

IMPACTS OF KENTUCKY BLUEGRASS AND PATCH-BURN GRAZING MANAGEMENT
ON SOIL PROPERTIES IN THE NORTHERN GREAT PLAINS

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MASTER OF SCIENCE

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ABSTRACT

Kentucky bluegrass (*Poa pratensis* L.; hereafter referred to as bluegrass) has rapidly invaded the northern Great Plains over the past three decades, leading to declines in native plant diversity. A knowledge gap exists regarding the below-ground impacts that bluegrass has on soil properties. To address this knowledge gap, we measured soil physical, chemical, and biological conditions associated with bluegrass dominance resulting from idle management. We compared these results to patch-burn grazed areas with greater native plant species expression. Our results indicate that bluegrass influences soil microclimate, promoting cooler and wetter soil conditions. However, this shift does not appear to alter soil microbial abundance or carbon and nitrogen pools. Additionally, patch-burn grazing has limited impact on measured soil properties. We concluded that patch-burn grazing may be a practical land management technique for controlling bluegrass invasions without negative impacts on soil, and for promoting biological heterogeneity.

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DEDICATION

This work is dedicated to all natural lands, and the future generations who will cherish the
splendor and tranquility of these areas as much as I do.

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GENERAL INTRODUCTION

Plants are often the most prominent component of an ecosystem, and can impact the environment on a wide range of scales. Plants provide habitat for animals, produce food, allow for recreational opportunities, and play an important role in climate stabilization. Yet none of these services would exist without soil to support plant growth. Plants and soils are categorically dependent on one another; plants have the ability to alter soil conditions, which in turn affects subsequent plant growth (Kulmatiski et al. 2008). This reciprocal relationship is referred to as plant-soil feedback (PSF). A positive or negative PSF refers to whether the changes in soil properties caused by a given plant species results in an increase or decrease in the growth of that plant species, respectively. The role of PSFs is especially significant in grassland systems, as grasslands allocate most of their resources to root production, resulting in large root to shoot ratios (Blair et al. 2014). Additionally, grasslands experience some of the most significant non-native invasions of plant species, and research suggests this is due in large part to PSFs (Kulmatiski et al. 2008).

Kentucky bluegrass (*Poa pratensis* L.; hereafter referred to as bluegrass) is one of the most aggressive non-native species invading grasslands and encroaching on the range of endemic plant species. Bluegrass is a non-native perennial grass that has become naturalized throughout the entire United States. The species has expanded rapidly in grasslands over the last 30 years (DeKeyser et al. 2009) and is particularly prominent in the northern Great Plains (Toledo et al. 2014; USDA 2014). As bluegrass becomes established in a given ecosystem, a thickened thatch layer of living and dead bluegrass biomass develops between the soil surface and plant canopy (Toledo et al. 2014). In the absence of natural disturbances this litter layer accumulates, and has the potential to alter the physical, mechanical, and chemical conditions at the soil surface (Taylor

and Blake 1982; Wedin and Tilman 1990; Sanderson et al. 2017; Dornbusch et al. 2018a). These altered conditions can contribute to the suppression of native seedling emergence and shoot extension (Bosy and Reader 1995), ultimately leading to a decline in the production of native plant species.

There has been substantial speculation that bluegrass and the development of thatch may alter the temperature and hydrology at the soils surface (Toledo et al. 2014), transforming the soil microclimate by buffering the soil from environmental fluctuations. Additionally, bluegrass monocultures result in a lower soil C:N ratio than native C₄ grass species (Wedin and Pastor 1993). Altered nutrient pools paired with the potentially engineered microclimate may plausibly affect soil microbial communities. Based on the prevalence and spread of bluegrass in recent years, the relationship between bluegrass and soil likely represents a positive PSF, where changes in soil properties favor the growth of bluegrass over native plant species, further facilitating the prevalence of bluegrass, and the persistence of those soil conditions.

PSFs are one of the primary reasons that bluegrass dominates the grasslands of the northern Great Plains, but not the only reason. Phenological niche separation is another mechanism that contributes to the competitive advantage of bluegrass. As a cool season (C₃) grass, bluegrass emerges earlier than many of the native warm season (C₄) grasses in the prairie. While C₄ grasses make up only ~30% of the grass species in North Dakota (Teeri and Stowe 1976), bluegrass still gains a marginal competitive advantage by breaking dormancy earlier in the season, acquiring access to limited resources of light, water, and soil nutrients prior to the active growth period for native C₄ species. When tested in greenhouse experiments, phenological niche separation proved to substantially reduce the production of native grass species compared to bluegrass (Ulrich and Perkins 2014).

In recent decades, regions of the northern Great Plains have experienced above average rainfall, which has also facilitated the initial invasion of bluegrass (Sanderson et al. 2015). Once initially established in a site, research suggests that bluegrass is more drought resistant than previously believed, likely as a result of the species' tendency to go dormant above-ground during periods of drought, allocating a higher amount of biomass production to below-ground root structures than native grassland species (Dong et al. 2014). This pattern of invasion due to environmental conditions, and subsequent persistence is consistent with the 'back-seat driver' model described by Bauer (2012). In the back-seat driver model, an invasive species initially establishes in a given area due to favorable underlying environmental conditions. Following establishment, the invasive species thrives, and begins to drive change in native plant communities.

In evaluating the interactions of bluegrass and native plant communities during the invasion process, White et al. (2013) also found support for the 'driver model'. Through quantification of factors involved in bluegrass invasions, and structural equation modelling, White et al. (2013) identified bluegrass as the driving force responsible for reduced native plant cover measured in the grassland study. The apparent ability of bluegrass to succeed under a wide range of environmental conditions has led to the spread, dominance, and persistence of the species throughout the northern Great Plains.

Anthropogenic influence has also played a role in the recent expansion of bluegrass. Historically, plant species in the northern Great Plains region evolved under the natural influence of disturbances such as fire and grazing (Samson et al. 2004). These disturbances remove biomass, particularly standing and fallen grass litter, thus reducing competition for limited resources such as light, and allowing new species to germinate and thrive. Shifts away from

natural grazing patterns that have been associated with domestic livestock production, paired with the suppression of wildfire to protect property, further facilitate the invasion of bluegrass (DeKeyser et al. 2009). As conditions persist that favor bluegrass, the extent and prevalence of the species will become more pervasive. As a result, biodiversity will decline and grasslands will struggle to provide adequate ecosystem provisioning services like forage production (Waller and Schmidt 1983; Hockensmith et al. 1997). Adaptive land management techniques must be adopted to control bluegrass and promote native grassland plant species.

