition between plants may be an underlying factor for the amount of regrowth. There may have to be a resting period after fire before grazing resumes (Gittins et al. 2011). In Nevada, grazing was found to have a neutral effect on plant fire recovery (Bruce et al. 2007).

Standing crop was increased by spring burning but was reduced when spring burning was combined with spring grazing by bison in Nebraska. In the absence of grazing, rhizomatous grass

standing crop was increased for two growing seasons following spring burns, and summer burns had no effect on the standing crop. Bunchgrasses showed the same pattern on spring burns without grazing and spring burns with grazing as rhizomatous grasses. There were no differences seen on areas that were not subjected to fire and then grazed (Pfeiffer and Steuter 1994).

Nutrients released during fire can be affected by defoliation, dependent on plant uptake and root activity. One study looked at the interactions of both grazing and fire on root and tissue carbon and nitrogen along with bulk carbon and nitrogen in Kansas. They found both carbon and nitrogen to decline as sampling depth increased. Soil carbon was higher in the non-burned, non-mowed control treatments at 0-10 cm depth. Carbon was reduced by mowing, fire, and their combination; the greatest reduction was from mowing alone. Carbon was reduced 22 percent by mowing and 11 percent by burning alone. Mowing treatments affected bulk soil carbon and nitrogen the most out of all the treatments (Kitchen et al. 2009).

There is no consistent evidence that grazing directly after fire or during the growing season after the fire will have long-term negative effects on plant regrowth and composition. Pre- and post-fire conditions may drastically change the type of livestock grazing management needed following a fire. Grazing deferment in the early part of the first growing season following fire may be required to prevent damage, but forage yields could actually be increased by burning and grazing after June (White and Currie 1983). Spring burning should be avoided if rangelands and pasturelands are needed before June. Initial stocking rates should be lowered to allocate for reduced forage caused by a loss in standing dead material (Vermeire et al. 2014). It may be necessary to individually evaluate each pasture and range unit before use (Bruce et al. 2007), but the lack of clear conclusions stems from the long evolutionary history the NGP has with fire and grazing (Antos et al. 1983; Steuter 1987; Grant 2010).

Diversity is becoming an important factor for the future of range management. Diversity can be increased by combining disturbances like fire and grazing to produce heterogeneity in species composition and structure (Fuhlendorf and Engle 2001). "Pyric herbivory" controls grazing by using fire to increase historical patterns of grazing that were prominent throughout the Great Plains. Livestock have the potential to reproduce what has evolutionarily molded and sustained rangelands (Fuhlendorf et al. 2009). Increased heterogeneity promotes different plant species to be active under various conditions,

times, and micro-habitats to maintain ecosystem function and provide continuous forage (Isbell et al. 2011). Negative impacts are often seen on disturbance-free areas, as they see decreases in diversity and single species becoming more dominant (Belsky 1992). Disturbance is a very important part of retaining the health and bolstering the resistance of rangelands in the NGP.

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CHAPTER 2: EFFECTS OF EARLY SPRING TARGETED GRAZING ON CRESTED WHEATGRASS PASTURELANDS OF SOUTH DAKOTA

Abstract

Crested wheatgrass production is used to calculate stocking rates, but it is rarely consumed to expected levels due to palatability issues. Imbalances in grazing pressure impacts the adjacent rangeland community, leading to overgrazing and degradation of native vegetation. In this study, we implemented an early, high stock density grazing system to increase crested wheatgrass utilization. We collected plant species composition, ground cover, and phytomass production collected in 2013 and 2014 on grazed and non-grazed treatments. Targeted grazing increased the grazing pressure on pasturelands, with degree of disappearance above 70 percent in 2013, decreasing to 35 percent in 2014. There were no changes to plant species composition or canopy cover ($p > 0.05$) from targeted grazing, but they did change ($p \leq 0.05$) over time. Degree of disappearance can be increased with targeted grazing, but other techniques should be combined to reduced crested wheatgrass cover.

Introduction

Crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) was introduced from Eurasia, extensively planted in the 1930s, and now occupies between 6 -11 million ha in North America (Hulet et al. 2010). Crested wheatgrass increased soil stabilization while providing the means for livestock production during the Dust Bowl Days on marginal farmland (Rogler and Lorenz 1983). Crested wheatgrass provides valuable early spring forage and some fall forage for livestock and wildlife (Hyder and Sneva 1963), but it senesces early, reduces plant and animal species diversity, and changes grassland ecosystem processes provided by native rangelands (Lesica and DeLuca 1996; Christian and Wilson 1999; Heidinga and Wilson 2002; Hulet et al. 2010; Fansler and Mangold 2011).

Management strategies are used to reduce the cover of crested wheatgrass within rangeland communities. Numerous studies look at reducing crested wheatgrass while increasing native plant species by implementing high density grazing, herbicides, seeding, disking, and fire (Lodge 1960; Wilson

and Partel 2003; Pellant and Lysne 2005; Vaness and Wilson 2008; Hulet et al. 2010). The best options include long-term primary and secondary treatments that combine several management practices (Hansen 2007; Hulet et al. 2010). Control options are dependent on moisture availability, since most studies only achieve crested wheatgrass reductions during drought (Hanson and Wilson 2006). It will be extremely valuable if crested wheatgrass pastures can be diversified with native species, especially warm-season grasses. A combination of warm- and cool-season grasses maintains successful production by providing group species diversity with various “temporal guilds” that produce adequate forage for the entire growing season (Steuter 1987).

Livestock production that utilizes both range and pasture lands can reduce deterioration and overuse of native plant communities by deferring grazing in the early spring and using crested wheatgrass pastures (Frischknecht et al. 1953; Lodge 1970; Hart et al. 1988). Animal gains per unit land area could increase with complimentary grazing at an optimal ratio of range and pasturelands when adequate precipitation occurs (Hart et al. 1988; Derner et al. 2008). Complimentary grazing with crested wheatgrass could also be referred to as targeted grazing, a tool used for controlling weedy species like leafy spurge (*Euphorbia esula* L.) and yellow star-thistle (*Centaurea solstitialis* L.) (Launchbaugh et al. 2006). However, little research is available on targeted grazing graminoid species with cattle. One study used cattle to reduce cheatgrass (*Bromus tectorum* L.) fuel loads (Diamond et al. 2009), but few studies have specifically used targeted grazing cattle during an explicit season for a defined length of time to reduce crested wheatgrass cover.

This study observed the effects of targeted grazing crested wheatgrass during the early growing season with high stock density cow/calf pairs. The objectives of this study were to: 1) increase the degree of livestock grazing on crested wheatgrass pasturelands on the Dakota Prairie Grasslands, Grand River District (GRD), 2) increase frequency and cover of native species, and 3) decrease the frequency and cover of crested wheatgrass. The early grazing strategy was designed to target crested wheatgrass plants with heavy use while reducing high grazing use on native plant species when they are more actively growing. Our hypothesis was that competitive pressures induced on crested wheatgrass will decrease crested wheatgrass cover while increasing the occurrence and cover of native plant species within crested wheatgrass pastures.

Methods

Site Description

We conducted the study within a 150 ha pasture on the GRD, Perkins County, South Dakota, USA (lat 45.924, long -102.720). The GRD covers approximately 60 000 ha and is part of the larger Dakota Prairie Grasslands managed by the United States Forest Service (USFS). The elevation of the study area ranges from 761 to 824 m. The continental climate found on the GRD has cold winters and hot summers. The average long-term temperature for the area is 5.5 °C with the lowest temperatures in January (-9 °C) and highest temperatures in July (22 °C). The average long-term precipitation for the area is 45 cm (NOAA 2014). We determined the average precipitation for the area by averaging data from weather stations in Hettinger, ND, Lemmon, SD, Lodgepole, SD, and Bison, SD, for 2013 and 2014. Average precipitation in 2013 was 68 cm and totaled 43 cm by October 2014 (NOAA 2014; UNL 2014). The GRD is located in the Major Land Resource Area (MLRA) 54 in the Northern Great Plains (NGP).

The major ecological sites on the study area were clayey, loamy, and sandy. Clayey sites were historically dominated by western wheatgrass (*Pascopyrum smithii* [Rydb.] A.Love) and green needlegrass (*Nasella viridula* (Trin.) Barkworth). Disturbed sites often classified as a blue grama (*Bouteloua gracilis* (Willd. Ex Kunth) Lag. Ex Griffiths) and western wheatgrass community as detailed in the state and transition model. Phytomass production averages between 784 - 2 350 kg · ha⁻¹. Sites with crested wheatgrass are expected to produce between 2 018 - 4 483 kg · ha⁻¹. Loamy sites also have a historical vegetation community dominated with western wheatgrass and green needlegrass, containing more needle-and-thread (*Heterostipa comata* (Trin. & Rupr.) Barkworth) than clayey sites. Degraded sites have more blue grama and western wheatgrass. Phytomass production averages between 784 - 2 688 kg · ha⁻¹. Sites with crested wheatgrass are expected to produce between 2 466 - 5 156 kg · ha⁻¹. Sandy sites were historically dominated by prairie sandreed (*Calamovilfa longifolia* [Hook.] Scribn.), bluestem species (*Andropogon* spp.), and little bluestem (*Schizachyrium scoparium* (Michx.) Nash). Degraded sites will have more threadleaf sedge (*Carex filifolia* Nutt.) and blue grama. Phytomass production averages between 895 - 2 688 kg · ha⁻¹. Sites with crested wheatgrass are expected to produce between 1 569 - 3 138 kg · ha⁻¹ (USDA-NRCS 2014).

Pasture Stocking and Grazing

We visually estimated the abundance of crested wheatgrass to stratified research in Pasture 1A within Allotment 1. This pasture had the cooperation of permittees, the greatest potential for early spring grazing, and a mixture of rangeland and crested wheatgrass plantings. We used a randomized paired plot design with two replications. We created two pastures within the larger 1 425 ha pasture, each built using a single, turbo wire. The pastures were 38 ha and 39 ha in size. The entire pasture was previously stocked at 1.17 animal unit months (AUM) · ha⁻¹. Producers turned cattle out as soon as possible in May of both years. In 2013, the pastures were stocked at 2.67 AUM · ha⁻¹ and 1.89 AUM · ha⁻¹. Cow/calf pairs grazed for approximately 30 d. In 2014, one pasture was stocked at 1.24 AUM · ha⁻¹ for 12 d, and the other was stocked 1.75 AUM · ha⁻¹ for 40 d. Producers monitored cattle to make sure there was sufficient available forage each year. Cattle were expected to graze until utilization was at approximately 70 percent (visually estimated). We advised producers to keep cattle on targeted grazing pastures until this point, but they could move cattle at their preference. At the end of each grazing period, producers moved cattle into the adjacent pasture for the remainder of the season. Producers made the final decision when cattle were moved and impacted the amount of time for each grazing period. Fences remained up to prevent cattle from grazing regrowth.

Data Sampling

We collected data using paired plots (grazed/ non-grazed), approximately 10 x 10 m in size, that were stratified on areas in Pasture 1A that had high ocular estimates of crested wheatgrass with intermixed native plant species. One paired plot in each replicate was randomly chosen as the non-grazed control and the other paired plot unfenced and accessible to free grazing cattle. We placed plots away from water tanks to reduce disproportional use. We marked grazed plots with four pins, a small pole, and GPS points using a Juno Trimble 3B (© Trimble Navigation Limited) handheld device to create permanent plots. We established transects at 2.44 m, 4.88 m, and 7.32 m, which ran from 0 - 9.75 m north to south. We collected vegetative data prior to cattle grazing in 2013 and again in late July during peak phytomass production in 2013 and 2014.

We collected forb and shrub density with a 0.25 m² quadrat frame and graminoid frequency in a 0.1 m² nested frame every 0.91 m (Dix 1958; Biondini et al. 1989). We also collected percent canopy cover every 0.91 m by using a modified Daubenmire frame (Daubenmire 1959) within the 0.25 m² quadrat. Modification was a 0.25 m² frame compared to 20 x 50 cm frames. Cover classes were not used. Cover for vegetation, bare ground, and litter was recorded as a percentage to the nearest whole number, totaling 100 percent per quadrat frame. We used a 3-mm diameter pin for point intercept every 0.61 m to record ground cover (Levy and Madden 1937; Eden and Bond 1945; Bonham 1989) for basal live, basal litter, and bare ground. We collected phytomass production within one to three days after all cattle grazing terminated using 4 - 0.178 m² hoops placed randomly within each plot in 2013 and 2014. We clipped by species and dried all the biomass at 60 °C for 72 hours. We weighed dried phytomass and averaged weights for each plot (grazed and non-grazed).

Statistical Analysis

We used permutation-based non-parametric multivariate analysis of variance (perMANOVA) using Relative Sorensen Distance Measures and a two-way factorial to compare ground cover, species frequency, density, and canopy cover between treatments (grazed and non-grazed) and between years (2013 and 2014) in PC-ORD version 6.0. We used 4 999 runs and a random number seed starting at the time of day. We transformed frequency data using an arcsine square root transformation ($b = [2 \cdot \pi] \arcsin [\sqrt{x_{ij}}]$). We graphically displayed perMANOVA results using non-metric multidimensional scaling (NMS) for ordination of plant species frequency and density, canopy cover, and ground cover using the autopilot slow and thorough setting with Relative Sorensen distance measures. We calculated plant species richness, evenness, Shannon-Weiner Index, and Simpson's Index (McCune and Mefford 2011). We used Student's *t* tests ($\alpha = 0.05$) to compare production and ground cover for treatments in each year (Steel et al. 1996).

Results

Plant Species Composition

PerMANOVA scores showed no year x treatment interaction or treatment effects ($p > 0.05$) after two years of early spring targeted grazing for plant species composition, frequency, density, canopy cover, and ground cover (Table 2.1). NMS ordination also showed no variation due to the grazing treatment (Figures 2.1 and 2.2). There was a time effect ($p \leq 0.05$) for all of the data parameters (Table 2.1). NMS ordination supported this time effect for frequency and density (Figure 2.3) and canopy cover (Figure 2.4).

Table 2.1. PerMANOVA p – values using Relative Sorensen distance measures for targeted grazing plots on the Dakota Prairie Grasslands - Grand River District, South Dakota, USA, using various sampling techniques to measure species composition and canopy and ground cover in 2013 and 2014.

Source	Frequency and Density	Canopy Cover	Ground Cover
Time	0.004	0.003	0.008
Treatment	0.643	0.769	0.448
Interaction	0.997	0.983	0.089

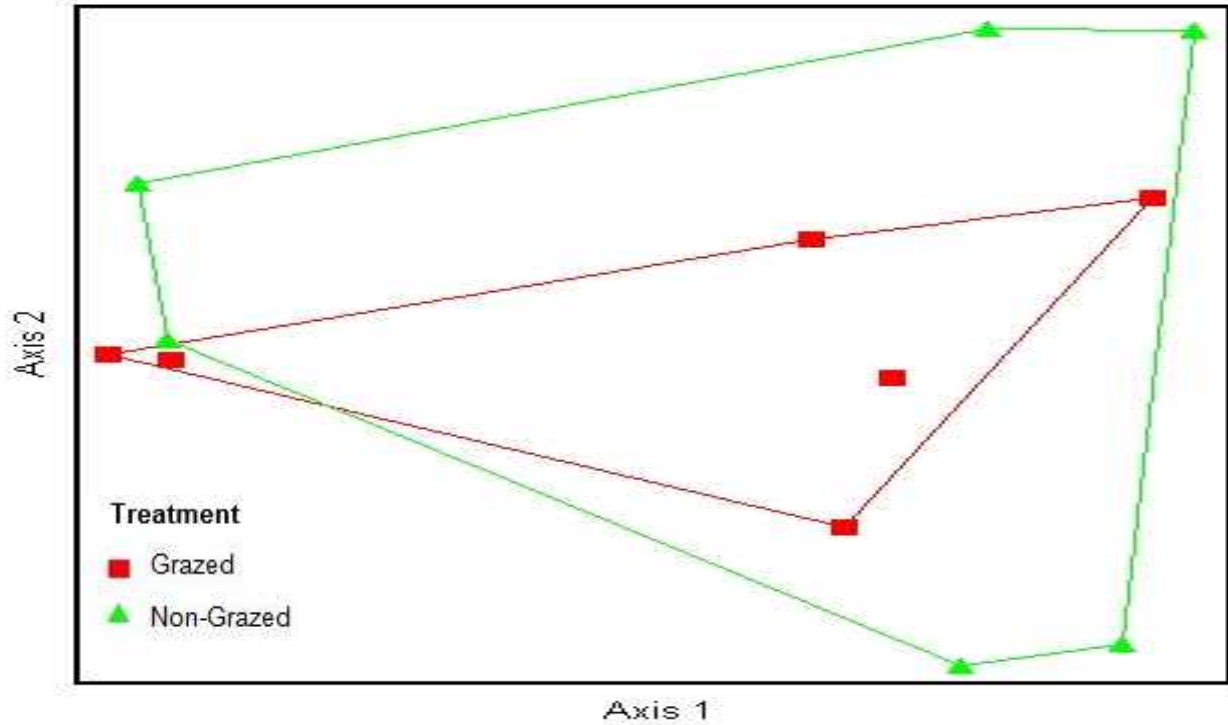


Figure 2.1. Plant species composition from quadrat frequency and density for targeted grazing by treatments (grazed and non-grazed) on the Dakota Prairie Grasslands - Grand River District, Perkins County, South Dakota, USA, for 2013 and 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.539. Axis 2 had an r^2 value of 0.398.