In the northern Great Plains patch-burn grazing is being explored as a land management technique to promote heterogeneity across the landscape, which may also control bluegrass. Intuitively, this disturbance-driven management model that applies fire and grazing to the landscape in natural patterns would favor native species that evolved under those conditions (DeKeyser et al. 2009). This idea is supported by an experiment conducted in the mixed-grass prairie of the Great Plains, showing that patch-burn grazing reduces the cover of bluegrass, as well as the influence of bluegrass on plant community composition over time compared to traditional management (Dornbusch et al. 2018b). One consideration when implementing this management model is that fire and grazing have varying and interactive impacts on soil properties, including physical, chemical, and biological conditions (Hobbs et al. 1991; Zhao et al. 2017; Alcañiz et al. 2018). In addition to the direct impact on soil, these disturbances may cause subsequent shifts in plant species assemblages, further affecting soil properties as a result of PSFs. As species assemblages shift away from bluegrass dominance towards those of native species that historically dominated the region, and natural disturbances are restored across the landscape, then it is reasonable to conclude that soils will also shift towards conditions prevalent before modern human settlement.

Considering the interplay of plant species assemblages, land management practices, and soil properties, we developed a research project exploring the reciprocal relationship between bluegrass dominance, pasture management with fire and grazing, and the physical, chemical and biological properties of soil. We hypothesized that the dominance of bluegrass and development of thatch may lead to a different soil microclimate, as well as organic-matter inputs that are different in quality and quantity compared to land managed with a combination of fire and grazing. The differences in microclimate and organic matter inputs may in turn affect soil nutrient and microbial dynamics. Additionally, fire and grazing may have direct impacts on these soil properties.

To address these hypotheses specifically, we designed two separate experiments. First, we compared soil hydrology, porosity, and temperature under the influence of bluegrass dominated vegetation to that of native vegetation, managed with fire and grazing. Next, we measured soil nutrient dynamics, microbial communities, and decomposition over a full season of active management with fire and grazing, again comparing that to soil under bluegrass dominated vegetation. Overall, soil in the study area proved to be resilient to variations in above-ground vegetation and management. Results indicated that soil moisture and temperature were directly impacted by bluegrass dominance, and that labile carbon and decomposition were affected by burning and grazing on a short-term scale. With those results aside, we found few significant differences between bluegrass dominated systems and patch-burn grazed pastures. The limited experimental design and high degree of spatial variability across the study area may have contributed to the lack of observed significant differences. In addition, intensive methodology and a broad research scope limited the opportunity for more extensive replication and spatial coverage in these experiments. A more robust experimental design may have revealed

additional subtle differences in soil conditions. With that said, the significant effects observed in this study are well founded, and the results have helped us to better understand consequences of bluegrass dominance and native grassland management approaches on fundamental soil processes.

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CHAPTER 1: QUANTIFYING THE EFFECT OF KENTUCKY BLUEGRASS THATCH ON PHYSICAL SOIL CONDITIONS

Abstract

Kentucky bluegrass (*Poa pratensis* L.; hereafter referred to as bluegrass) is a non-native, cool season (C₃), perennial grass that has become widespread across the northern Great Plains. This species produces a thick litter thatch layer above, and intermingled with, the soil surface, which can act as a buffer between above- and below-ground environmental conditions. A shift from native grass species to bluegrass has the potential to significantly influence ecosystem services by altering physical conditions in shallow soil, and at the soil surface. We hypothesized that the thatch layer and rooting structure of bluegrass would be associated with soil abiotic conditions (hydrology, porosity, and temperature) that differ from those managed for native plant community promotion using fire and grazing. To test this hypothesis, we conducted a series of in-field infiltration measurements and *in situ* soil water and temperature monitoring across study areas with different vegetative coverage characteristics and management regimes. Study areas included plots dominated by bluegrass with and without thatch, and plots with greater diversity of native plant species due to management with fire and grazing, with prescribed burns performed in the spring of 2017, and in the spring of 2018. *In situ* sensor records and infiltration testing indicate that bluegrass thatch had minimal impact on soil response to individual rainfall events. However, in bluegrass dominated plots, soil volumetric water content remained higher (by 31% and 28% at 5- and 15-cm, respectively) and soil temperature remained cooler (by 15% and 17% at 5- and 15-cm, respectively) throughout growing season months compared to burned and grazed plots with greater native plant diversity. Altering the soil abiotic environment in this

manner could contribute to a novel bluegrass ecosystem, potentially limiting the competitive ability of native plant species.

Introduction

For centuries, human development has shaped soil properties and functions on a global scale, often in degrading ways through activities such as poor agricultural practices, pollution, construction, and mining. One of the most significant ways that human activities have affected the soil is by altering plant communities. Plants have a strong influence on soil biological and physio-chemical properties (van der Putten et al. 2016), and when these properties are altered there can be adverse consequences for the health and productivity of the soil, and ultimately the ecosystem.

A significant anthropogenic influence on plant communities comes from the introduction of non-native, and often invasive, plant species to a given region. Approximately 42% of U.S. endangered and threatened species are currently in decline as a direct result of invasive species (Pimental et al. 2005). While invasive plant species pose an obvious threat above-ground by reducing biodiversity, they can also pose a threat below-ground. Invasive plants often have feedback interactions with soil (Klironomos 2002; Mangla et al. 2008; Lee et al. 2012), which help to promote their competitive advantage in a given ecosystem. These changes to plant community and soil may have detrimental effects on ecosystem provisioning services like forage production (Waller and Schmidt 1983; Hockensmith et al. 1997). For these reasons, it is crucial to conduct research exploring the relationship between invasive plant species and soil properties.

The grasslands of the northern Great Plains are a prominent example of an ecosystem significantly impacted by human disturbances and the expansion of invasive species. A publication from 2004 estimates that nearly 70% of the historical land-cover extent of native

Great Plains bioregion has been lost (Samson et al. 2004). *Poa pratensis* L. (Kentucky bluegrass; hereafter referred to as bluegrass) is one of the main invasive species encroaching on endemic prairie species in the northern Great Plains. Bluegrass is an effective competitor and has been rapidly expanding throughout the Great Plains over the last 30 years (DeKeyser et al. 2009) at the expense of declining native species and reduced biodiversity. In fact, bluegrass is now the most common plant species in the Prairie Pothole Region of the northern Great Plains (DeKeyser et al. 2015).