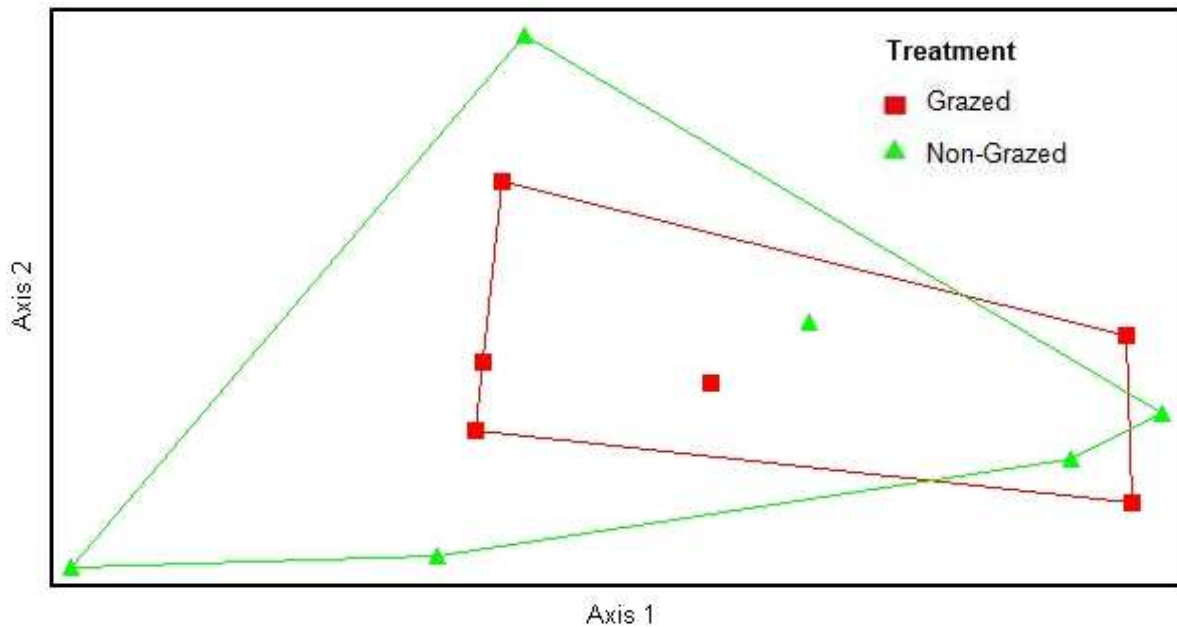


Figure 2.2. Plant species composition from canopy cover for targeted grazing by treatments (grazed and non-grazed) in the Dakota Prairie Grasslands - Grand River District, Perkins County, South Dakota, USA, for 2013 and 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.828. Axis 2 had an r^2 value of 0.108.

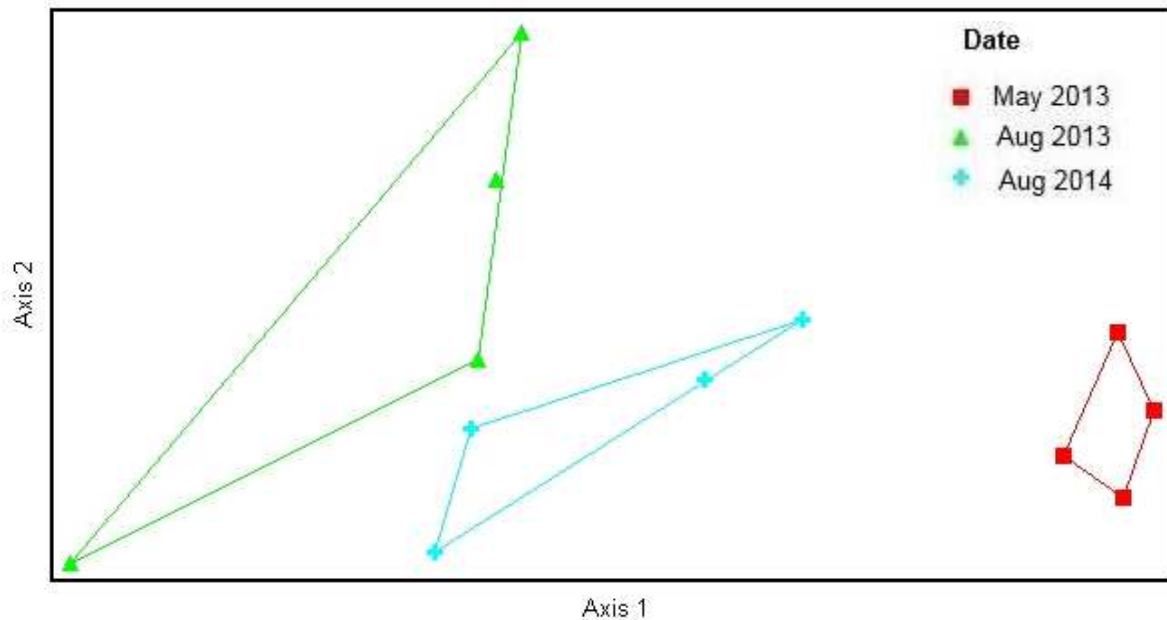


Figure 2.3. Plant species composition for canopy cover for targeted grazing through time on the Dakota Prairie Grasslands - Grand River District, Perkins County, South Dakota, USA, from 2013 to 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.828. Axis 2 had an r^2 value of 0.108.

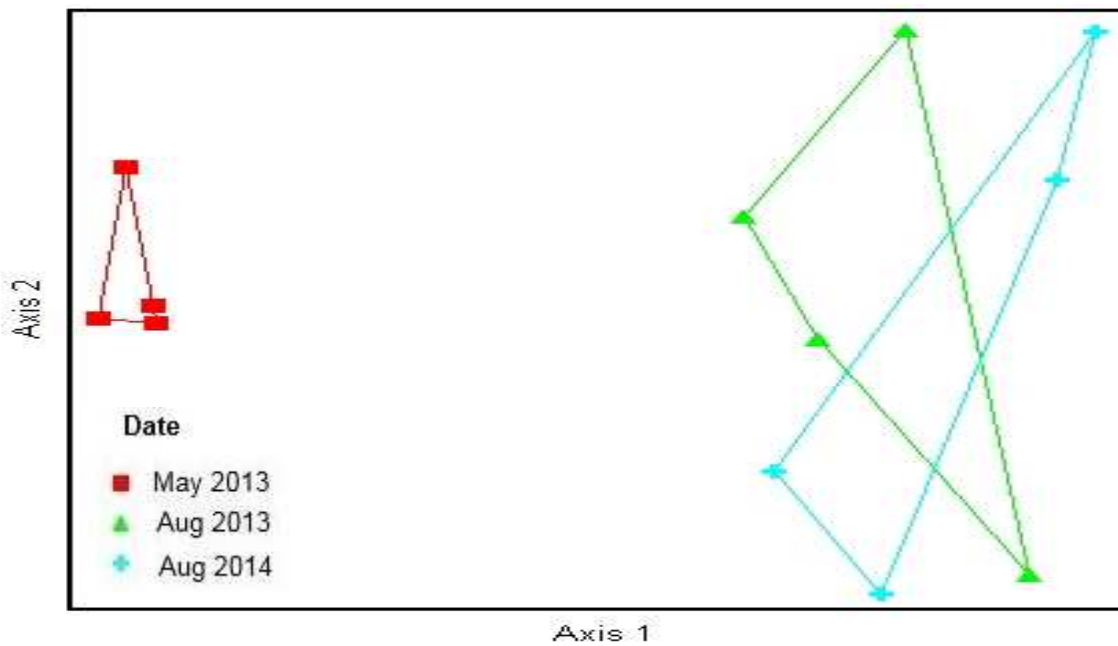


Figure 2.4. Plant species composition from quadrat frequency and density for targeted grazing through time on the Dakota Prairie Grasslands - Grand River District, Perkins County, South Dakota, USA, from 2013 to 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.539. Axis 2 had an r^2 value of 0.398.

Both grazed and non-grazed plots had high amounts of crested wheatgrass (> 80 percent frequency). Species changed the most through time, since there was a trend for fewer species present during the baseline data collection period in May 2013 compared to post-treatment data collections in August 2013 and 2014. Although not significant, some trends started to appear with specific species. Green needlegrass and sun sedge (*Carex inops* L.H. Bailey ssp. *heliophila* (Mack.) Crins) canopy cover increased over time in both treatments averaging zero percent in 2013 to three percent in 2014. Threadleaf sedge (*Carex filifolia* Nutt.) canopy cover decreased in both treatments over time from an average of five percent in 2013 to below one percentage in 2014. Blue grama and crested wheatgrass maintained canopy cover amounts over time and within treatments, averaging 13 percent and 30 percent, respectively.

Sweetclover canopy cover increased ($p \leq 0.05$) over time. In August 2013, sweetclover cover was 4-8 percent, increasing to 12-18 percent cover in August 2014. Some forbs and grasses found on both treatments in 2013 were not detected after sweetclover invaded in 2014, including purple threeawn (*Aristida purpurea* Nutt.), dandelion (*Taraxacum officinale* F.H. Wigg.), and prostrate spurge (*Chamaesyce maculate* (L.) Small). Black medic (*Medicago lupulina* L.) was one forb that was detected on grazed treatments over time. There were several other species like Canadian horseweed (*Conyza canadensis* (L.) Cronquist) detected over time on both treatments and increased ($p \leq 0.05$) on grazed plots.

There was no change ($p > 0.05$) in plant species diversity due to treatment. Richness was higher on grazed plots in August 2013 compared to non-grazed plots the same year; however, this trend did not continue into 2014 with additional targeted grazing. Richness increased and evenness decreased from baseline sampling in May 2013 when compared to August 2013 and 2014 samples. Shannon Diversity indices remained stable throughout time and treatment. Simpson's Index for May 2013 in both grazed and non-grazed trended to be higher than the August 2013 grazed and non-grazed treatments. Otherwise, Simpson's Index remained similar through the study period (Table 2.2).

Table 2.2. Species diversity indices (\pm standard error) for targeted grazing plots on the Dakota Prairie Grasslands - Grand River District, South Dakota, USA, after two years of grazing (2013, 2014).

Treatment	Date	Richness	Evenness	Shannon Diversity	Simpson's Index
Grazed					
	May 2013	9.00 \pm 1.00	0.89 \pm 0.06	1.94 \pm 0.03	0.83 \pm 0.02
	August 2013	25.50 \pm 3.50	0.58 \pm 0.12	1.85 \pm 0.30	0.69 \pm 0.13
	August 2014	22.00 \pm 4.00	0.64 \pm 0.12	2.00 \pm 0.47	0.71 \pm 0.14
Non-Grazed					
	May 2013	9.50 \pm 0.50	0.87 \pm 0.04	1.96 \pm 0.14	0.84 \pm 0.02
	August 2013	16.50 \pm 1.50	0.65 \pm 0.02	1.81 \pm 0.01	0.72 \pm 0.03
	August 2014	19.50 \pm 3.50	0.59 \pm 0.12	1.75 \pm 0.46	0.66 \pm 0.16

Ground Cover

PerMANOVA scores showed differences ($p \leq 0.05$) on ground cover over time. There was no treatment or year x treatment interaction ($p > 0.05$) (Table 2.1). Point intercept and the modified Daubenmire frames had higher ($p \leq 0.05$) basal litter cover in the baseline data in May 2013 compared to subsequent collections in 2013 (post first year grazing) and 2014. Point intercept showed more ($p \leq 0.05$) bare ground on grazed plots compared to non-grazed plots in 2014, but there was no difference between treatments in 2013. Modified Daubenmire's showed less ($p \leq 0.05$) bare ground in August 2014 compared to August 2013 on non-grazed plots alone. There was no difference ($p > 0.05$) in other parameters between years or treatments. There was no change ($p > 0.05$) in live cover or crested wheatgrass cover between treatments or years. Grazed and non-grazed treatments did not group separately in NMS ordination but differed between May 2013 from August 2013 and 2014 when grouped by date (Figures 2.5 and 2.6).

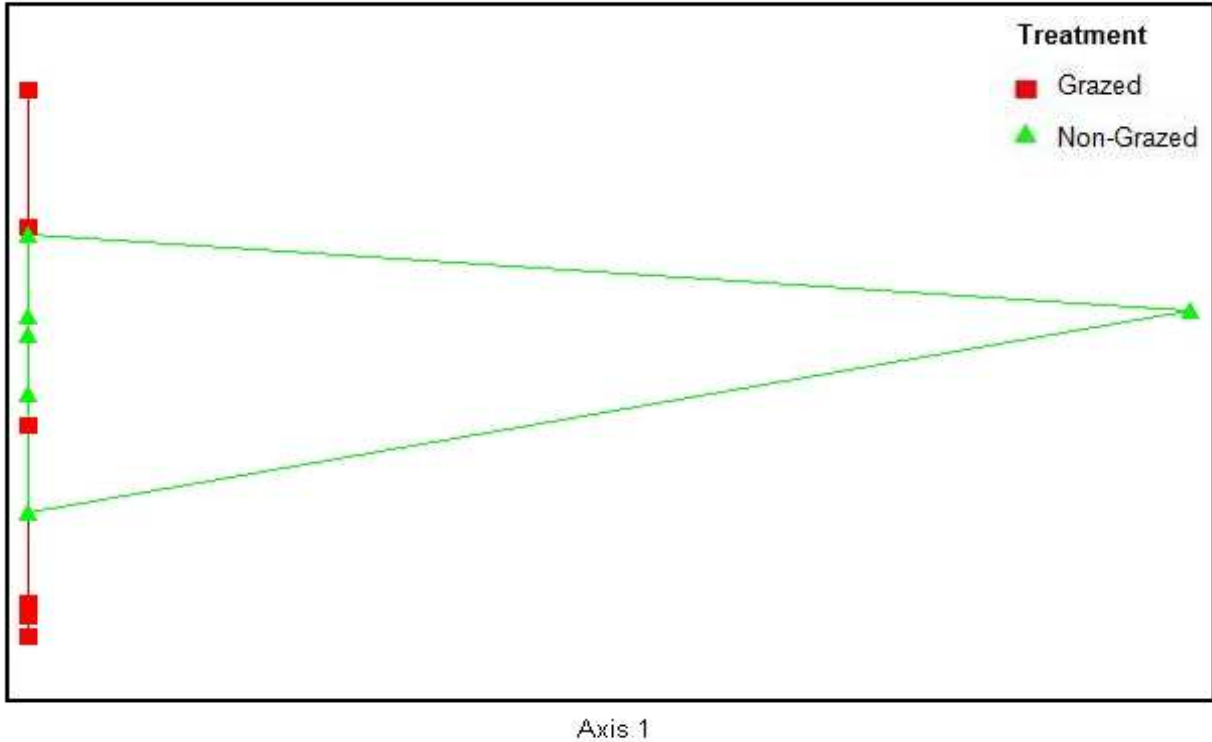


Figure 2.5. Ground cover (live, litter, and bare ground) for targeted grazing treatments (grazed and non-grazed) on the Dakota Prairie Grasslands - Grand River District, Perkins County, South Dakota, USA, from 2013 and 2014. The ordination found a one-dimensional solution. Axis 1 had an r^2 value of 0.645.