One characteristic of bluegrass that makes it such an effective competitor is the formation of a thatch layer between the soil surface and plant canopy. Thatch is made up of living and dead plant biomass atop a dense rhizomatous-root layer intermingled with the soil surface. This thatch layer acts to inhibit the germination and emergence of native species by altering the physical and chemical soil environment, as well as by acting as a physical barrier to shoot extension (Bosy and Reader 1995). This novel environment can lead to a synergistic positive feedback cycle perpetuating the proliferation of bluegrass.

Currently, very little published literature exists on the impact that bluegrass may have on soil conditions in grasslands. There has been much speculation that the presence of bluegrass thatch affects hydrological soil properties (Taylor and Blake 1982; Toledo et al. 2014). Bluegrass thatch may affect the way that water enters and is stored in the soil, as well as create a buffer between the soil and atmospheric temperature (Toledo et al. 2014). Additionally, bluegrass has a very dense and shallow rooting structure, which could affect the porosity and structure of soil aggregates (Toledo et al. 2014). If bluegrass is indeed altering soil moisture, porosity, and temperature dynamics, conditions may shift in a way that favors the species even further, reducing the likelihood of native species regeneration in the future. Research is necessary to

understand exactly how bluegrass may be impacting the physical properties of the soil in the northern Great Plains region where this species has become so prominent.

Rainfall and soil moisture are often the most limiting factors for vegetation growth in the semi-arid grassland ecosystems of the northern Great Plains (USDA NRCS ESD 2017), so it is vital to understand the relationship between the presence of bluegrass thatch and soil moisture dynamics. Currently, there are conflicting ideas as to how thatch may impact soil hydrological properties. Some studies indicate that thatch may interfere with and delay water entry into the soil profile (Pierson et al. 2002); however, Hurto et al. (1980) found that the pores in bluegrass thatch are larger than those in the soil, so a delay in water reaching the soil profile cannot be attributed to the pore size of bluegrass thatch. Other study results suggest that bluegrass thatch has little effect on water infiltration (Taylor and Blake 1982). With no clear consensus on how, or even if, bluegrass thatch is affecting hydrological properties, further research is necessary.

The dense and shallow rooting structure typical of bluegrass may also play a role in altering the hydrology of the shallow soil. Ledebor and Skogley (1967) found that in bluegrass turf systems fine roots were prevalent in the thatch layer, while very few roots extended into and throughout the soil below. Increased root density in the shallow depths of soil, with limited penetration to deeper depths, may affect the distribution and connectivity of soil pores, which in turn may alter the way that water moves through the soil profile, as well as how it is stored and utilized. Soil porosity is important to understand not only in relation to water movement, but also because it is essential to plant health and growth. At this time, there is no published research on how the phenotypic rooting characteristics of bluegrass may impact soil porosity in natural grassland systems.

Soil temperature is another property that is potentially impacted by the presence of bluegrass dominance and the development of a thatch layer, as the accumulation of biomass can act as a buffer to moderate temperature extremes (Beard 1973). Boserup and Reader (1995) found that the presence of bluegrass litter ($715 \pm 104 \text{ g m}^{-2}$) statistically significantly reduced mean soil maximum daily temperature, and solar irradiance on the soil surface. Ultimately, the presence of thatch significantly reduced the germination of seeds from the majority of species tested in the experiment (Boserup and Reader 1995). These results suggest that the changes in environmental conditions at the soil surface may be a key mechanism in altering environmental cues, such as temperature, for germination of non-bluegrass seeds. Additional researchers agree that bluegrass litter can affect soil temperature, and ultimately even shift the competitive advantage towards cool-season growing species (i.e. bluegrass) that are adapted to lower light and temperature soil conditions (Printz and Hendrickson 2015). By limiting the germination of native seeds, and creating conditions favorable for bluegrass growth, the positive feedback cycle resulting in bluegrass dominance is even further enhanced.

In addition to the primary competitive advantage that bluegrass presumably has over native prairie species resulting from the development of a thatch layer, the species' rapid dominance can also be attributed to historically inefficient management practices (DeKeyser et al. 2009). The plant communities of the northern Great Plains evolved with a combination of periodic fire and ungulate grazing (Fuhlendorf and Engle 2001). However, conventional management practices in recent history are in stark contrast to these historical conditions. Conventional management practices such as range readiness (delaying grazing until June 1st), deferment (non-grazing periods greater than one year), and spot grazing (over and underutilization of resources in certain areas) – all practices that are often paired with fire

suppression – have contributed to shifting the competitive advantage further towards bluegrass (Printz and Hendrickson 2015).

The rapid expansion of bluegrass across the northern Great Plains, resulting from a combination of conditions encouraging its growth, has largely gone unnoticed and unchecked by most land owners (Printz and Hendrickson 2015). One reason for this may be that bluegrass can serve as a nutritious forage during certain periods of the growing season (Toledo et al. 2014). However, during the warm dry summer months of the growing season, bluegrass produces lower forage quality (Hockensmith et al. 1997). Additionally, Fu et al. (2004) found that bluegrass required soil moisture at nearly 100% of field capacity to meet evapotranspiration needs for optimal growth, more than any of the other species measured in the study. As increased temperature and variable precipitation become more common with global climate change (USGCRP 2018), range systems dominated by bluegrass may become more susceptible to drought, and unable to provide adequate provisioning ecosystem services compared to those with a diverse native composition of plant species.

As bluegrass prevalence continues to increase, the need for land management techniques that both control bluegrass and promote native plant species also increases. Reintroducing the coupled interaction of fire and natural grazing patterns has been proposed as a practical option for restoring biodiversity to prairie ecosystems (Fuhlendorf and Engle 2001) and preventing thatch accumulation and bluegrass perpetuation. In the northern Great Plains, patch-burn grazing is being explored as a land management technique that more closely mimics historical disturbance patterns which favor native species. Patch-burn grazing consists of applying spatially-discrete fires to the landscape each year, and allowing grazing animals access to the entire pasture containing a mosaic of patches varying in time since disturbance (Fuhlendorf and

Engle 2004). Unlike conventional range management practices, this disturbance-driven management model helps to promote heterogeneity and diversity across the landscape (Fuhlendorf and Engle 2001). Additionally, grazing and fire help to remove the thatch layer, which reduces the competitive advantage of bluegrass (Naeth et al. 1991; Biondini et al. 1998; Kral et al. 2018).

To better understand the relationship between bluegrass and the physical properties of soil, we developed a study measuring soil hydrological, structural, and thermal characteristics in soils affected by bluegrass. Specifically, we measured soil infiltration, runoff, moisture, temperature, and porosity (assessed qualitatively) across bluegrass-dominated exclosures and patch-burn grazed systems favoring native grass and forb species. We selected managed native systems to compare to bluegrass-dominated sites because native remnant patches are increasingly rare in the mixed-grass prairie of North Dakota where this study was conducted. By understanding how bluegrass may alter the physical properties of soil, we will gain a better understanding of whether this species poses a threat below-ground to the function of soil, in the same way that it does above-ground to species biodiversity.

Methods

Study Area

We conducted this research in the mixed-grass prairie at North Dakota State University's Central Grasslands Research Extension Center (CGREC) in south-central North Dakota. The research center is located near Streeter, North Dakota, USA, which lies within the Missouri Coteau region. The Missouri Coteau region is of glacial origin, and is typically characterized by irregular, rolling, rocky plains (Rogers et al. 2005). Prairies in this region are characterized by the dominance of cool season, perennial grasses (Toledo et al. 2014). We conducted our study in

thin loamy ecological sites, which have a reference plant community that includes little bluestem (*Schizachyrium scoparium*), porcupine grass (*Hesperostipa spartea*), sideoats grama (*Bouteloua curtipendula*), green needlegrass (*Nassella viridula*), plains muhly (*Muhlenbergia cuspidata*), blue grama (*Bouteloua gracilis*), sedges (*Carex spp.*), purple coneflower (*Echinacea angustifolia*), American vetch (*Vicia americana*), leadplant (*Amorpha canescens*) and fringed sagewort (*Artemisia frigida*) (Sedivec and Printz 2012). However, in the absence of fire and grazing, this plant community shifts towards one dominated by bluegrass, smooth brome (*Bromus inermis*), scurfpea (*Pediomelum argophyllum*), western yarrow (*Achillea millefolium*), western snowberry (*Symphoricarpos occidentalis*), and rose (*Rosa arkansana*) (Sedivec and Printz 2012).

Extreme temperatures and light rainfall are characteristic in this region. Rainfall is typically the limiting factor for vegetation, with annual precipitation ranging from approximately 38-51-cm per year (USDA NRCS ESD 2017), the majority of which falls during the growing season months. Soils in the study area are typically well drained and have low runoff depending on the slope and the vegetative cover (USDA NRCS ESD 2017). Experimental sites were selected from map units identified as the Zahl-Williams-Zahill complex by the USDA NRCS online Web Soil Survey (2017). Typical soil profiles for these units include a thin layer of loam (0-15-cm), on top of clay loam extending through the substrata (USDA NRCS WSS 2017).

Experimental Design

In 2017, we established eight original sampling plots for this study distributed across the research center. The plots were selected based on the historical management of the land, and a visual assessment of expressed plant community, which we validated through plant community

surveys. Half of the plots were composed of predominantly native vegetation, representing the reference plant community, and the remaining plots were dominated by bluegrass.

The plots established in 2017 with predominantly native plant species were managed with fire and grazing, in the context of a larger patch-burn grazing management system (hereafter referred to as '17 burn, $n=4$). We burned these plots in the spring of 2017, in conjunction with season-long, moderate grazing between mid-May and October. The pastures are stocked at a rate so that approximately 50% of the vegetation produced during the growing season remains after grazing. In contrast, the bluegrass dominated plots (hereafter referred to as KBG, $n=4$) were excluded from grazing, and have not been burned in recent decades. These conditions have resulted in bluegrass dominance and the accumulation of a thick thatch layer. Additional burned and grazed plots were added in 2018 (hereafter referred to as '18 burn, $n=4$) following the spring burns in order to compare to those burned in 2017 which had a year to recover post-disturbance when measured in 2018. Figure 1.1 outlines essential components of the experiment, including: a site map, treatment descriptions, and the timeline of experimental testing. Details of specific experimental tests are included in the *Data Collection* section of this paper.