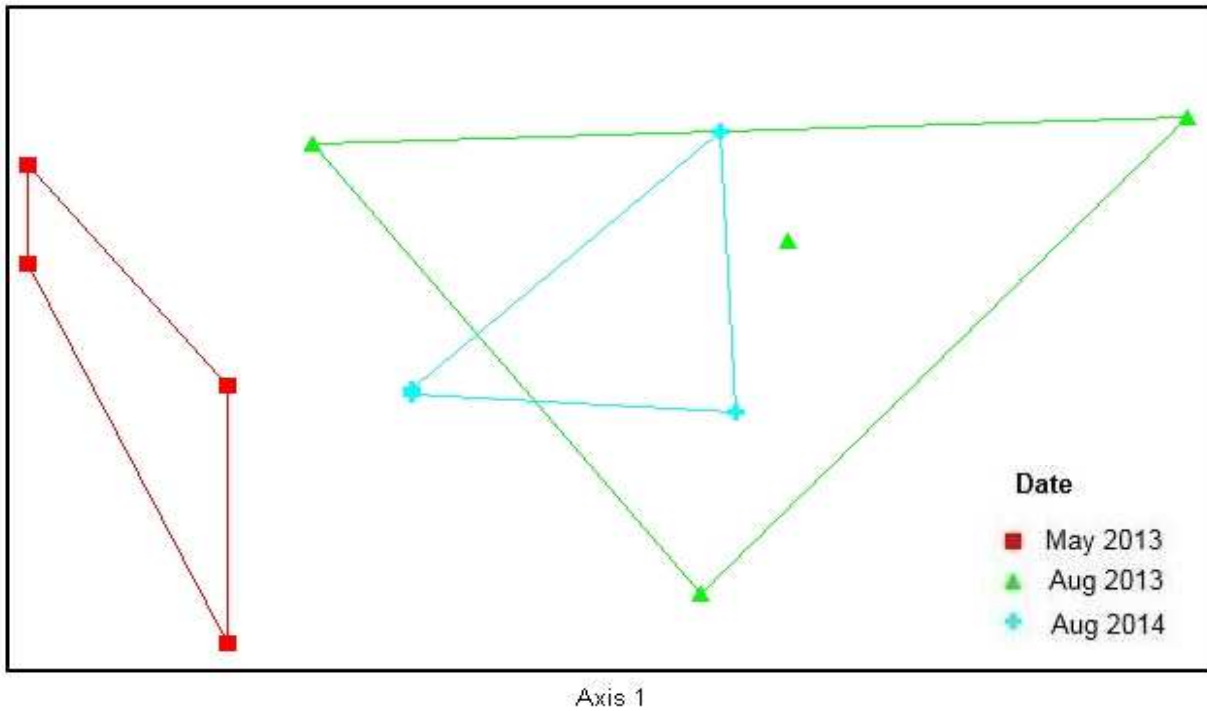


Figure 2.6. Ground cover (live, litter, and bare ground) for targeted grazing through time on the Dakota Prairie Grasslands - Grand River District, Perkins County, South Dakota, USA, from 2013 to 2014. The ordination found a one-dimensional solution. Axis 1 had an r^2 value of 0.986.

Phytomass Production and Degree of Disappearance

Average phytomass production in 2013 on non-grazed plots was 2 500 kg · ha⁻¹. This increased to 3 300 kg · ha⁻¹ in 2014. Phytomass production was different ($p \leq 0.05$) between grazed and non-grazed plots in 2013 due to an average degree of disappearance by grazing cattle of 73 percent on grazed plots. Phytomass production did not differ ($p > 0.05$) between grazed and non-grazed plots in 2014 due to high rainfall, sweetclover growth, and a lower degree of disappearance by grazing cattle. Average degree of disappearance was 36 percent in 2014. There was an average 67 kg · ha⁻¹ of standing dead material in 2013 and 64 kg · ha⁻¹ of standing dead in 2014 on non-grazed plots. There was no standing dead on grazed plots in either year. Sweetclover production averaged between 4-10 kg · ha⁻¹ in 2013, increasing to 100-2 000 kg · ha⁻¹ in 2014 on grazed and non-grazed plots.

Discussion

We found few changes to the plant community using targeted grazing on crested wheatgrass after two years on the GRD. Early spring grazing is often used for managing crested wheatgrass pastures (Angell 1997; Wilson and Partel 2003; Vaness and Wilson 2008), but using a high stock density, targeted grazing approach to increase utilization during a critical time period to decrease the population of crested wheatgrass is new. Heavy stocking rates and density are important for reducing standing dead material and uneven grazing distribution commonly associated with crested wheatgrass (Frischknecht et al. 1953; Balph and Malechek 1985), since standing dead increases with grazing avoidance (Ganskopp et al. 1993). After two years of grazing, our grazing treatment eliminated standing dead while the non-grazed treatment had approximately 64 kg · ha⁻¹ of standing dead. Grazed plots were heavily utilized both years, but the effects were more pronounced in 2013 when degree of disappearance was 73 percent. Degree of disappearance was lower in 2014 at 36 percent because of sweetclover invasion, which occurred after the grazing treatment, and lower stocking rates induced by producers. Cattle were unable to manage the rapid growth of sweetclover in combination with high amounts of precipitation.

High grazing pressure can decrease the amount of basal live plant cover (Naeth et al. 1991). However, we did not find any changes to live cover after two grazing years. Bare ground did increase on grazed plots in 2014. Both grazed and non-grazed treatments had decreased basal litter and increased

bare ground in 2014. This was most likely due to sweetclover. Sweetclover outcompetes native grasses and forbs for water with its large taproot and shades out lower growing plants, creating bare ground and less herbaceous material for cover (Blackshaw et al. 2001). Changing the basal cover of crested wheatgrass is extremely difficult and may not be possible with extreme management changes like cultivation (Lodge 1960).

Overall, plant species diversity was not influenced by targeted grazing. Forbs and graminoids tended to increase over the season from May to August as seen in other studies in the NGP (Biondini et al. 1998). Many of the changes seen in species composition may be attributed to increases in sweetclover. Sweetclover puts additional nitrogen into the soil that can cause native species to decrease and exotic species to increase, but it can increase production of some native species (Tilman 1987). Sweetclover is much taller than other plants found in the mixed-grass portion of the NGP and can easily shade out shorter statured plants (DeLuca and Lesica 2000).

Research shows variable responses of forbs to grazing and weather conditions. Forbs can increase (Biondini and Manske 1996; Vermeire et al. 2008) or decrease (Schuman et al. 1999) under heavy grazing, so individual responses are commonly associated with climatic factors (Biondini and Manske 1996). We did not attribute changes in forb species composition to grazing, since both treatments experience an increase in forb diversity. Grazed and non-grazed plots all started out with eight different forb species, but summer sampling periods found over 20 different species of forbs on both grazed and non-grazed plots. Climatic factors and seasonal growth patterns are most likely controlling this response. Precipitation was well above average for both years, and more forbs were expressed during the summer season than late spring season.

We found no change in the cover or frequency of blue grama, but most studies found blue grama to increase under heavy grazing after it experienced competitor release from taller plants (Hart et al. 1993; Biondini and Manske 1996; Schuman et al 1999; Derner and Hart 2007). It is possible to reduce blue grama cover under extremely heavy grazing (Hart et al. 1993; Biondini et al. 1998). However, a longer treatment effect with sustained heavy grazing may be needed to create this impact. Heavy grazing was achieved in 2013, but producers removed cattle early in 2014 and did not achieve the same amount of grazing as 2013.

Western wheatgrass frequently becomes the dominant species on moderately grazed plots, while non-grazed sites have lower amounts of western wheatgrass and other native cool-season grasses (Schuman et al. 1999; Ingram et al. 2008). Western wheatgrass started to decrease on our non-grazed plots, since grazing was removed. We found green needlegrass to increase over time on both treatments. Sun sedge increased, while threadleaf sedge decreased. Sedge species tend to remain stable on grazed and non-grazed sites (Biondini and Manske 1996). Overall, sedges in our study did remain stable, since one decreased and the other increased. Sun sedge appeared to tolerate shading from sweetclover better than threadleaf sedge.

Crested wheatgrass frequency and cover did not change because it is grazing tolerant and the area received high amounts of precipitation during the study. Precipitation was well above average and helped maintain the vigor of crested wheatgrass, along with native species, after it was heavily grazed during the early stages of growth. Managing crested wheatgrass is easier during dry periods because plants experience both water and defoliation stress, but it goes dormant faster and is not readily grazed by herbivores. Management is even more important during wet years because crested wheatgrass will put more energy into seed production (Miller et al. 1990; Hansen and Wilson 2006). Species compositional changes were not attributed to targeted grazing. Long-term climatic factors are the major influence on plant communities of the NGP instead of short-term grazing systems (Currie 1970; Hart et al. 1993; Derner and Hart 2007; Vermeire et al. 2008).

It is important to graze species like crested wheatgrass early, since crude protein decreases as the growing season progresses (Kamstra 1973; Johnson et al. 1998). It is even more important to have species group diversity (cool- and warm-season) to provide uninterrupted forage throughout the growing season for wildlife and livestock (Kamstra 1973; Steuter 1987). Diversifying crested wheatgrass pastures may not be sensible or manageable, since most strategies are inconsistent, temporary, or solely implemented (Hansen and Wilson 2006; Hansen 2007; Hulet et al. 2010). Constant management with secondary treatments is necessary. The best use of crested wheatgrass is to use it early in the season to delay grazing on more sensitive native rangelands. Complementary grazing allows native rangelands to rest and produce more forage (Frischknecht et al. 1953; Smoliak 1968; Lodge 1970; Hart et al. 1988).

Fire may prove to be the best combination with targeted grazing to diversify and control crested wheatgrass invasion on pasturelands. Fire decreases the amount of standing dead, thus increasing forage quality and decreasing avoidance (Gittins et al. 2001; Augustine and Milchunas 2009; Vermeire et al. 2014), and may help reduce the vigor of crested wheatgrass plants and seedling survival. Caespitose plants like crested wheatgrass have more fuel availability in a small area to increase consumption and heat stress compared to rhizomatous plants (Antos et al. 1983; Pfeiffer and Steuter 1994; Ericksen-Arychuk et al. 2002; Snyman 2003). Fire can control grazing distribution without additional fencing by attracting livestock to new growth (Lodge 1960; Wright 1974; Fuhlendorf and Engle 2001; Ericksen-Arychuk et al. 2002; Vermeire et al. 2004; Fuhlendorf et al. 2009). A combination of targeted grazing and fire would be effective at increasing crested wheatgrass use, along with structural heterogeneity and biodiversity (Fuhlendorf and Engle 2001).

Conclusions

Many crested wheatgrass pastures in the GRD are underutilized and adjacent rangelands are over-utilized. Most producers turn out cattle on June 1. Crested wheatgrass is available for use at least two to three weeks prior to this turn out date, but it may not be usable long after July 1 because it senesces and becomes rank. It is effective to utilize crested wheatgrass before native species start growing in May. Targeted grazing crested wheatgrass will decrease the pressure on native rangelands early in the season, maintaining range readiness, and decrease the amount of additional feed needed for cattle before they are turned out on pasture. Most cattle will not graze crested wheatgrass if native rangelands are available after late-June, but we were effective at manipulating cattle into targeted grazing crested wheatgrass with electric fencing. Although crested wheatgrass was not reduced by our high stock density, early grazing treatments, native plant species were also deemed unaffected. If cattle are confined or drawn to areas dominated by crested wheatgrass, crested wheatgrass utilization will increase and native sites can be reserved until later in the grazing season. Other options besides fencing, like fire, could also be used to maneuver livestock to crested wheatgrass pastures.

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CHAPTER 3: EFFECTS OF EARLY SPRING WILDFIRE ON CRESTED WHEATGRASS DOMINATED PASTURELANDS AND RANGELANDS OF SOUTH DAKOTA

Abstract

Fire research increases in the Southern Great Plains, but limited data is available for the Northern Great Plains. We looked at plant species composition, ground cover, phytomass production, and soil nutrient responses to fire on the Grand River District in South Dakota, part of the Dakota Prairie Grasslands, for two growing seasons following a spring wildfire. There were no species composition or phytomass production differences ($p > 0.05$) four or 16 months after fire (MAF). Non-metric multi-dimensional scaling showed plant species composition changes were related to site and year variability. Bare ground increased ($p \leq 0.05$) and basal litter decreased ($p \leq 0.05$) on burned plots four MAF, but no differences ($p > 0.05$) were found 16 MAF. Soil nutrients tended to be higher five MAF. Above average precipitation following the fire kept most variables similar between burned and non-burned plots.

Introduction

Prescribed fire is not a commonly used tool in the Northern Great Plains (NGP), especially in the mixed-grass prairie despite the long history of fire in this region, both lightning- and man-caused. Shortly after European colonization, fire suppression techniques and fragmentation sharply decreased fire frequency (Sauer 1950; Higgins 1984; Axelrod 1985; Umbanhowar 1996; Brown et al. 2005) even though grasslands evolved with fire (Steuter 1987; Anderson 2006; Grant 2010). Few management strategies incorporate fire because there are risks, logistic difficulties, and societal concerns associated with using prescribed fire (White and Currie 1983). The impacts of fire on grasslands are extensively studied in the Southern Great Plains (Hulbert 1969; Abrams and Hulbert 1987; Belsky 1992; Ansley and Castellano 2006; Limb et al. 2011; McGranahan et al. 2012) compared to the NGP. Recently, fire has become a fast growing research topic, but there is still much to be learned in regards to the role of fire in the NGP.

There are variable responses to fire in regards to biomass (Steuter 1987; Vermeire et al. 2011), species composition (White and Currie 1983; Whisenant and Uresk 1989), and soil properties (Raison 1979; Wan et al. 2001). Generally, warm-season grasses increase biomass and cover after spring fires when cool-season grasses are more vulnerable to fire damage, and cool-season grasses increase biomass and cover after summer or fall fires when warm-season grasses are more vulnerable to fire damage (White and Currie 1983; Redmann et al. 1993; Howe 1995; Shay et al. 2001). There may not be species compositional changes associated with fire (Brockway et al. 2002; Grant et al. 2010; Smith and McDermid 2014). However, most individual plant species responses vary (Antos et al. 1983; Vermeire et al. 2011). Basal litter decreases, bare ground increases, and standing dead decreases in every fire (Engle and Bultsma 1984; Redmann et al. 1993; Shay et al. 2001; Vermeire et al. 2014). Fire releases nutrients from litter into available forms for plant uptake (Higgins et al. 1986; Blair 1997; Snyman 2003). Many of the changes seen in regards to plant species composition, phytomass production, and soil nutrients are related to precipitation (Wright 1974; White and Currie 1983; Gartner et al. 1986; Biondini et al. 1989; Whisenant and Uresk 1989; Biondini and Manske 1996; White and Loftin 2000; Vermeire and Rinella 2009; Vermeire et al. 2011; Vermeire et al. 2014), since the NGP is resistant to various forms of disturbance (Biondini et al. 1998).

Fire can be a valuable management tool in the NGP by offering managers the ability to manipulate herbivory without fencing (Ericksen-Arychuck et al. 2002; Vermeire et al. 2004) and mimic historic grazing patterns (Augustine and Milchunas 2009) that can help increase structural heterogeneity and biodiversity (Fuhlendorf and Engle 2001; Vermeire et al. 2004; Fuhlendorf et al. 2009). Fire is also used to improve wildlife habitat, control undesired species, and increase palatability (Wright 1974; Augustine and Milchunas 2009). Fire could be helpful in controlling the use and spread of crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) and other problem herbaceous species like Kentucky bluegrass (*Poa pratensis* L.), field brome (*Bromus avensis* L.), and cheatgrass (*Bromus tectorum* L.) by direct mortality and increasing grazing utilization (Lodge 1960; Engle and Bulktsma 1984; Whisenant and Uresk 1990; Vermeire et al. 2011).

This study observed the effects of a spring wildfire on the plant community composition, ground cover, phytomass production, and soil nutrients on mixed-grass prairies in the NGP. Our objectives were

to use a natural experiment to (1) compare pre- and post-burned species composition, ground cover, and phytomass production on burned and non-burned native sandy and shallow sandy, and crested wheatgrass sandy ecological sites in South Dakota; and (2) compare soil nutrient amounts on non-defoliation and June defoliation treatments to simulate grazing and non-grazing within burned and non-burned plots 4-16 months after fire (MAF). Our study hypothesis was no effect to species composition and phytomass production, with bare ground increasing and litter decreasing on burned plots. We also hypothesize that soil nutrient availability will increase following the fire, and with the fire and defoliation treatment interaction.