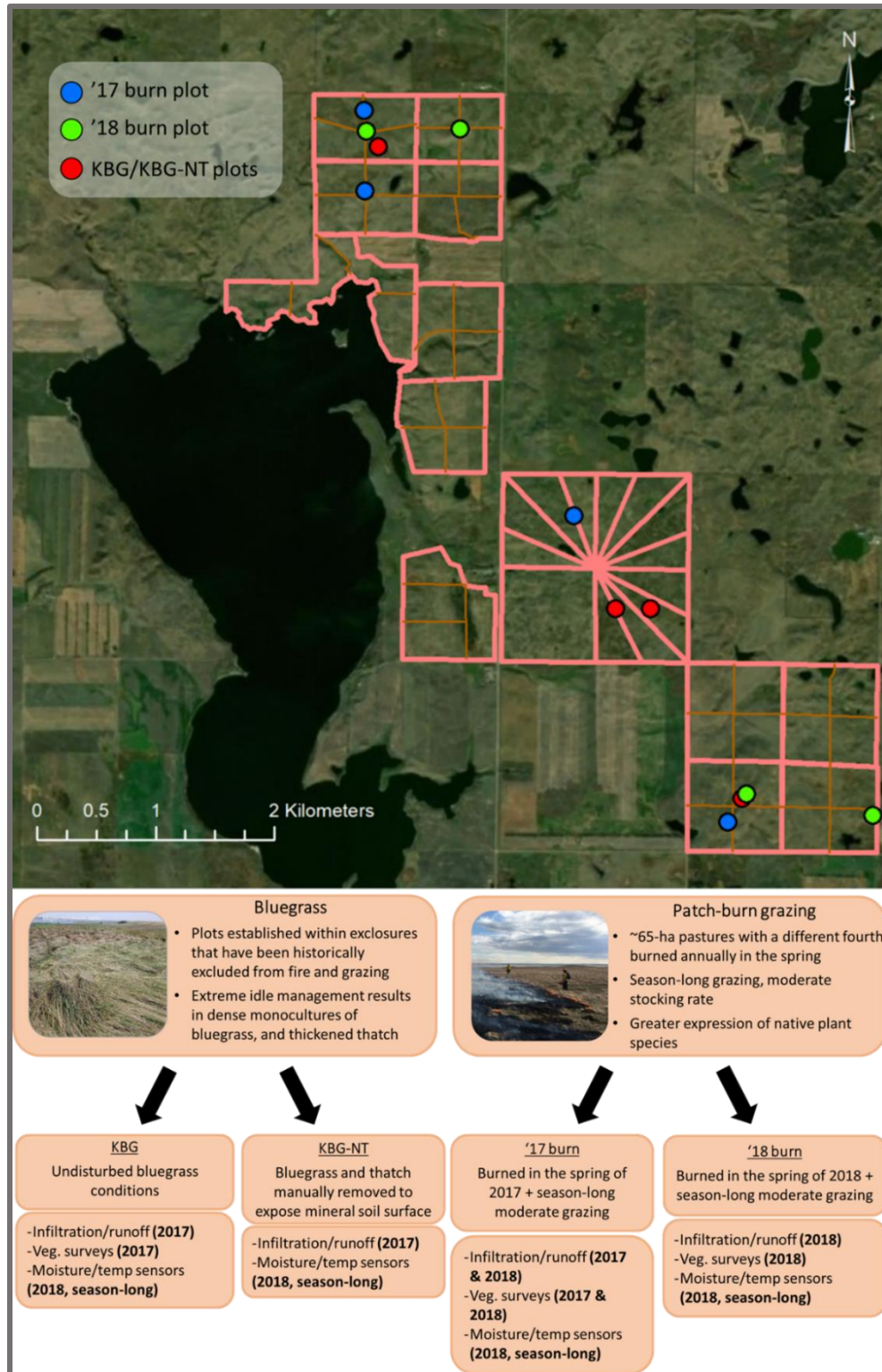


Figure 1.1. Site map and experimental design. Summary of experimental conditions, from top to bottom: Site map with the spatial distribution of experimental plots, ecological conditions, specific experimental treatments, and the timing of experimental testing.

Data Collection

Vegetation

Vegetation data were intended to be mainly descriptive rather than highly investigative, as the focus of this study was on below-ground properties rather than above-ground. Following establishments of plots, we conducted plant surveys in early August of each growing season. We recorded surface characteristics (e.g. standing dead litter, basal cover, rock cover, etc.) and canopy cover of each plant species within regularly spaced 0.5 x 0.5-m quadrats anchored around a center point. Cover classes were established at: 1=0-1%, 2=1-2%, 3=2-5%, 4=5-15%, 5=15-25%, 6=25-35%, 7=35-45%, 8=45-55%, 9=55-65%, 10=65-75%, 11=75-85%, 12=85-95%, 13=95-98%, 14=98-99%, 15=99-100%. We then calculated midpoint values of each class and aggregated species across functional groups (forb, shrub, graminoid, or sedge; further stratified as native or exotic to North Dakota) for further statistical analysis. For each treatment we determined mean species richness, and percent cover (for both native and non-native species), and calculated inverse Simpson's diversity:

$$invSimp = 1 / \sum p_i^2$$

where p_i is the proportional abundance of species i (Simpson 1949; Lande 1996).

Rainfall infiltration/runoff

We used a Cornell sprinkler infiltrometer to provide infiltration rate and runoff data at each plot. The Cornell sprinkler infiltrometer measures infiltration through ponded ring infiltration paired with simulated rainfall (Ogden et al. 1997). This method most effectively mimics real rainfall conditions while capturing runoff rates. The difference between the simulated rainfall rate (r) and runoff rates (ro_i) was used to calculate the infiltration rate (i_i).

We measured infiltration and runoff rates at each plot late in the growing season of 2017, and again at each burned and grazed plot in 2018. In the KBG treatment, infiltration tests were conducted in 2017 on two separate conditions in close proximity within each experimental plot: 1) undisturbed bluegrass and thatch conditions (KBG), and 2) manual removal of above-ground bluegrass and thatch to the mineral soil surface, clipped immediately prior to testing (KBG-NT). The difference between these two conditions helps to interpret the direct impact of thatch on infiltration rate.

We then calculated the percent of total simulated rainfall infiltration relative to total rainfall runoff based on cumulative data from these tests. Additionally, soil samples were collected after each test at depths of 0-5-cm and 5-15-cm, and analyzed for gravimetric water content following the simulated rainfall event. We calculated the ratio of water content at 0-5-cm (surface) relative to 5-15-cm (shallow) in order to quantify the evenness of soil water storage throughout the upper soil profile. This ratio also serves as a relative quantitative indicator of soil pore distribution.

Using the infiltration data, parameters were inversely fitted to a model and then used for comparison to better understand soil response to rainfall across treatments. Parameters include sorptivity (S), which is a hydraulic property describing early infiltration independent of rainfall rate:

$$S = (2T_{ro})^{0.5} * r$$

where T_{ro} refers to time-to-runoff, and r is the simulated rainfall rate (Kutilek 1980). We also calculated field-saturated infiltrability (i_{fs}), a parameter estimating the steady-state infiltration capacity of the soil after an initial wetting period:

$$i_{fs} = i_t * 0.80$$

where i_t is the steady-state infiltration rate, and 0.8 is the conversion factor accounting for soil type and ring-insertion depth (Ogden et al. 1997). The adjustment factor used in the i_{fs} calculation is based on modeling from Reynolds and Elrick (1990). The i_{fs} parameter is not independent from rainfall rate, so relative values between treatments are variable.

Additionally, we measured soils at each plot for water repellency, as fire may influence repellency, and repellency may play a significant role in infiltration and runoff rates. Measurements were taken at all plots early in the 2018 growing season, and again for the '18 burn treatment immediately after spring burning to determine if the fire event may trigger soil water repellency. We used the water drop penetration time (WDPT) method, placing droplets of distilled water on soil cores and timing how long the droplets persisted before complete infiltration (Dekker et al. 2009). Soil cores were collected to a depth of 15-cm, and water droplets were spaced 1-cm apart along the length of the core. This particular method was used because it is an efficient field test compared to other methods that involve laboratory manipulation, which may not be as relevant to the field conditions that we were interested in for this study.