Methods

Site Description

We conducted the study on the Grand River District (GRD) of the Dakota Prairie Grasslands, Perkins County, South Dakota, USA. The GRD covers approximately 60 000 ha, with our study comprising a small subset of pastures affected by a 2013 wildfire and adjacent pastures unaffected by the fire. The United States Forest Service (USFS), with input from the Grand River Grazing Association, manages the GRD. The elevation of the study area ranges from 761 to 849 m. The continental climate found in the GRD has cold winters and hot summers due to open landscapes without large bodies of water or landforms. The lowest temperatures occur in January (-9°C), and the highest temperatures occur in July (22°C). The average long-term temperature for the area is 5.5°C . The average long-term precipitation for the area is 45 cm (NOAA 2014). We found the average precipitation for the area by combining data from weather stations in Lemmon, SD, Lodgepole SD, and Bison, SD, for 2013 and 2014, since precipitation amounts were variable and well above the long-term average. Average precipitation in 2013 was 74 cm, and average precipitation up to October 2014 was 45 cm (NOAA 2014; UNL 2014).

The GRD is located in the Major Land Resource Area (MLRA) 54 in the NGP. The ecological sites used in this study were sandy and shallow sandy. Sandy and shallow sandy sites were historically dominated by prairie sandreed (*Calamovilfa longifolia* [Hook.] Scribn.), and bluestems (little bluestem (*Schizachyrium scoparium* (Michx.) Nash var. *scoparium*), bluestem species (*Andropogon* spp), and little bluestem (*Schizachyrium scoparium* (Michx.) Nash). Degraded sites have more threadleaf sedge (*Carex*

filifolia Nutt.) and blue grama (*Bouteloua gracilis* (Willd. Ex Kunth) Lag.) (USDA-NRCS 2014). A large portion of the GRD was previously farmed and reseeded to crested wheatgrass (*Agropyron cirstatum* [L.] Gaertn.) in the 1930s to 1950s (Weisner 1980). All of the grazing allotments, but one, are under 3-5 pasture deferred rotational grazing varying in dates from May to October. One allotment is grazed in early May and left non-grazed for the remainder of the growing season for cervidae habitat (Table 3.1).

Table 3.1. Stocking rates for burned and non-burned pastures in the GRD, Perkins County, South Dakota, for 2012-2014 with approximate grazing dates.

Pasture	2012	2013	2014	Grazing Dates (mo)
	-----AUM · ha ⁻¹ -----			
3B	0.91	1.38	0.94	June – October
3BH	0.55	0.23	0.65	May
4ANW	0.91	0.99	0.60	June – September
4ASE	0.83	0.87	0.90	July - September
4BS	0.74	0.62	0.60	May - August
5ASE	0.85	0.42	0.85	May - October
5ASW	0.53	0.53	1.09	June - October
5AWA	1.18	1.95	0.00	May - June
5ANW	1.12	1.20	0.81	May - October
5BSNE	0.77	1.18	0.44	May - September
5BSNW	0.64	0.28	0.72	May - September

Wildfire Description

A wildfire occurred on April 3, 2013, on the GRD from an escaped prescribed fire in Allotment three. The prescribed fire burned 53 ha, and the wildfire burned approximately 4 047 ha of range and pasture lands in portions of Allotment three, four, and five, along with private land (Figure 3.1). Prior to the prescribed burn, the average precipitation was 40% below the 30-yr long-term average (NOAA 2014). Weather conditions taken on site reported average wind speeds of 7.5 kph with gusts up 13 kph from the south, southwest. The average relative humidity was 34%, and temperatures ranged from 15-22 °C. A weather station in the area reported wind speeds averaging 20 kph with maximum wind speeds around 60 kph and temperatures ranged from -2.2-21 °C (NDAWN 2014).

Vegetative Plot Selection

We utilized data previously collected in 2008 that was randomly selected for monitoring plots. After the wildfire occurred in 2013, we stratified plots to resample four and 16 MAF from the previously monitored plots impacted by the wildfire based on vegetation and ecological sites to reduce variability. The 2008 plots were obtained by overlaying U.S. Forest Service pasture units and ecological sites in ArcGIS version 9.1. Buffers were placed around perennial water sources (200 m); roads, trails, and fence lines (50 m); and the edge of ecological site polygons (75 m). For each grazing unit 259 ha or smaller, three plots were randomly chosen in the two most dominant ecological sites outside of the buffers (Gearhart 2011).

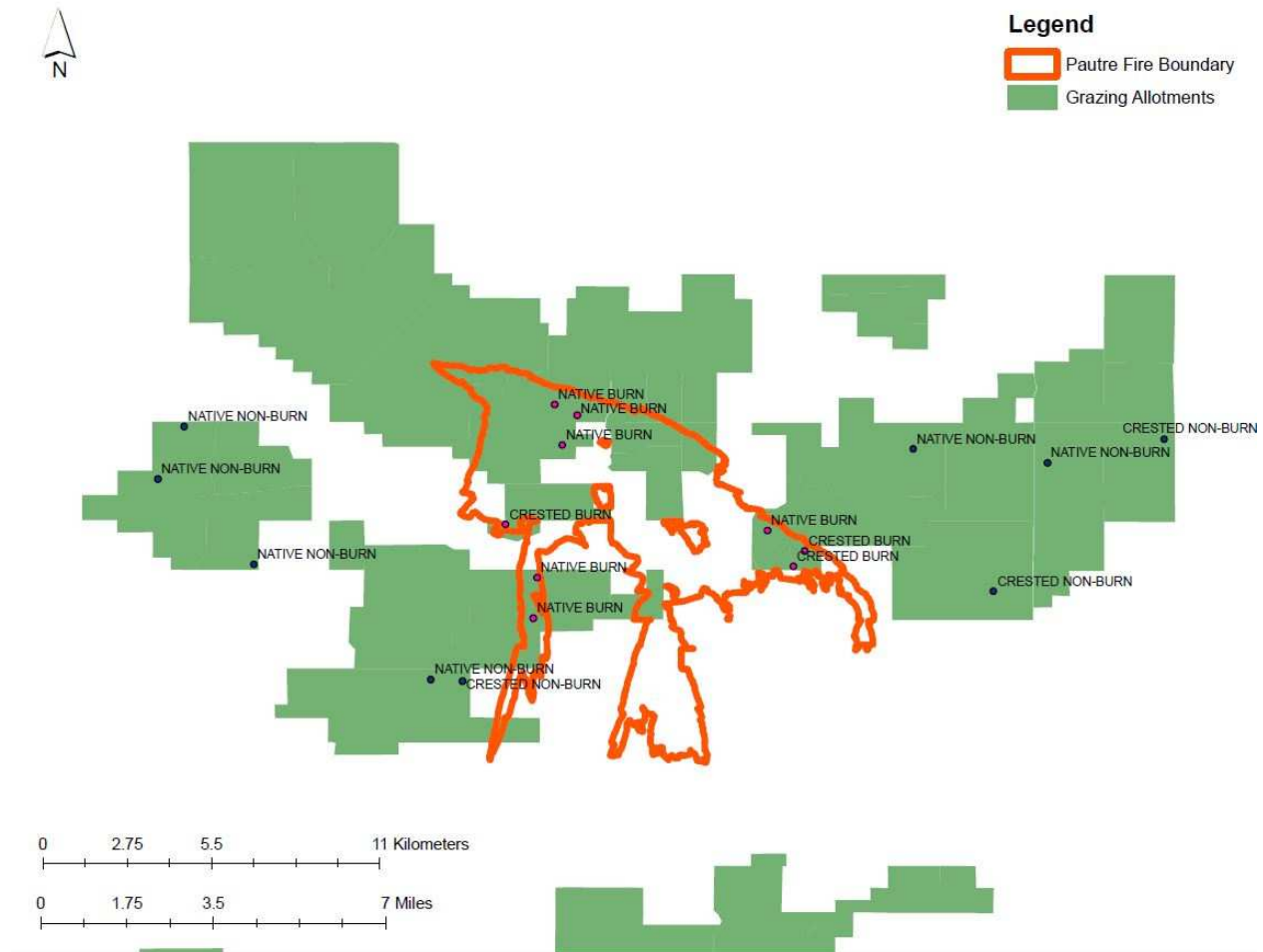


Figure 3.1. Pautre fire outline with USFS Grazing Allotments and 2008, 2013, and 2014 sampled plots. The fire outline covers approximately 4 047 ha of private and public lands near Lemmon, SD, USA.

From the plots surveyed in 2008, we chose nine plots from inside the fire boundary and nine similar plots outside the fire boundary to create a natural experiment. We based similarity on the plant species composition from frequency and production data output collected in 2008 (Gearhart 2011). We systemically chose three replicates for each fire status (burn and non-burn) from native sandy, native shallow sandy, and crested wheatgrass sandy plots because sandy and shallow sandy ecological sites were the most numerous types available. We relocated the plots using the original GPS coordinates and a Trimble June 3B (© Trimble Navigation Limited) GPS system.

Vegetative Data Sampling

We set up a 150 m perpendicular transects for each plot, with 75 m as the center point. We determined the ecological site by soil texture and horizon depths after digging a hole to 50 cm. We collected frequency and density data using a 0.1 m² nested frame within a 0.25 m² frame (Dix 1958; Biondini et al. 1989) every 10 m. We collected ground cover (live, litter, and bare ground) every 5 m using a 10-pin point frame (Levy and Madden 1933; Tinney et al. 1937; Bonham 1989). Phytomass production was clipped by plant species at 30, 60, 90, and 120 m using a 0.178 m² hoop for a total of 8 points per plot. We oven dried the samples at 60 °C for 72 hours, weighed the dried biomass, and averaged the weight over each plot.

Soil Nutrient Sampling

We set up three study blocks (used as replicates) in three separate pastures using 2.44 m cattle panels to create 214 m² enclosures. Enclosures were stratified on the fire line, one being on the burned side and the other on the non-burned side to create a naturally occurring design. Enclosures had a complete randomized block design with four defoliation treatments: non-defoliation (control), spring defoliation (June 12, 2013), summer defoliation (August 7, 2013), and fall defoliation (November 6, 2013). Vegetation was defoliated to a stubble height of 8.4 cm with a walk-behind lawnmower. We collected all plant clippings. Our study focused on the no defoliation and spring defoliation treatments.

We installed four sets of plant root simulator (PRS) TM probes in pairs (anion and cation) in the no defoliation and spring defoliation treatments, with each enclosure having a total of 16 probes. Probes were installed per manufacturer's instructions by vertically placing them in soil cuts that were backfilled to

create adequate contact between the soil and membrane approximately 10 cm below the surface (Western Ag 2014). We installed the first set on August 6, 2013, removing them on October 27, 2013, for 82 days. During the spring of 2014, probes were installed on May 10, 2014, and removed July 7, 2014, for 57 days. The last set of probes were installed on July 7, 2014, and removed September 5, 2014, for 61 days. We sent PRS probes back to Western Ag Innovations (Saskatoon, SK, CAN) for complete analyses (NO_3^- -N, NH_4^+ -N, P, K, S, Ca, Mg, Mn, Al, Fe, Cu, Zn, B, Pb, and Cd) using automated flow injection analysis and inductively-coupled plasma spectrometry (Liang and Schoenau 1995; Qian and Schoenau 1995; Qian et al. 1996). Results were reported as $\mu\text{g} \cdot 10 \text{ cm}^2 \cdot \text{length of burial}$ (Western Ag 2014).

Statistical Analysis

We used PC-ORD version 6.0 to compare species frequency and density data between treatments (burn and non-burn) and between years (2008, 2013, and 2014) with a permutation-based non-parametric multivariate analysis of variance (perMANOVA) using Relative Sorensen Distance Measures and a two-way factorial for burn treatment and year. We used 4 999 runs starting at the time of day for the source of random number seeds. We also used non-metric multi-dimensional scaling (NMS) in PC-ORD version 6.0 for plant species ordination (McCune and Mefford 2011) using the auto pilot slow and thorough setting with Relative Sorensen distance measures. Species diversity indices (richness, evenness, Shannon Diversity, and Simpson's Index) were compared using univariate generalized linear modeling (UNIANOVA) in SPSS version 14.0 (SPSS 2005). We used two-tailed Student's *t* tests ($\alpha = 0.05$) to compare ground cover and production between years and between treatments separately (Steel et al. 1996).

We analyzed soil nutrient data dependently using a two-way factorial perMANOVA using Relative Sorensen Distance Measures in PC-ORD version 6.0 for defoliation and fire treatments. We used 4 999 runs starting at the time of day for the source of random number seeds (McCune and Mefford 2011). We independently analyzed soil nutrients with an UNIANOVA and post-hoc Tukey tests in SPSS version 14.0 (SPSS 2005).

Results

Vegetative Species Composition

There were no changes ($p > 0.05$) to species composition on any of the sites due to the fire treatment four and 16 MAF. Sandy native sites had a year and treatment effect ($p \leq 0.05$), but no year x treatment interaction effects ($p > 0.05$). Sandy crested wheatgrass sites had a year effect ($p \leq 0.05$), but the treatment and year x treatment interaction effects showed no difference ($p > 0.05$) between burned and non-burned sites. Native shallow sandy sites had a treatment effect ($p \leq 0.05$), but there was no year or year x treatment interaction effects ($p > 0.05$) (Table 3.1). Even when graminoids and forbs species were separated into groups, there were no interactions between year and treatment ($p > 0.05$).

Table 3.2. Sandy and shallow sandy ecological site perMANOVA results for species composition on both native rangelands and crested wheatgrass pasturelands on the GRD, Perkins County, South Dakota, USA. Data collected from 2008, 2013, and 2014 for graminoid frequency and forb/shrub density.

Ecological Site	Dominant Vegetation	Source	<i>p</i> -value
Sandy			
	Native	Year	0.036
		Treatment ¹	0.037
		Interaction	0.962
	Crested Wheatgrass	Year	0.049
		Treatment ¹	0.057
		Interaction	0.972
Shallow Sandy			
	Native	Year	0.066
		Treatment ¹	0.036
		Interaction	0.839

¹ Treatments were burned and non-burned.

The NMS graphics show the same interaction effects since treatments did not group separately for sandy native, sandy crested wheatgrass, and shallow sandy native sites (Figures 3.1 - 3.3). Plots from both treatments overlap on all ecological sites and are grouped together. Sandy crested wheatgrass plots grouped by year (Figure 3.4), instead of treatment, showed some differentiation for plant species composition between 2008 and 2013, and 2014. Successional vectors also go in various directions and do not follow specific patterns for year or treatment (Figures 3.5 – 3.7). Individual plant species were not analyzed because they were not considered independent from each other. Major species that seem to be

driving the system are bluegrasses (*Poa spp.*), smooth brome (*Bromus inermis* Leyss.), and sweetclover (*Melilotus officinalis* (L.) Lam.) (data not shown).

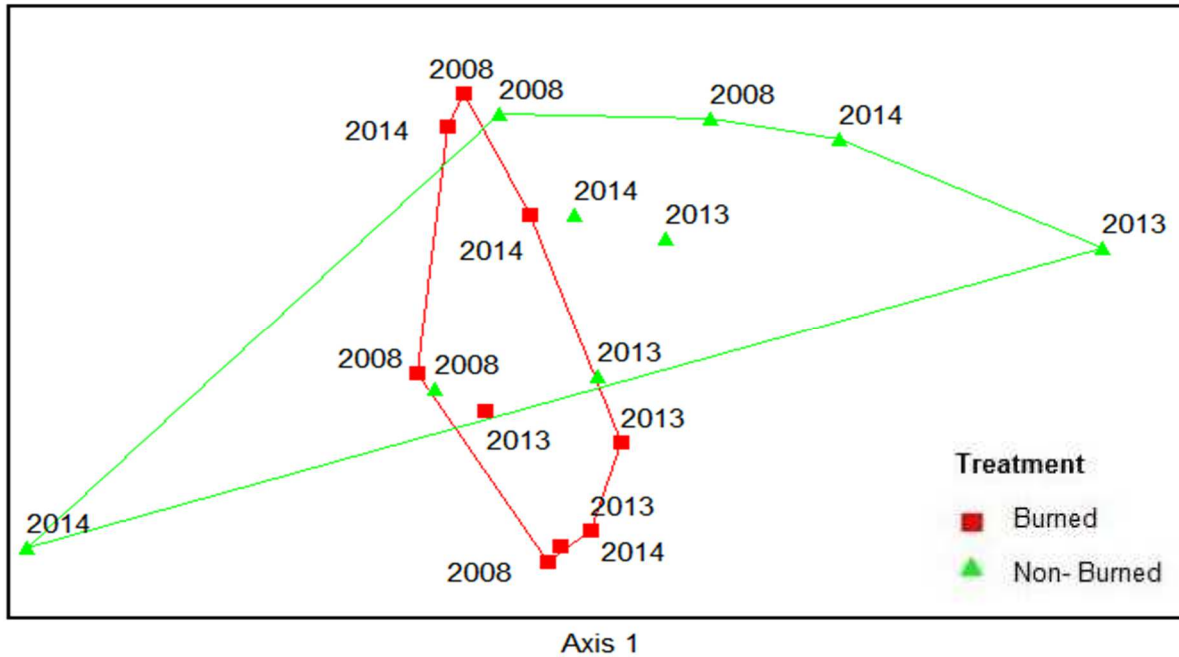


Figure 3.2. Sandy native plots for plant species composition ordination for all sampled plots on the GRD, Perkins County, South Dakota, USA, by treatment. Data collected was from 2008, 2013, and 2014. The ordination found a one dimensional solution. Axis 1 had an r^2 value of 0.702.