Pore structure

To understand the effect of bluegrass rooting structure on soil and water flow dynamics we measured soil root density and bulk density, and assessed porosity of intact soil cores. We collected soil core samples of known volume (2.5-cm radius x 15-cm height cylinder, $n=3$) from each plot in 2017 and measured root density. Separating live roots from organic matter and dead roots is recommended (Bledsoe et al. 1999), however due to the dense and tangled rooting nature of bluegrass, identifying and then separating living from dead roots was not possible. Once separated from the soil, roots were oven dried at 150 °C for at least 48-hours and then weighed.

We determined root density by dividing the dry weight of roots (g) by the volume of the soil core (cm^3). Additionally, we collected soil cores of known volume (2.5-cm radius x 15-cm height cylinder, $n=3$) from each plot to measure the bulk density of the soil (Blake and Hartge, 1986).

Lastly, we collected a single intact soil core (3.8-cm radius x 15-cm height cylinder) from each plot in 2018, which was kept intact in an acetate sleeve. For each core, we depicted internal soil pore and rooting characteristics through three-dimensional computed tomography (CT) imagery scanned at the Electron Microscopy Center at North Dakota State University.

Soil moisture and temperature

We measured the influence of bluegrass thatch as a buffer between environmental conditions and the soil surface by installing GS3 sensors (METER, Inc., Pullman, WA), which measure the water content and temperature of the soil, as well as EM-50 data loggers (METER, Inc., Pullman, WA) to record sensor readings. The sensors were installed in the soil at depths of 5- and 15-cm, logging data every two hours throughout the 2018 growing season. Absolute volumetric water content readings from GS3 sensors were adjusted based on a single point bias correction. Gravimetric soil water content was obtained from field soils in the spring of 2019 and converted to volumetric water content using bulk density (assuming a particle density of 2.65 g/cm^3). At the time of field sampling, we downloaded sensor readings, and then adjusted the sensor record according to the difference between the sensor volumetric water content and the field volumetric water content.

Sensor readings span from May 1, 2018 – September 30, 2018, although there are periods of missing data, due to sensor or data logger malfunction. From this period, mean values of the absolute calibrated sensor data were calculated and analyzed on a monthly basis. For descriptive purposes, the sensor readings were aggregated as follows: daily average and coefficient of

variation (CV; $CV = (\text{standard deviation} / \text{mean}) * 100$) by plot, bihourly average and CV by treatment, and daily average and CV by treatment. To explore the response to rainfall events, we developed an automated algorithm in R (R Core Team 2018) to identify peaks in the sensor record that adhere to certain parameters (including that the peak is preceded by rainfall > 0 -cm in the previous 48 hours, and that the peak is greater than $0.01\text{-m}^3/\text{m}^3$ in magnitude from pre-peak records). From these peaks, we extracted and calculated the following parameters: 1) peak magnitude (the difference between max volumetric water content (VWC) following a rainfall event and the VWC prior to that rainfall event; calculated in absolute units, m^3/m^3 , as well as in terms of change in VWC per cm of rainfall, $\text{m}^3/\text{m}^3/\text{cm}$) and 2) drying rate ($\text{m}^3/\text{m}^3/\text{minute}$) over 2, 6, 12, and 24 hours after the peak. We analyzed drying rate over a 24 hour period to help us to isolate the effect of a singular rainfall event by minimizing the probability of subsequent rainfall events occurring in such a short time window.

Data Analysis

All sensor processing was completed in R (R Core Team 2018), with assistance from ‘dplyr’ (Wickham et al. 2019), ‘fields’ (Nychka, et al. 2015), ‘goeveg’ (Friedemann and Schellenberg 2018), ‘lubridate’ (Spinu et al. 2018), ‘plyr’ (Wickham 2011), and ‘scales’ (Wickham 2018) packages for data processing, analysis, and visualization.

Statistical differences not assessed in R were analyzed using a one-way analysis of variance (ANOVA) test with Tukey’s honest significance test used for mean comparisons. For experimental tests that were repeated on the same treatments in 2017 and 2018, we performed a paired *t*-test to evaluate if measured properties changed over a one year period. These analyses were performed in JMP[®], Version 13. SAS Institute Inc., Cary, NC, 1989-2019, and significant differences were determined at a $p \leq 0.05$.

One-way ANOVAs assume that a population is normally distributed, has uniform variances, and samples are independent. We have no reason to believe that that populations in this study are not normally distributed, or that the variances are not equal, and we are limited in our ability to assess these characteristics across $n = 4$ samples. While the unique pasture histories across experimental plots in this study could possibly affect the independence of samples, many studies show that soil properties become independent within 5-10 meters – across many different ecosystems, including grasslands (summarized by Ettema and Wardle 2002; Ritz et al. 2004). Additional differences in topography, land use history, and soils all increased the spatial independence of experimental units. Statistical analyses prior to publication efforts may re-evaluate data using non-parametric, linear, and/or generalized mixed-effect models.

Results

Results from vegetation analysis are summarized in Table 1.1. We found native plant species richness and inverse Simpson's diversity to be greatest in recently burned and grazed plots, and lowest in bluegrass monocultures with no management. Mean values for plant species percent cover seemed to vary considerably across study years, suggesting that observed changes were more likely a result of fluctuating environmental conditions than treatment differences. When measured in the same year, the '17 burn treatment had nearly twice the native plant cover, and half the non-native plant cover, compared to the KBG treatment.

Table 1.1. Site vegetation characteristics. Average values for inverse Simpson’s diversity, richness, and percent cover of native and non-native plant species, from plots dominated by Kentucky bluegrass (KBG, $n=4$), and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 (’17 burn, $n=4$), and in the spring of 2018 (’18 burn, $n=4$). Vegetation measurements were collected from the North Dakota State University Central Grasslands Research Extension Center in the growing seasons of 2017 and 2018.