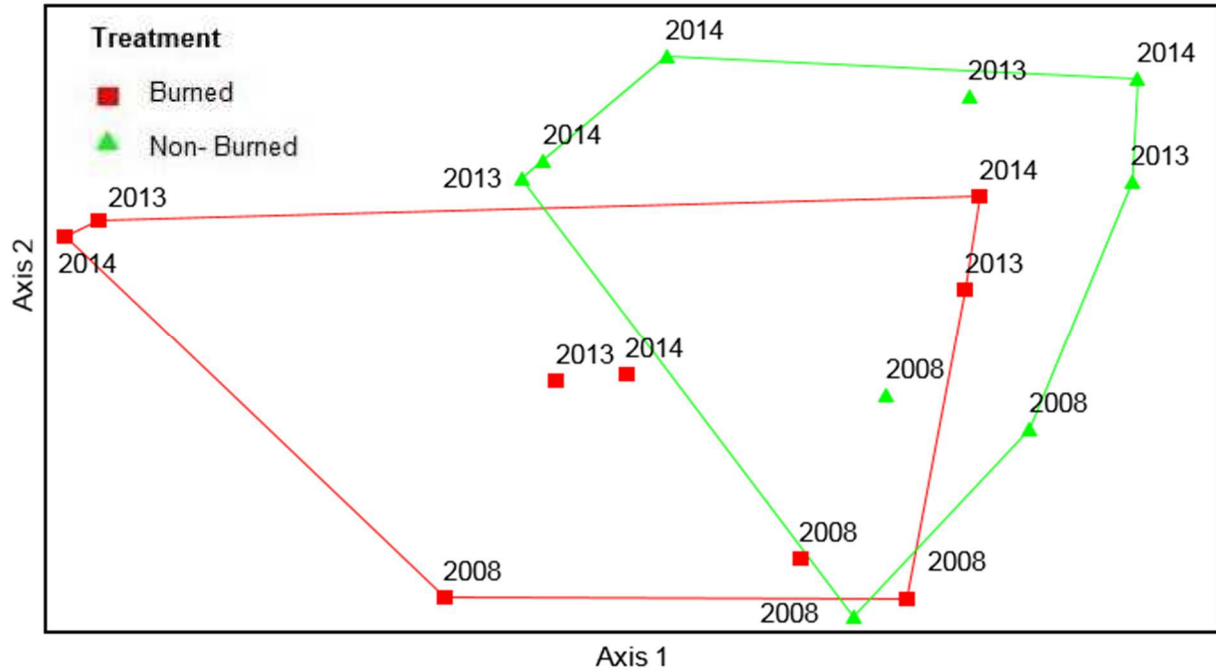


Figure 3.3. Sandy crested wheatgrass plots ordination for plant species composition for all sampled plots on the GRD, Perkins County, South Dakota, USA, by treatment. Data collected was from 2008, 2013, and 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.548. Axis 2 had an r^2 value of 0.225.

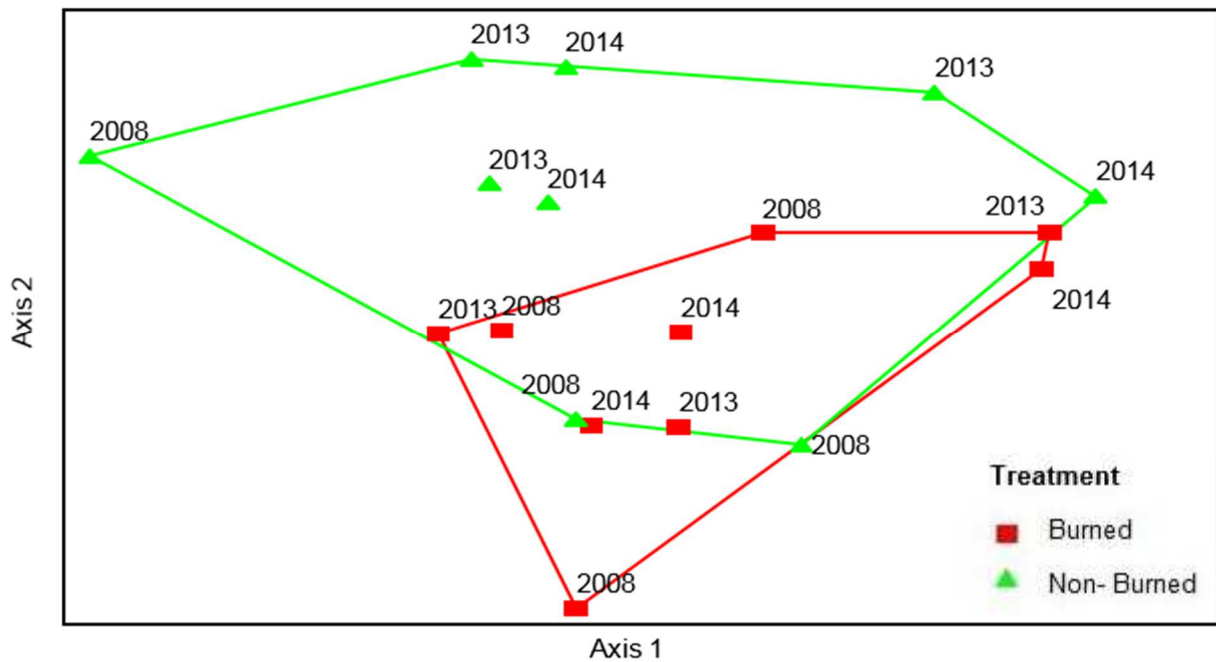


Figure 3.4. Shallow sandy native plots ordination for plant species composition for all sampled plots on the GRD, Perkins County, South Dakota, USA, by treatment. Data collected was from 2008, 2013, and 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.399. Axis 2 had an r^2 value of 0.376.

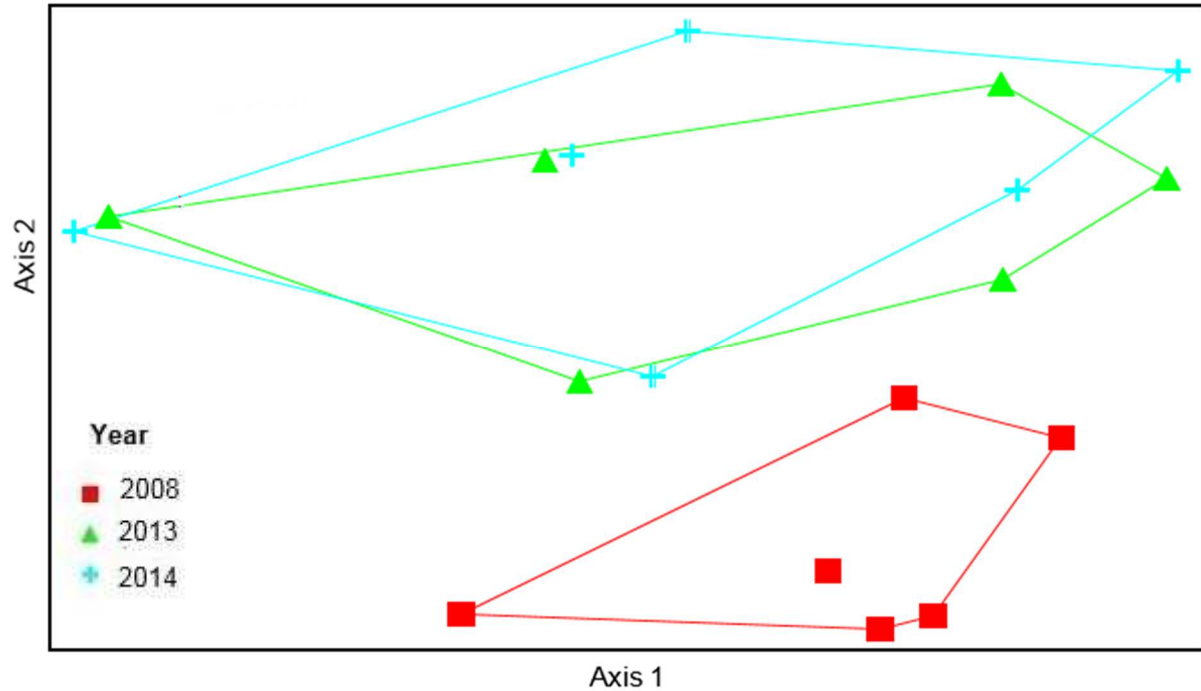


Figure 3.5. Sandy crested wheatgrass ordination for plant species composition on all sampled plots on the GRD, Perkins County, South Dakota, USA, by year. Data collected was from 2008, 2013, 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.548. Axis 2 had an r^2 value of 0.225.

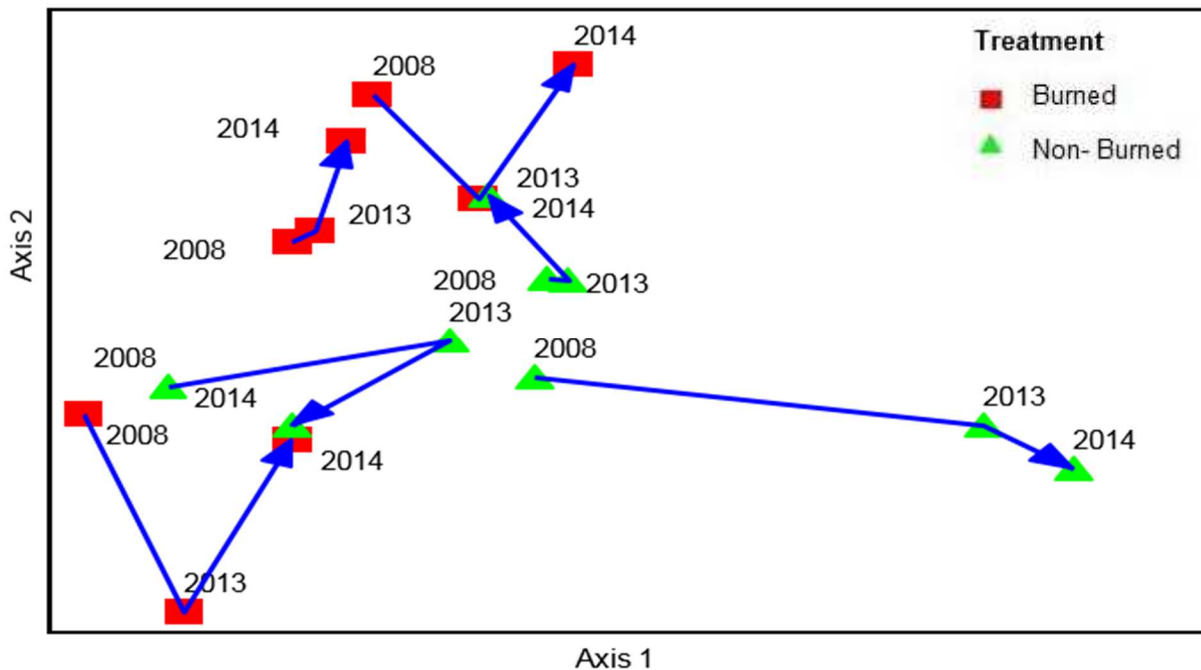


Figure 3.6. Successional vectors for sandy native plots for plant species composition for all sampled plots on the GRD, Perkins County, South Dakota, USA. Data collected was from 2008, 2013, and 2014. Arrows show the succession from 2008 to 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.530. Axis 2 had an r^2 value of 0.246.

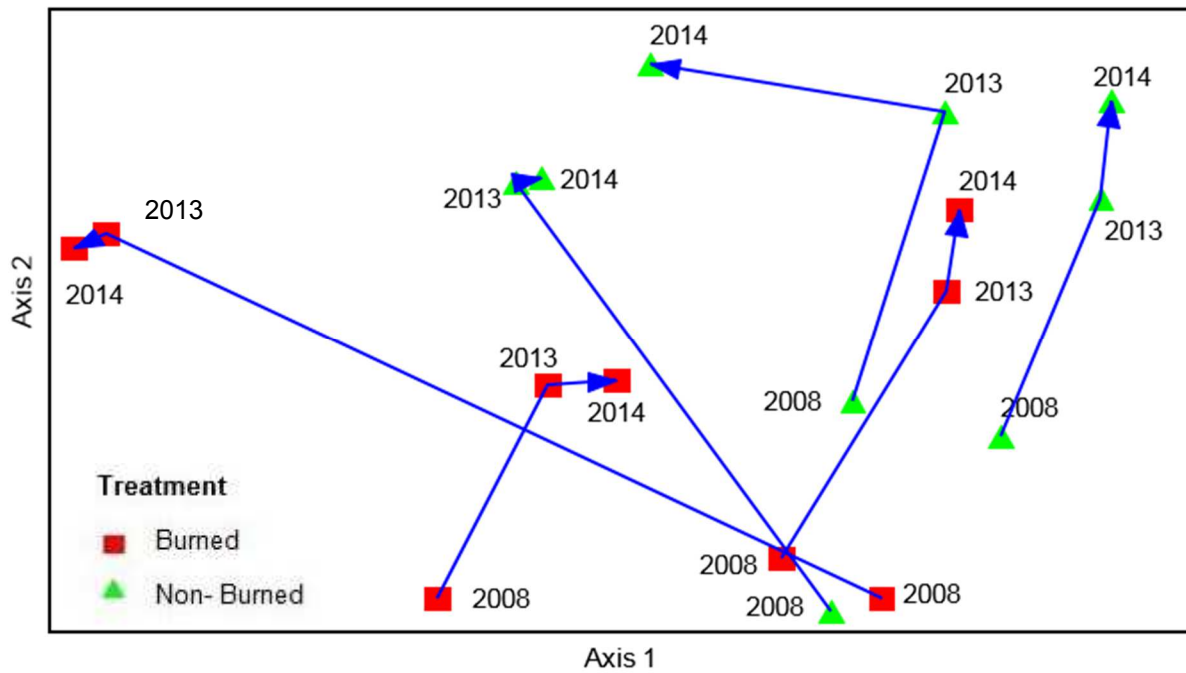


Figure 3.7. Successional vectors for sandy crested wheatgrass plots for plant species composition on all sampled plots on the GRD, Perkins County, South Dakota, USA. Data collected was from 2008, 2013, and 2014. Arrows show the succession from 2008 to 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.548. Axis 2 had an r^2 value of 0.225.

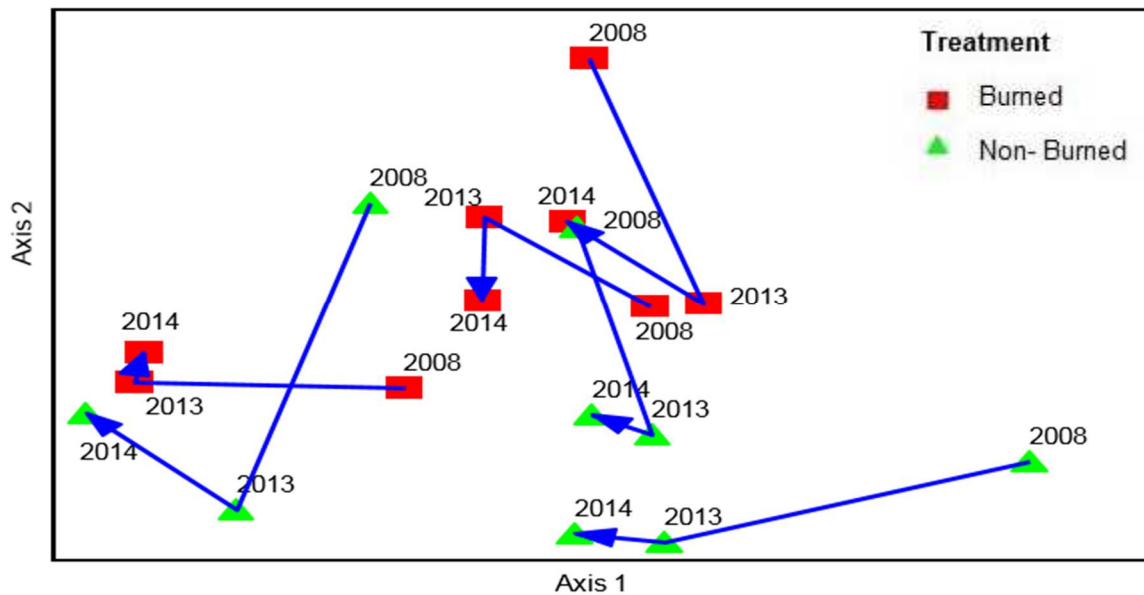


Figure 3.8. Successional vectors for shallow sandy native plots for plant species composition on all sampled plots on the GRD, Perkins County, South Dakota, USA. Data collected was from 2008, 2013, and 2014. Arrows show the succession from 2008 to 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.395. Axis 2 had an r^2 value of 0.382.

Ground Cover

The perMANOVA analysis for ground cover (live, litter, and bare ground) showed sandy native plots had a year and year x treatment interaction effect ($p \leq 0.05$) but no treatment effect ($p > 0.05$). The sandy crested wheatgrass plots only had a year effect ($p \leq 0.05$). The shallow sandy plots had year, treatment, and year x treatment interaction effect ($p \leq 0.05$) for ground cover (Table 3.2).

Table 3.3. Sandy and shallow sandy ecological site perMANOVA results for both native rangelands and crested wheatgrass pasturelands ground cover on the GRD, Perkins County, South Dakota, USA. Data collected was from 2008, 2013, and 2014.

Ecological Site	Dominant Vegetation	Source	p-value
Sandy	Native	Year	0.008
		Treatment ¹	0.099
		Interaction	≤ 0.001
	Crested Wheatgrass	Year	0.009
		Treatment ¹	0.065
		Interaction	0.097
Shallow Sandy	Native	Year	≤ 0.001
		Treatment ¹	0.003
		Interaction	0.024

¹ Treatment represents burn and non-burned treatments.

All of the different ecological sites and vegetation types had one strongly correlated axis with r^2 values greater than 0.8 in NMS. Basal litter was strongly correlated ($r^2 > \pm 0.900$) with non-burned plots and bare ground was strongly correlated ($r^2 > \pm 0.900$) with burned plots for sandy native, shallow sandy native, and sandy crested wheatgrass plots. Live basal cover was also strongly correlated ($r^2 = -0.734$) with burned plots on sandy crested wheatgrass plots (Figure 3.8).

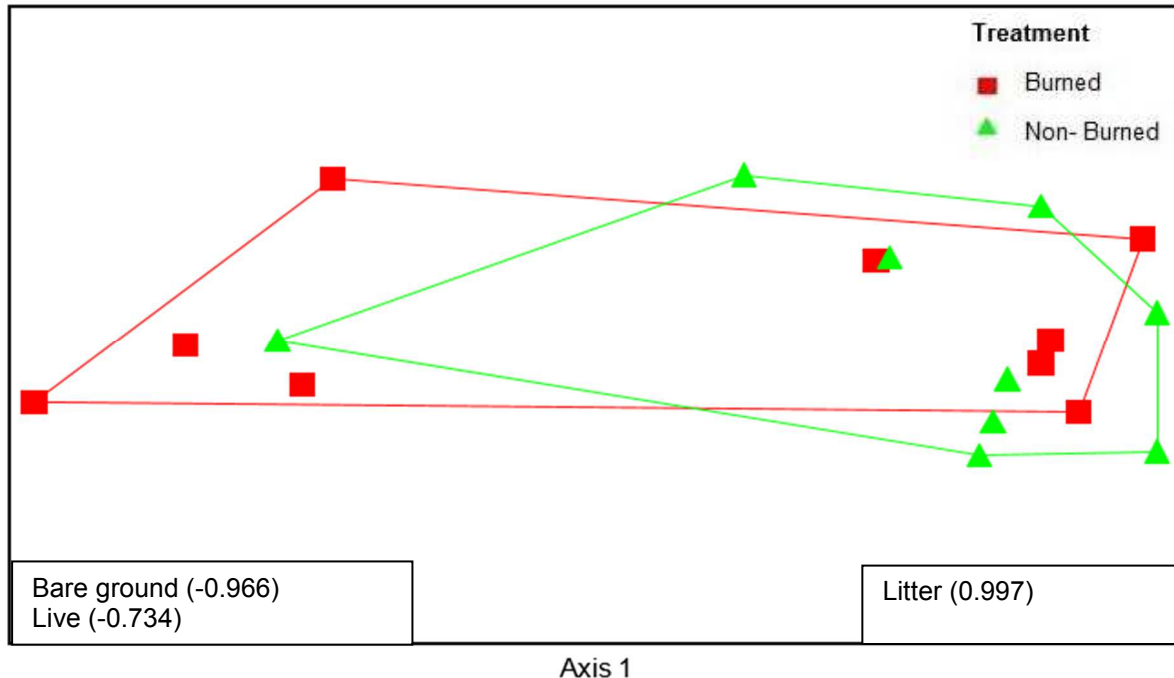


Figure 3.9. Ordination for sandy ecological sites with crested wheatgrass vegetation depicting ground cover for all sampled plots on the GRD, Perkins County, South Dakota, USA, by treatment. Data collected was from 2008, 2013, and 2014. Correlation scores for litter, bare ground, and live cover are included. The ordination found a one-dimensional solution. Axis 1 had an r^2 value of 0.981.

In 2008, before the wildfire, there was more ($p \leq 0.05$) bare ground on shallow sandy native plots classified as burned compared to non-burned plots. Ground cover did not differ ($p > 0.05$) between sandy native and sandy crested wheatgrass plots in 2008. There was no change ($p > 0.05$) in live cover or crested wheatgrass cover due to fire treatment four or 16 MAF. There was more ($p \leq 0.05$) bare ground on all burned plots four MAF in 2013 compared to non-burned plots in 2013. There was no difference ($p > 0.05$) in bare ground 16 MAF in 2014 between burned and non-burned plots (Figure 3.10). The average bare ground on sandy native plots was 29 percent in 2013. Sandy crested wheatgrass plots had similar bare ground at 24 percent. Shallow sandy native plots had the highest amount of bare ground at 40 percent in 2013. In 2014, 16 MAF, bare ground was three, five, and 13 percent on sandy native, sandy crested wheatgrass, and shallow sandy native, respectively.

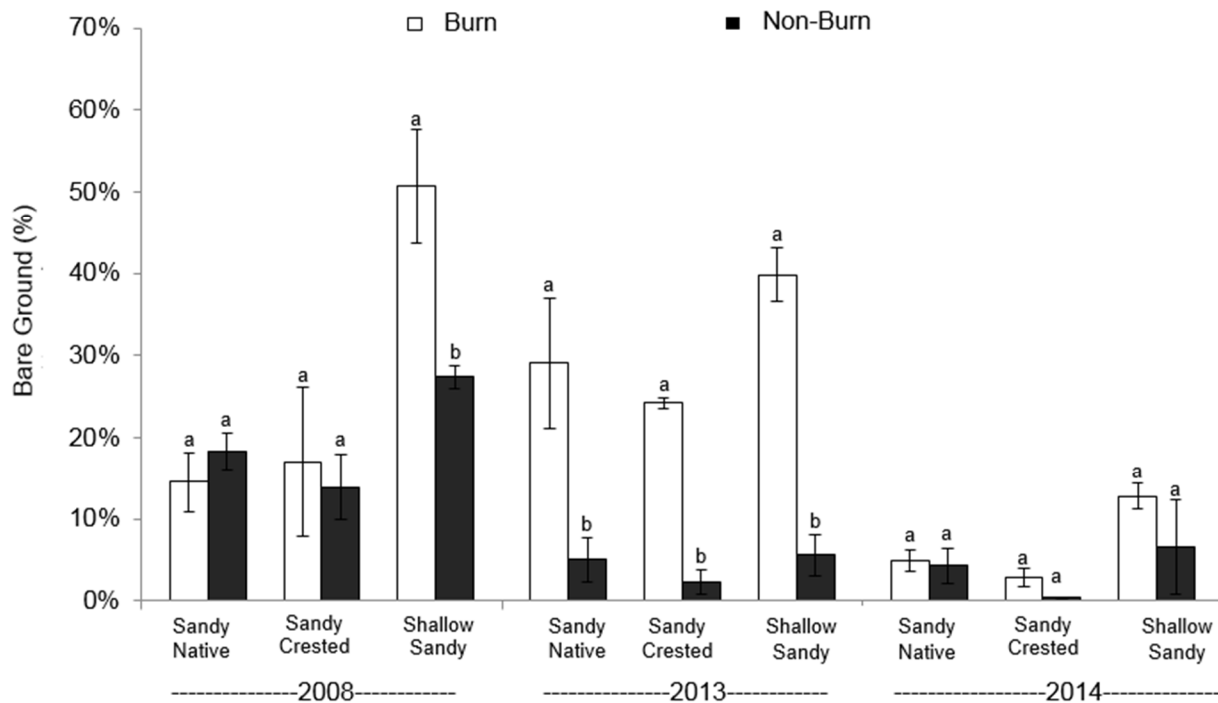


Figure 3.10. Bare ground percentages for all ecological sites and vegetation types for burned and non-burned treatments in 2008, 2013, and 2014 on the GRD, Perkins County, South Dakota, USA. Standard error bars are included and centered on the average bare ground for each group. Different letters differ at $p \leq 0.05$ within grouped bars.

In 2008, before the wildfire, litter basal cover amounts were not different ($p > 0.05$) on all plots. Basal litter decreased ($p \leq 0.05$) on all burned sites four MAF in 2013. There was a difference ($p \leq 0.05$) in litter cover 16 MAF in 2014 on sandy native plots, but there was more ($p \leq 0.05$) litter on burned plots than non-burned plots. There was no difference ($p > 0.05$) in litter basal cover 16 MAF on sandy crested wheatgrass and shallow sandy native plots (Figure 3.11). Basal litter cover decreased 21, 13, and two percent four MAF; however, litter basal cover was 93, 96, and 82 percent 16 MAF for sandy native, sandy crested wheatgrass, and shallow sandy native, respectively.

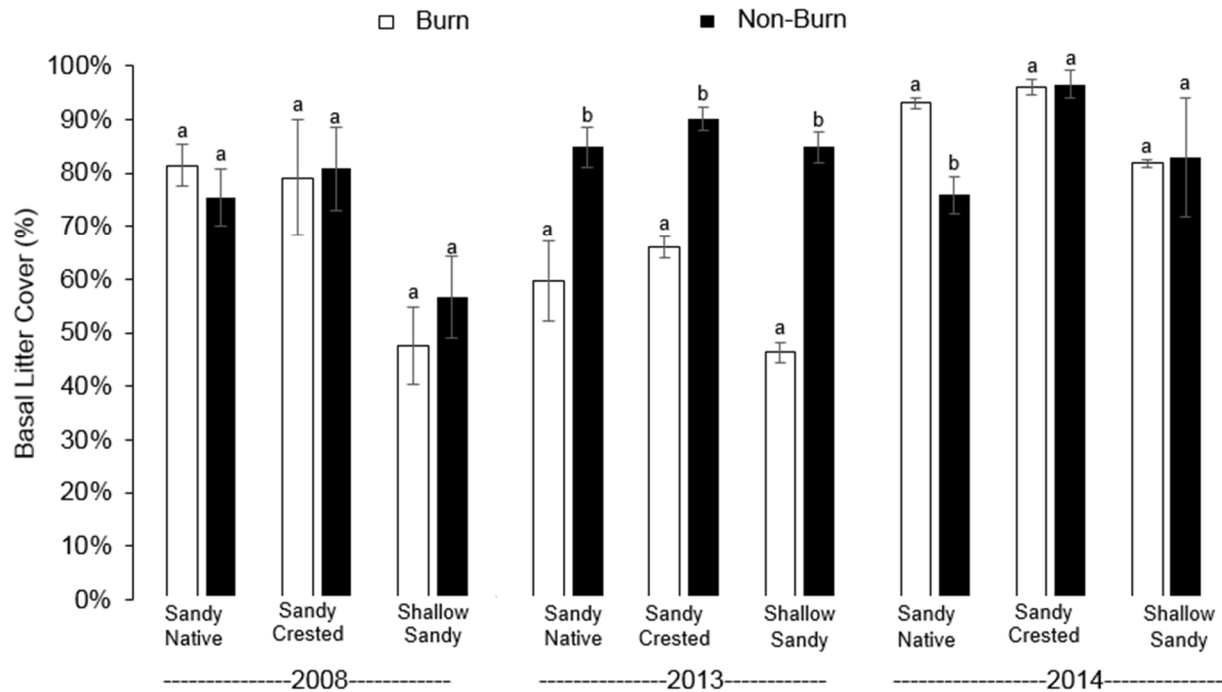


Figure 3.11. Basal litter cover percentages for all ecological sites and vegetation types for burned and non-burned treatments in 2008, 2013, and 2014 on the GRD, Perkins County, South Dakota, USA. Standard error bars are included and centered on the average bare ground cover point for each group. Different letters differ at $p \leq 0.05$ within grouped bars.

Phytomass Production

There was no difference ($p > 0.05$) in phytomass production between similar plots classified as burned and non-burned in 2008 before the wildfire. Fire had no effect ($p > 0.05$) on annual phytomass production four or 16 MAF. There were some differences in the annual production between years (Figure 3.12). Sweetclover was a predominant component of the annual production in 2014. Approximately 50 percent of the plots had sweetclover clipped for annual production. On average, sweetclover provided approximately $600 \text{ kg} \cdot \text{ha}^{-1}$ of production on each site where it occurred. Sweetclover produced on average $172 \text{ kg} \cdot \text{ha}^{-1}$ in 2013. Fire did affect ($p \leq 0.05$) the amount of litter (basal detached litter and standing attached litter) on burned and non-burned plots in 2013. Litter production decreased four MAF on sandy native and sandy crested wheatgrass plots. Sandy native burned plots produced an average of $82 \text{ kg} \cdot \text{ha}^{-1}$ of litter, while non-burned plots produced approximately $1120 \text{ kg} \cdot \text{ha}^{-1}$. Sandy crested wheatgrass burned plots produced on average $115 \text{ kg} \cdot \text{ha}^{-1}$ of litter, and non-burned plots produced

around 1 850 kg ·ha⁻¹. These differences ($p \leq 0.05$) continued 16 MAF. Native shallow sandy sites also had large decreases ($p \leq 0.05$) in litter weight four MAF, which continued ($p \leq 0.05$) 16 MAF. The average litter production continually increased from 2008 to 2014 on non-burned plots. Litter production on average for non-burned plots was 1 115 kg ·ha⁻¹ in 2008. Litter production increased in 2013 to 1 436 kg ·ha⁻¹ and 1 774 kg ·ha⁻¹ in 2014. Litter production on average for burned plots in 2008 was 778 kg ·ha⁻¹. Litter production decreased in 2013 to 145 kg ·ha⁻¹ and increased to 721 kg ·ha⁻¹ in 2014.