	2017					
	Inverse Simpson	Native Richness	Non-Native Richness	Native Cover (%)	Non-Native Cover (%)	Bluegrass Cover (%)
KBG	2.56	8	3	21	44	39
’17 burn	7.24	23	7	41	25	19
	2018					
	Inverse Simpson	Native Richness	Non-Native Richness	Native Cover (%)	Non-Native Cover (%)	Bluegrass Cover (%)
’17 burn	6.75	21	6	24	18	13
’18 burn	5.44	15	5	17	23	12

Infiltration and runoff results proved to be extremely variable, and we found no overall significant differences across treatments, although we did observe changes within treatment over repeated measures (Fig. 1.2). Tests conducted in 2017 showed highest infiltration and lowest runoff in KBG-NT, KBG, and ’17 burn, yr. 1, respectively. When testing was repeated in 2018, we found that the infiltration rate in the ’17 burn treatment increased significantly and runoff rate decreased significantly compared to the previous year, following 24 minutes of steady simulated rainfall. The percent of total simulated rainfall infiltration relative to total runoff mirrored these results, with infiltration increasing significantly from $27.3 \pm 13.2\%$ to $56.1 \pm 14.2\%$, and runoff decreasing significantly from $72.7 \pm 13.2\%$ to $43.9 \pm 14.2\%$ between testing in 2017 and 2018 ($p \leq 0.05$). Due to study limitations, this was the only treatment that was tested in both years. Results from the ’18 burn, yr. 1 (also measured in 2018) showed relatively high levels of infiltration compared to ’17 burn, yr. 1, despite being experimentally equivalent treatments.

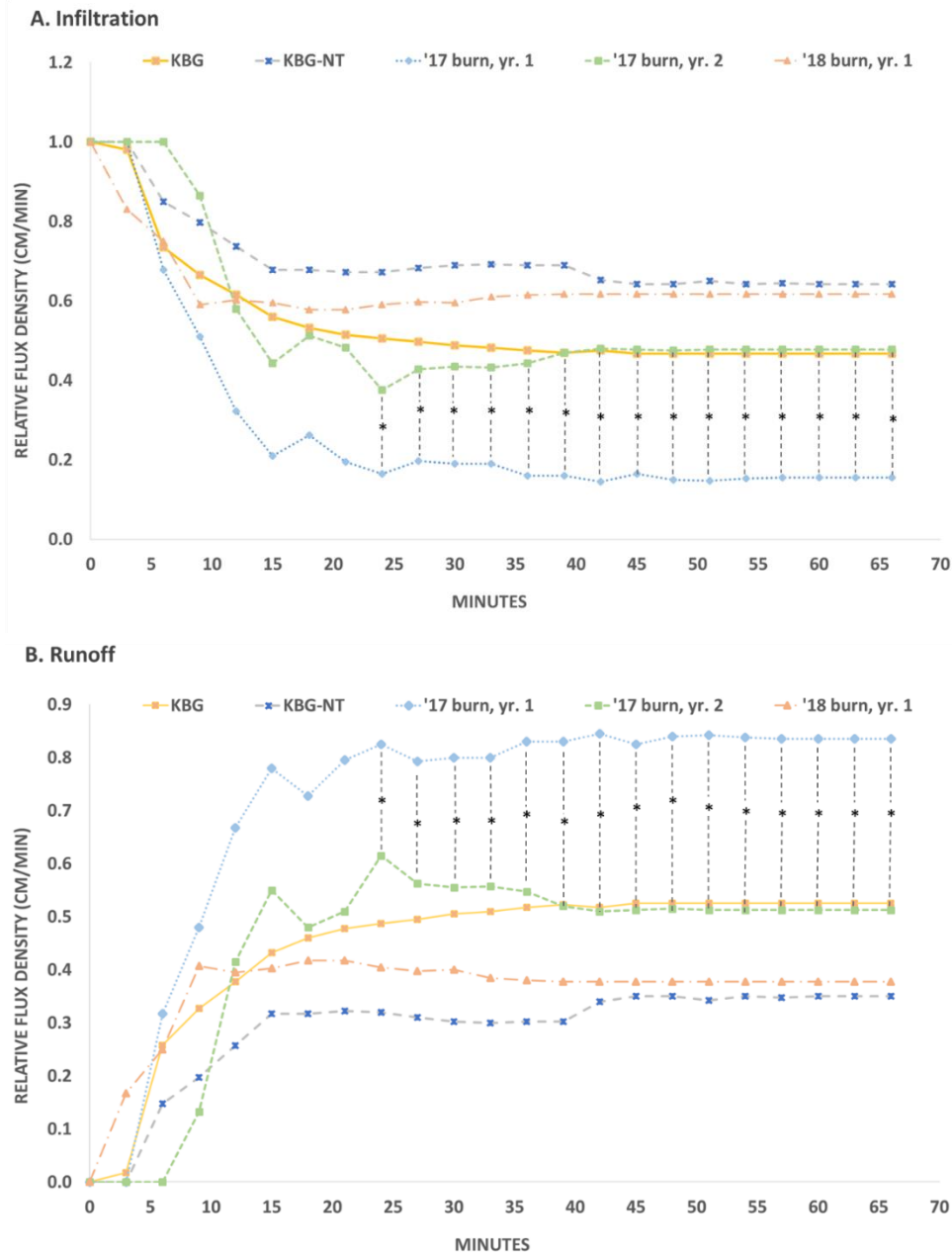


Figure 1.2. Average infiltration and runoff rates across treatments and study year. Infiltration (A) and runoff (B) rates, displayed in terms of relative flux density of water (cm/min), from simulated rainfall testing conducted in 2017 at plots dominated by Kentucky bluegrass with thatch (KBG, $n=4$) and with thatch manually removed (KBG-NT, $n=4$), and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, yr.1, $n=4$; '17 burn, yr. 2 when repeated in 2018, $n=4$). Also displayed, results from simulated rainfall testing conducted in 2018 at plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2018 ('18 burn, yr. 1, $n=4$). Asterisks connected with dashed lines indicate statistical significance within treatment, determined by t -test comparisons. Statistically significant differences were determined at ($p \leq 0.05$).

Parameters extracted from the simulated rainfall data are displayed in Table 1.2. Analysis revealed no statistically significant differences across treatments, or within the '17 burn treatment between the measurements in 2017 and 2018. Overall, steady state infiltration (field-saturated infiltrability) was greatest in KBG and KBG-NT treatments. There were no consistent trends for early infiltration (sorptivity), or the ratio of water distribution in the soil profile following a rainfall event (surface:shallow soil H₂O). The results of water repellency testing following prescribed burns in 2018 revealed three of four study plots to have repellent tendencies on the soil surface immediately following fire. Yet when the same plots were tested again a week following the fire, we found no repellency.