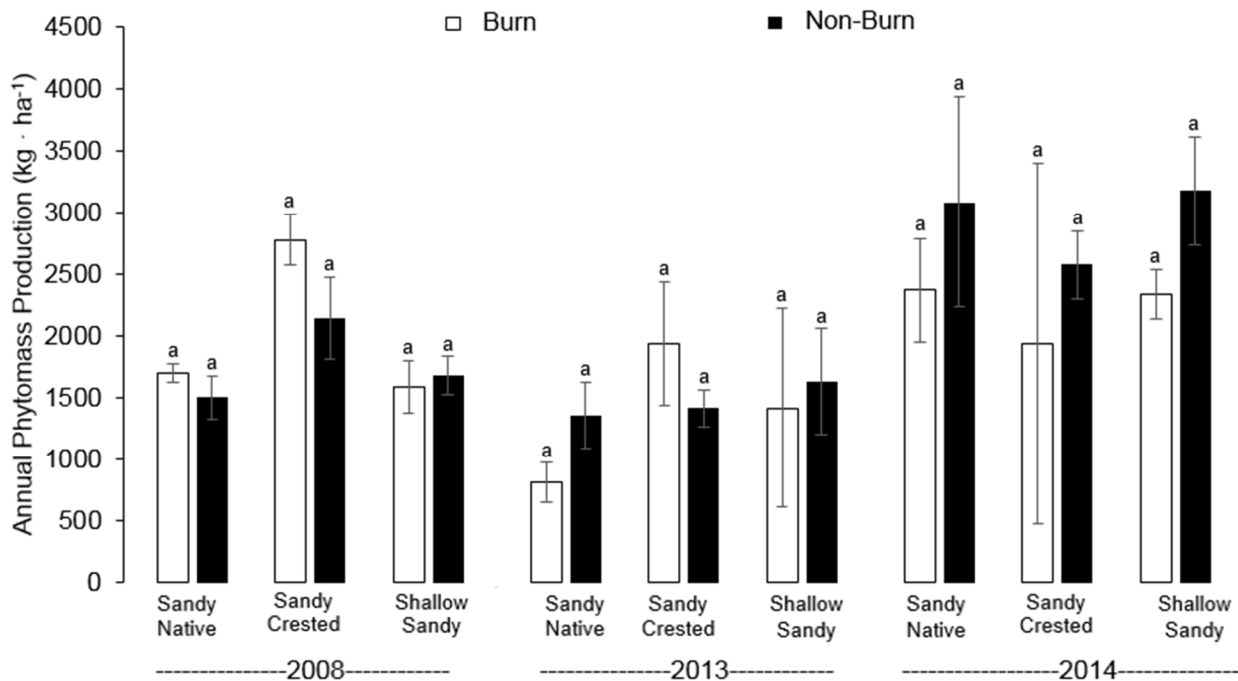


Figure 3.12. Annual phytomass production for 2008, 2013, and 2014 on burned and non-burned treatments for each ecological site and vegetation on the GRD. No differences ($p > 0.05$) were found between treatments in each year between grouped bars.

Soil Nutrients

Defoliation and fire treatments did not affect ($p > 0.05$) soil nutrient amounts five, 14, and 16 MAF. NMS ordination also did not show fire x defoliation treatment interactions (Figure 3.13), defoliation effects (Figure 3.14), fire effects (Figure 3.15), or date effects (Figure 3.16).

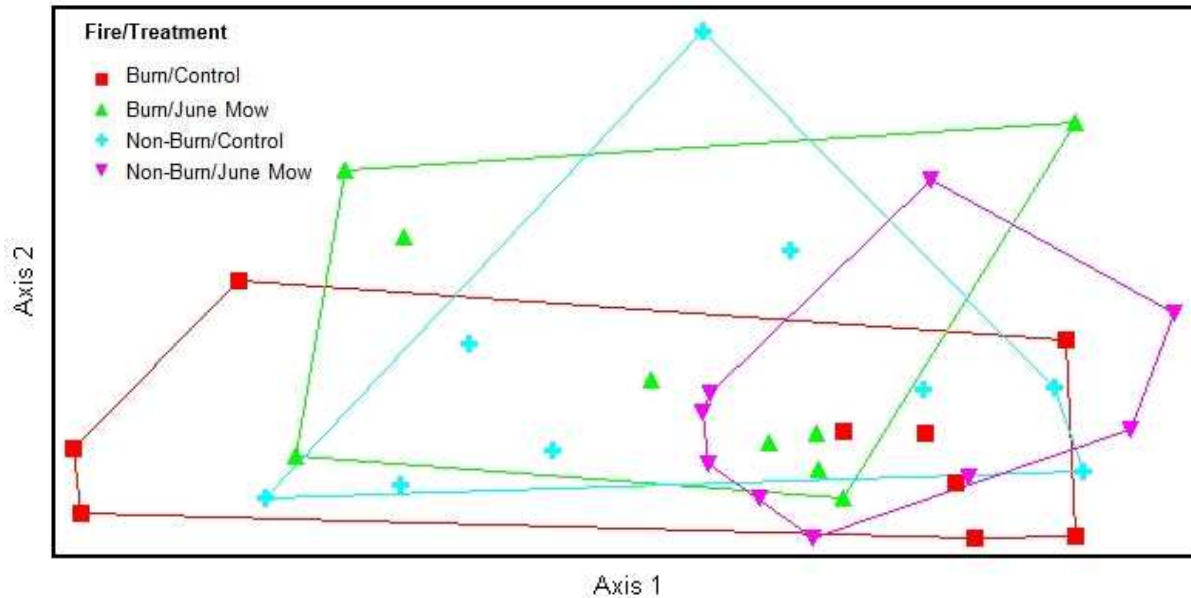


Figure 3.13. Ordination for soil nutrients on no defoliation and June defoliation treatments within burned and non-burned plots on the GRD 4-16 months after fire in 2013 and 2014. Axis 1 had an r^2 value of 0.897. The ordination found a two-dimensional solution. Axis 2 had an r^2 value of 0.089.

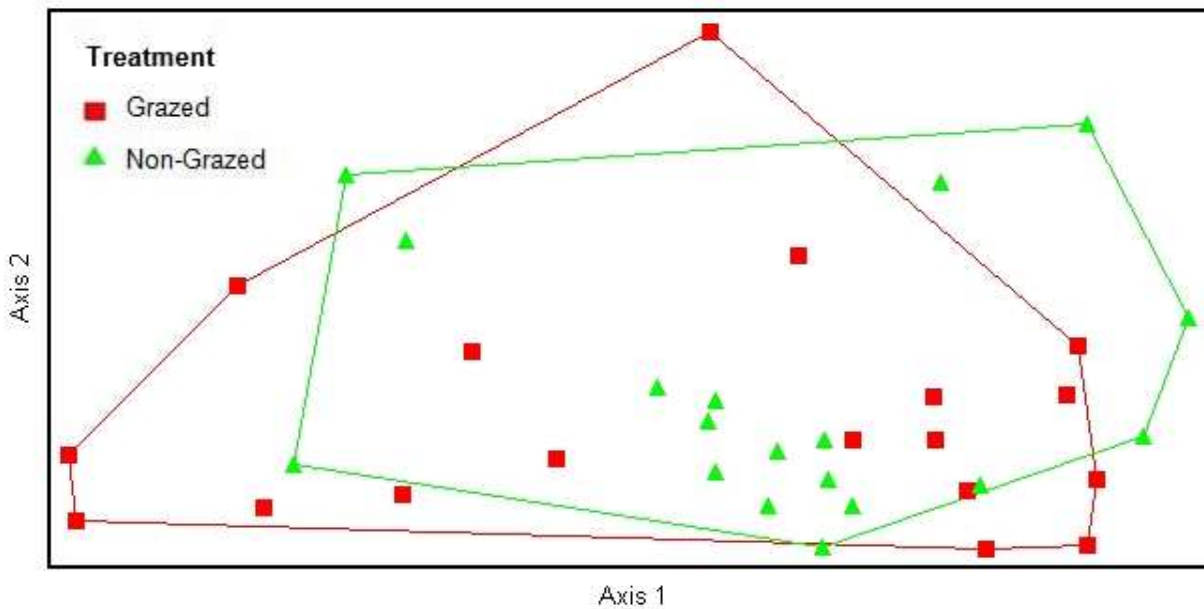


Figure 3.14. Ordination for soil nutrients on June defoliated and non-defoliated treatments on the GRD 4-16 months after fire in 2013 and 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.897. Axis 2 had an r^2 value of 0.089.

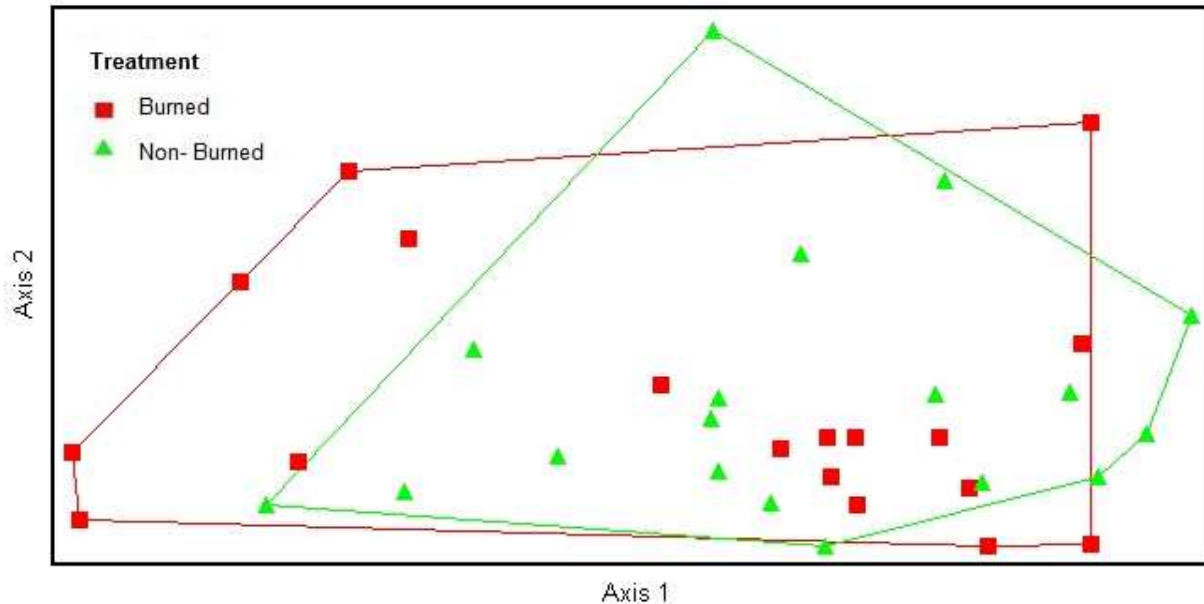


Figure 3.15. Ordination for soil nutrients on burned and non-burned treatments on the GRD 4-16 months after fire in 2013 and 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.897. Axis 2 had an r^2 value of 0.089.

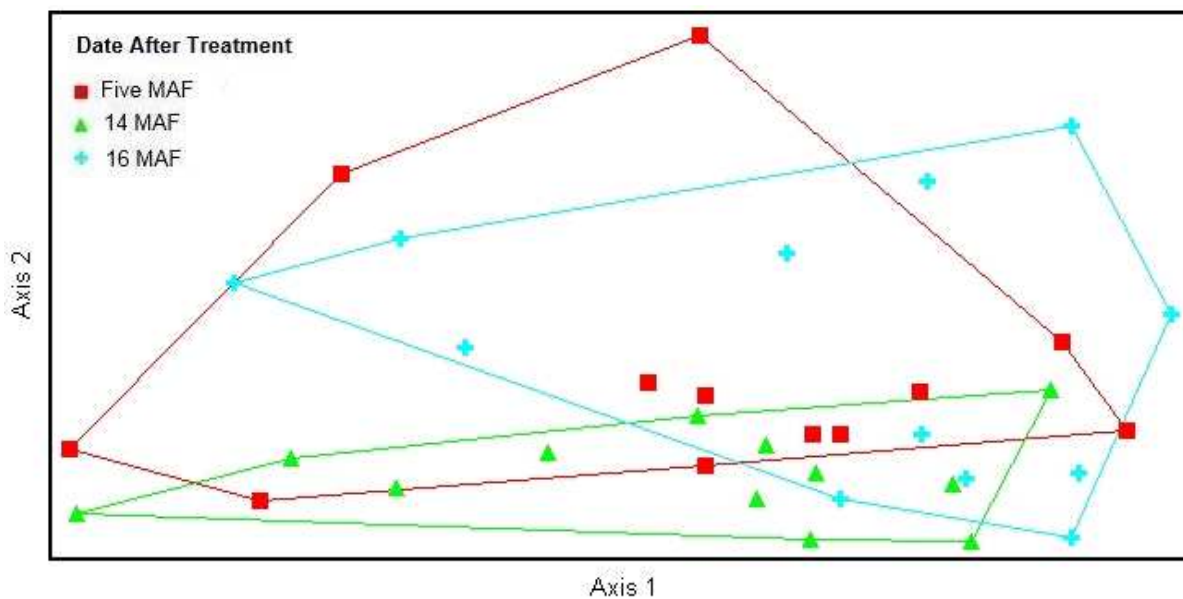


Figure 3.16. Ordination for soil nutrients by five, 14, and 16 month after fire (MAF) on the GRD for 2013 and 2014. The ordination found a two-dimensional solution. Axis 1 had an r^2 value of 0.897. Axis 2 had an r^2 value of 0.089.

Soil nutrients analyzed independently showed inconsistent responses to treatments, defoliated, non-defoliated, burned, or non-burned. There was no difference ($p > 0.05$) for boron and lead in regards to fire treatments, defoliation treatments, date since fire, or any interaction. Nitrate was the main driver of

total nitrogen amounts. It comprised an average 65 percent of total weight per area per sampling period. Ammonium was higher ($p = 0.054$) on non-burned plots compared to burned plots, regardless of defoliation treatment. Phosphorus levels were higher ($p = 0.053$) on non-defoliated treatments compared to June defoliation treatments, but within each defoliation treatment, levels were higher on burned plots compared to non-burned plots.

Calcium was higher ($p = 0.055$) on burn only treatments compared to burn followed by June defoliation treatments. There was a fire x date interaction ($p = 0.055$) with more calcium five MAF compared to other sampling dates. The 14 and 16 MAF samples had more calcium on burned plots. There was a defoliation x fire x date interaction effect ($p = 0.05$) for magnesium that was highest five MAF on non-burned plots on June defoliation treatments, but June defoliation treatments tended to have higher amounts. Magnesium tended to be lower on non-burn, non-defoliation treatments. There was also a fire x date interaction where magnesium was higher ($p = 0.051$) 16 MAF on non-burned plots. Non-burned plots five and 16 MAF had larger amounts of magnesium than burned plots from the same sampling period and either plot 14 MAF. Non-burned plots had more magnesium compared to burned plots 14 MAF. There was more ($p = 0.05$) potassium on June defoliation treatments compared to non-defoliation treatments, but no interactions with date or fire treatments.

Iron tends to be higher ($p = 0.058$) on June defoliation treatments, especially 14 MAF. June defoliation treatments had more iron than control treatments 16 MAF. Copper trends showed a fire x date interaction that decrease ($p = 0.052$) from time since fire, and burned plots had higher copper concentrations compared to non-burned plots within each sampling period. Zinc trends showed increased ($p = 0.056$) amounts on non-burned plots compared to burned plots. There was a defoliation x fire x date interaction ($p = 0.056$) for aluminum that tended to be higher on non-defoliation treatments on burned and non-burned sites closer to the fire event. There was a defoliation x fire interaction effect ($p = 0.05$) for cadmium that showed higher ($p = 0.05$) concentrations on burned plots compared to non-burned plots, but non-defoliation treatments were higher within burned and non-burned plots compared to June defoliation treatments. Cadmium tended to be higher ($p = 0.052$) on non-defoliation treatments compared to June defoliation treatments (Table 3.4-3.6).

Table 3.4. Soils nutrients five months after fire for defoliation treatments (non-defoliation, June defoliation) and burn treatments (burn, non-burn) on the GRD in 2013.

	Burn, Non-Defoliation	Non-Burn, Non-Defoliation	Burn, June Defoliation	Non-Burn, June Defoliation
Soil Nutrient	-----µg · 10 cm ² · length of burial-----			
Total N	33.81	60.67	58.18	25.29
NO3-N	28.69	55.51	52.39	20.24
NH4-N	5.13	5.15	5.79	5.05
Ca	2 602.05	2 820.79	3 039.15	2 414.18
Mg	337.46	331.88	384.02	445.25
K	86.46	80.43	84.10	98.08
P	7.04	10.25	9.76	5.26
Fe	2.98	5.62	5.61	3.02
Mn	2.77	2.54	5.15	1.80
Cu	0.46	0.47	0.67	0.44
Zn	0.94	1.49	1.19	0.81
B	0.47	1.28	0.51	0.49
S	25.29	29.59	34.81	18.20
Pb	0.23	0.39	0.57	0.11
Al	15.21	21.00	21.84	24.43
Cd	0.09	0.03	0.14	0.01

Table 3.5. Soils nutrients 14 months after fire for defoliation treatments (non-defoliation, June defoliation) and burn treatments (burn, non-burn) on the GRD in 2014.

	Burn, Non-Defoliation	Non-Burn, Non-Defoliation	Burn, June Defoliation	Non-Burn, June Defoliation
Soil Nutrient	-----µg · 10 cm ² · length of burial-----			
Total N	9.25	14.05	15.09	10.18
NO3-N	3.54	5.25	8.23	4.31
NH4-N	5.71	8.80	6.86	5.87
Ca	2 419.06	2 425.60	2 385.12	2 141.18
Mg	353.75	334.17	329.05	372.78
K	86.83	86.98	78.21	64.62
P	11.51	4.18	3.52	5.17
Fe	8.08	3.29	7.73	4.33
Mn	3.02	1.51	2.03	1.47
Cu	0.28	0.16	0.36	0.23
Zn	0.74	0.88	1.13	0.86
B	0.57	0.39	0.62	0.51
S	27.16	19.83	27.66	21.15
Pb	0.39	0.15	0.78	0.37
Al	17.15	16.21	17.95	15.92
Cd	0.05	0.01	0.06	0.06

Table 3.6. Soils nutrients 16 months after fire for defoliation treatments (non-defoliation, June defoliation) and burn treatments (burn, non-burn) on the GRD in 2014.

	Burn, Non-Defoliation	Non-Burn, Non-Defoliation	Burn, June Defoliation	Non-Burn, June Defoliation
Soil Nutrient	-----µg · 10 cm ² · length of burial-----			
Total N	15.85	24.29	61.61	30.79
NO3-N	9.20	18.68	56.43	24.97
NH4-N	6.65	5.61	5.18	5.81
Ca	2 295.57	2 390.58	2560.17	2 224.22
Mg	359.42	360.31	386.12	435.83
K	130.20	134.50	118.12	163.44
P	5.30	6.72	6.28	7.68
Fe	4.32	2.91	4.79	3.86
Mn	1.53	1.14	2.10	2.08
Cu	0.19	0.06	0.17	0.07
Zn	0.63	0.73	0.62	0.60
B	0.55	0.72	0.49	0.74
S	18.23	16.07	25.73	22.37
Pb	0.33	0.01	0.49	0.17
Al	19.38	22.84	19.58	22.48
Cd	0.13	0.08	0.06	0.03

The post-hoc Tukey test showed general differences between nutrients and sampling date, five, 14, and 16 MAF, independent of various treatments. There was more ($p \leq 0.05$) total nitrogen and nitrate five MAF compared to 14 MAF. There were no differences ($p > 0.05$) between the five MAF to 16 MAF samples and 16 MAF to 14 MAF samples. Zinc, manganese, and sulfur were higher ($p \leq 0.05$) five MAF compared to 16 MAF. Copper levels were higher ($p \leq 0.05$) five MAF compared to 14 and 16 MAF. Potassium was higher ($p \leq 0.05$) 16 MAF compared to five and 14 MAF (Table 3.7).

Table 3.7. Post-hoc Tukey tests for soil nutrient differences between five, 14, and 16 months after fire (MAF) on both burned and non-burned plots on the GRD for 2013 and 2014.

	5 MAF	14 MAF	16 MAF
Soil Nutrient	----- $\mu\text{g} \cdot 10 \text{ cm}^2 \cdot \text{length of burial}$ -----		
Total N	44.48 ^b	12.15 ^a	33.13 ^{ab}
NO₃-N	39.21 ^b	5.34 ^a	27.32 ^{ab}
NH₄-N	5.28 ^a	6.81 ^a	5.81 ^a
Ca	2 719.04 ^a	2 342.74 ^a	2 367.63 ^a
Mg	374.65 ^a	347.44 ^a	385.42 ^a
K	87.27 ^a	79.16 ^a	136.56 ^b
P	8.08 ^a	6.10 ^a	6.50 ^a
Fe	4.31 ^a	5.86 ^a	3.97 ^a
Mn	3.06 ^b	2.01 ^{ab}	1.71 ^a
Cu	0.51 ^b	0.26 ^a	0.12 ^a
Zn	1.11 ^b	0.90 ^{ab}	0.65 ^a
B	0.69 ^a	0.53 ^a	0.62 ^a
S	26.98 ^b	23.95 ^{ab}	20.60 ^a
Pb	0.32 ^a	0.42 ^a	0.25 ^a
Al	20.62 ^a	16.81 ^a	21.07 ^a

Discussion

Species Composition

We did not analyze by plant species since they are part of the entire plant community on the GRD. We did not find treatment x year interaction effects for species composition or species diversity indices (richness, evenness, Shannon Diversity, and Simpson's Index) on any of the ecological sites or vegetation types four and 16 MAF. Quick responses are rarely seen after a single disturbance event (Steuter 1987), but there may be a plant community response lag (Dix 1960). This lag was not evident 16 MAF, but time can be a significant factor in species compositional changes (Brockway et al. 2009). We would not expect changes to the native plant community, since the NGP is well adapted to fire disturbance (Antos et al. 1983; Steuter 1987; Redmann et al. 1993; Grant et al. 2010). Changes found between burned and non-burned sites were due to site differences found naturally between sites in various pastures and between years.

We did look at overall trends in species composition. An early spring fire may be expected to favor warm-season grasses (White and Currie 1983; Redmann et al. 1993; Howe 1995; Shay et al. 2001),

but we found no change from fire on overall cool- and warm-season graminoid frequency. This has also been found after summer fires in the NGP (Vermeire et al. 2011). All of our sites were on sandy and shallow sandy ecological sites. This could also explain some of the results. Sands range sites in Canada were also found not to change from short-term fire effects. Sandier soils encouraged roots to grow deeper. Consequently, belowground biomass was farther from the surface and did not experience as much heat stress (Smith and McDermid 2014). Some upland sites can experience more damage from fire because they do not have additional litter insulating plants compared to lowland sites (Whisenant and Uresk 1989). This same litter can increase the fire intensity of an area (Snyman 2003). Responses are dependent on pre-fire conditions, fire intensity and severity, and climatic conditions (Higgins et al. 1986).

There were some slight trends in individually plant species frequency that were most likely tied to site differences, but previous research has connected individual plant species responses to fire. Blue grama tended to remain the same between burned and non-burned plots. This can be expected, since it is low growing and often remains unaffected by fire and grazing (Belsky 1992; Erichsen-Arychuk et al. 2002). Western wheatgrass (*Pascopyrum smithii* [Rydb.] A.Love) frequency increased or remained stable on burned and non-burned sites. Often, western wheatgrass decreases after fire (Antos et al. 1983; White and Currie 1983; Whisenant and Uresk 1989), but fire proceeded by wet springs can create western wheatgrass dominated sites (Vermeire et al. 2011). Sedge species (*Carex* spp.) decreased on shallow sandy native and sandy crested wheatgrass sites and maintained its frequency on sandy native sites. Sedge species in Montana also decreased (Vermeire et al. 2011), but sedge species responses are variable (White and Currie 1983; Whisenant and Uresk 1990). Green needlegrass (*Nasella viridula* (Trin.) Barkworth) trends increased on burned sites and were variable on non-burned sites. Needle-and-thread (*Heterostipa comata* (Trin. & Rupr.) Barkworth) was also variable, but it did decrease on non-burned shallow sandy native sites. Most work in the NGP has shown a decrease in these species (Whisenant and Uresk 1990; Vermeire et al. 2011).

Fire can decrease the cover and frequency of Kentucky bluegrass (Engle and Bultsma 1984; Gartner et al. 1986). Kentucky bluegrass frequency tended to decrease slightly on burned sites four MAF, but increased 16 MAF. Overall, all plots experienced an increased trend of Kentucky bluegrass frequency through time.

We saw variable responses from forbs to wildfire. Most of the forb species tended to remain stable and did not change in response to fire. Some forbs like prairie sagewort (*Artemisia frigida* Willd.) decreased four MAF, but this trend was not evident 16 MAF in 2014. Depending on species, forbs will decrease (Biondini et al. 1989; Shay et al. 2001), increase (Antos et al. 1983; Biondini et al. 1989), or remain neutral (Erichsen-Arychuk et al. 2002). Forb composition and density normally change due to fire timing and ecosystem responses (Biondini et al. 1989), so fire affects each species differently (Antos et al. 1983).

Ground Cover

Every study in the NGP has found decreased litter cover and increased bare ground following fire in any season (Engle and Bultsma 1984; Redmann et al. 1993; Shay et al. 2001; Vermeire et al. 2014). Loss of litter can be a large concern for areas experiencing drought stress, since water is one of the most limiting factors for plant growth and reproduction (Chapin et al. 1987; Reece et al. 2007). Normally, sufficient litter remains after a fire to perform ecological functions like reducing runoff and insulating soil temperatures and moisture (Dix 1960; Redmann et al. 1993). We found the same results four MAF. Bare ground, on average, was 24 percent higher on burned sites compared to non-burned sites. Live cover did not change, so this reduced the amount of litter basal cover. These same effects were not evident between burned and non-burned sites 16 MAF in 2014. Litter recovery after fire often takes between 2 - 5 years in the NGP (Dix 1960; Antos et al. 1983; Whisenant and Uresk 1989; Whisenant and Uresk 1990; Redmann et al. 1993; White and Loftin 2000; Ericksen-Arychuk et al. 2002; Grant et al. 2010; Gittins et al. 2011; Vermeire et al. 2011). In the case of our April wildfire, litter cover was re-established 16 MAF, or one growing season, when above average precipitation occurred. Litter stress may still exist in the future. Burned plots in Canada only experienced more bare ground when drought affected the sites two years after fire (Erichsen-Arychuk et al. 2002).

Phytomass Production

There were no changes in annual phytomass production for any ecological site or vegetation type due to the wildfire. Biomass is not expected to change when precipitation is above average (Vermeire et al. 2011). The same results would be expected if wet areas were burned (Shay et al. 2011). Production

responses are reliant more on precipitation compared to fire (White and Currie 1983; Gartner et al. 1986; Biondini et al. 1989; Whisenant and Uresk 1989; Biondini and Manske 1996; White and Loftin 2000; Vermeire and Rinella 2009; Vermeire et al. 2011; Vermeire et al. 2014). Precipitation is the most important factor, and fire will not help increase the amount of annual production on burned sites compared to non-burned sites (Engle and Bultsma 1984; Gartner et al. 1986). There was above average precipitation during the growing season in 2008 (+ 5.78 cm), 2013 (+ 10.32 cm), and 2014 (+ 8.20 cm) compared to the 30-year average, offsetting any negative or positive effects associated with fire. Herbivores will select recently burned areas compared to non-burned areas (Vermeire et al. 2004), so less production would be expected on burned sites. This change would be driven by herbivore consumption rather than fire effects. Cattle were visually observed heavily grazing burned areas compared to non-grazed areas (not quantified) in 2013 and 2014, but we still found no changes, positive or negative, in the amount of annual phytomass production. Some production responses may be more individualized, since threadleaf sedge production increased after fire in other areas of the NGP (White and Currie 1983). Precipitation events may dictate how plant species respond to disturbances like fire and subsequent grazing.

Soil Nutrients

Analyzed together, none of the soil nutrients showed any differences between defoliation treatments, fire treatments, date, or an interaction of these parameters. Independently, the different soil nutrients did not show a clear distinction between treatments. This was most likely due to the timing of samples. Most studies found increases or decreases in nutrients immediately following fire (Smith 1970; Higgins et al 1986; Hamilton and Frank 2001). We did not start soil sampling until five MAF. Grazing induces root mortality which promotes microbial activity and increases nutrient availability for plants (Hamilton and Frank 2001). Fire also increases nutrient availability by releasing nutrients bound in plant litter (Savadogo et al. 2007). Wildfires show higher levels of increased soil nutrients compared to prescribed fires, since fuels tend to build up more on wildfires (Wan et al. 2001).

Most nutrients were the highest five MAF, the closest sampling date to the fire event. Nutrients should be higher immediately after fire when they are released from litter and microbial activity increases

(Smith 1970; Higgins et al 1986; Hamilton and Frank 2001) if nutrients are not volatilized (Raison 1979; Boerner 1982). Most of the soil nutrients in our study were higher in the closest sampling date to fire (5 MAF). During this sampling period, plants were senescing, plots were accumulating litter, and the study area received significant precipitation. This could also account for the increases in soil nutrients (Facelli and Pickett 1991), with the exception of potassium and iron which were higher in later sampling periods.

Nitrogen generally stays the same following fire (Higgins et al. 1986; Wan et al. 2001), but it has been found to increase following fire (Seagle et al. 1992; Redmann et al. 1993; Johnson and Matchett 2001; Brockway et al. 2002). Nitrate made up most of the nitrogen allocation in our study, and this tended to increase after fire. In Kansas, nitrate only increased after grazing and was unaffected by fire or fire – grazing interactions (Veen et al. 2008). Ammonium trends increased on non-burned plots, but any changes in ammonium were offset by large amounts of nitrate for total nitrogen. Grazing and fire can increase nitrogen levels (Anderson et al. 2006), but heavy grazing can cause nitrogen levels to decrease on burned and non-burned sites (Savadogo et al. 2007). Heavy grazing and repeated fires reduces the root mass so much that microbes no longer have a food source to breakdown (Johnson and Matchett 2001). We did not consider our study to use heavy grazing. Plots were defoliated to a moderate grazing intensity, since all of the vegetation was removed evenly to 8.4 cm (Taylor and Lacey 1987).

We found higher trends of phosphorus on burned plots. Phosphorus increases after fire (Smith 1970; Brockway et al. 2002), but it can decrease outside the Great Plains (Snyman 2003). Cations normally increase following fire (Christensen 1976; Higgins et al. 1986; Brockway et al. 2002; Snyman 2003). Calcium was the only cation that showed increased trends on burned plots compared to non-burned plots. Magnesium tended to decrease on burned plots. Some cations will decrease after fire if they are heavily grazed (Savadogo et al. 2007) or leached out of the soil (Smith 1970).

Defoliation did not change soil nutrient responses to fire in the NGP (Christensen 1976). Some nutrients will decrease after fire and defoliation, with defoliation alone decreasing levels the most (Kitchen et al. 2009). Phosphorus, cadmium, and calcium were lowered by defoliation in our study. It may be difficult to transfer these results, since defoliation and grazing are separate techniques. Grazing is highly selective compared to our defoliation treatment, since herbivores will graze higher quality plants. Also, during grazing there is continual feedback to plant species and soil nutrients with added feces and urine

from herbivores (McNaughton et al. 1988; Augustine et al. 2003). There are discrete responses from each plant species when fire and grazing treatments are combined (Vermeire et al. 2014). This may also be true for soil nutrient responses, since none of the nutrients showed the same responses to date, fire, and defoliation.

Initially, there were concerns about forage availability and damage reducing basal live cover and phytomass production, since the fire occurred during a droughty period. There were reductions in basal litter, but production remained the same. Changes in phytomass production can often be offset by increases in crude protein following fire (Bloom-Cornelius 2008). A dormant season fire, with additional precipitation, is more likely to show fewer changes in plant species composition compared to growing seasonal fires (Wright 1974; Brockway et al. 2002). Defoliation after fire did not change soil nutrient properties. Grazing and fire have historically interacted with each other (Fuhlendorf et al. 2009), but few studies combine these disturbances early in the season (White and Currie 1983). Since there can be negative effects, it is important to have a management goal that positively offsets negative consequences (Snyman 2003). Inventory and proper management will increase the likelihood of fewer negative impacts after fire (Wright 1974; Bruce et al. 2007), but singular fires conducted during a drought are not expected to affect rangelands (Shay et al. 2001).

Conclusions

Managers can use research done on wildfires to help predict certain effects of prescribed fires on range and pasture lands when fire is an uncommonly used tool. Prescribed fire is a viable tool that is used for the management of herbaceous species and herbivores. There have been varying responses of specific plant species to fire in the NGP, but this is explained by climatic factors, especially precipitation. The evolutionary history of this area makes plants resistant to fire and defoliation. Soils also seem resistant to fire, since no clear patterns were seen between fire and defoliation treatments from the early April wildfire. Previous research shows that fire can increase nutrients, but grazing may reduce these soil nutrients. There may be distinct plant species and soil nutrient responses to fire and defoliation, and no one relationship explains every species or nutrient. We expected bare ground to increase while basal

litter decreased on burned sites. Since not all of the litter was consumed during the fire and precipitation was above average, these same differences were not seen one year after fire.

Overall, rangelands with or without crested wheatgrass in the NGP were unaffected by an April wildfire based on our study. Any negative effects were absent 16 MAF. Fire frequency is expected to increase in the Mountain West because of increased spring and summer temperatures. With earlier snowmelts, wildfires will increase in size, intensity, and frequency (Westerling et al. 2006). These same patterns could affect the NGP. Prescribed fire is one management strategy that could decrease the likelihood of wildfires or decrease their effected size (Fernandes and Botelho 2003). Management strategies will be determined with more work in the NGP, since fire management strategies used in other areas will not always be applicable in different moisture and temperature regimes (Lohmann et al. 2014). Fire is a historical part of the NGP. Phytomass production, wildlife habitat, heterogeneity, and biodiversity can all be improved by using evolutionary factors of fire and grazing that formed and shaped grasslands.

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