Table 1.2. Rainfall response parameters. Hydraulic properties describing early infiltration (sorptivity), steady-state infiltration (field-saturated infiltrability), and the ratio of water distribution in the soil profile at 0-5-cm (surface) compared to 5-15-cm (shallow), following a rainfall event. Parameters calculated from simulated rainfall testing conducted in 2017 at plots dominated by Kentucky bluegrass with thatch (KBG, $n=4$) and with thatch manually removed (KBG-NT, $n=4$), and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, yr. 1, $n=4$; '17 burn, yr. 2 when repeated in 2018, $n=4$). Also displayed, results from simulated rainfall testing conducted in 2018 at plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2018 ('18 burn, yr. 1, $n=4$).

Treatment	Sorptivity ($\text{cm} \cdot \text{min}^{-1/2}$)	Field-Sat Infiltrability (cm/min)	Surface:Shallow Soil H ₂ O
'17 burn, yr. 1	1.40	0.10	2.5:1
'17 burn, yr. 2	1.58	0.12	1:1
'18 burn, yr. 1	0.70	0.14	1.8:1
KBG	1.10	0.24	2:1
KBG-NT	2.82	0.36	2:1

Visual representations of undisturbed soil conditions obtained from X-ray computed tomography (CT), depicted in Figure 1.3, provided additional insight into internal soil structure

and porosity, and have allowed us to speculate on how these soil properties may play a role in soil hydrology. Although there was variation between study plots within treatment, visual assessment of the scans showed that the '18 burn and KBG treatments have higher visible porosity and larger root channels deeper into the soil profile. Greater porosity and connectivity of root channels allows for greater rates of water infiltration into the soil profile. This assessment parallels in-field observations of soil structure, which found greater soil aggregation in these treatments than in the '17 burn. These results also correspond with the results from simulated rainfall testing, which showed greatest rates of infiltration and lowest runoff in KBG, KBG-NT, and '18 burn treatments. Perhaps as a consequence of the inherent soil variability in study areas, analysis of root and soil density showed no statistically significant differences across treatments (Table 1.3).

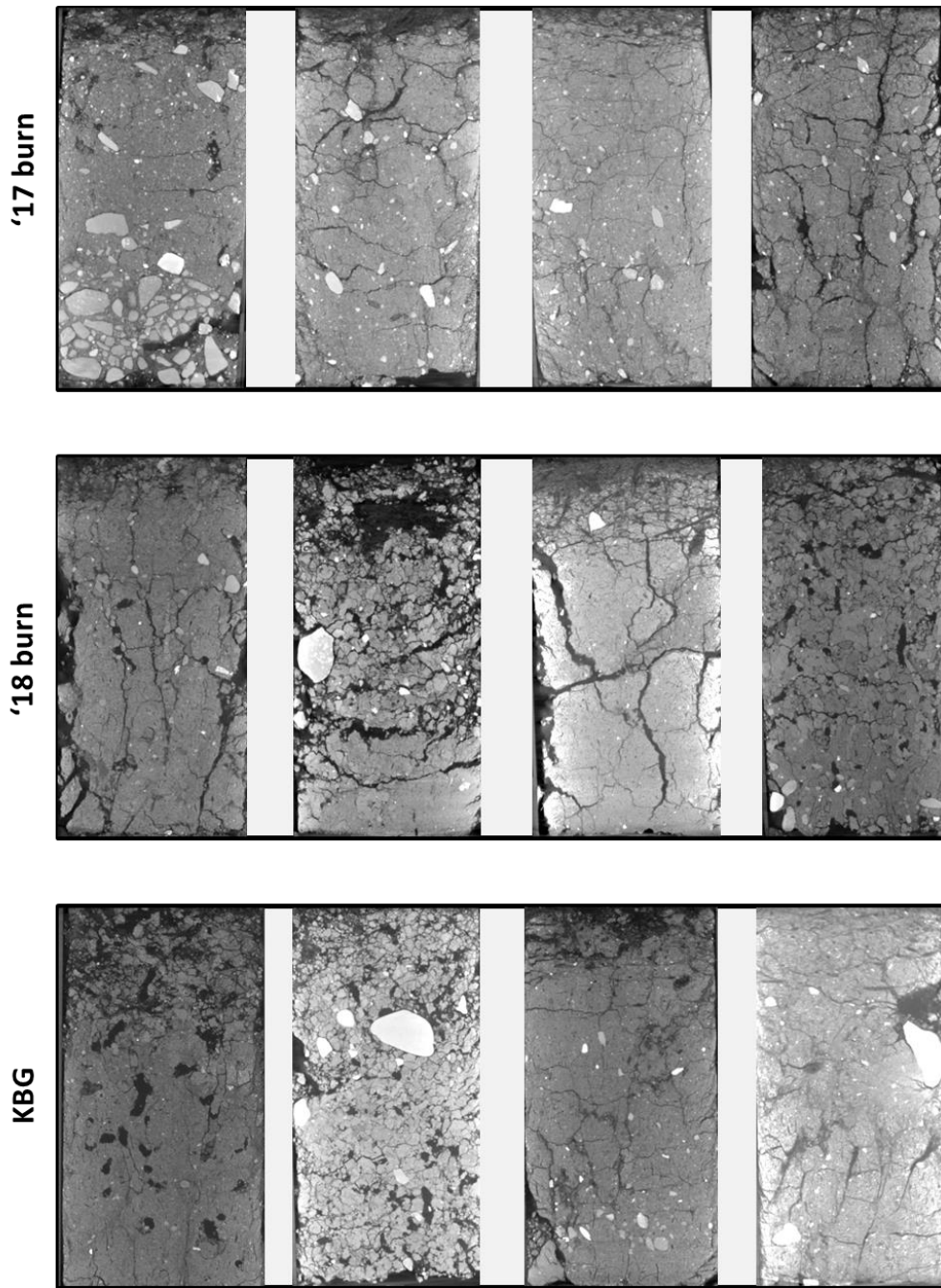


Figure 1.3. Orthogonal cross-section view of three-dimensional computed tomography (CT) soil core scans. Images represent internal structure of intact soil cores (dimensions 3.8-cm radius x 15-cm height) collected from plots dominated by Kentucky bluegrass (KBG, $n=4$) and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Cores were collected from the North Dakota State University Central Grasslands Research Extension Center in the summer of 2018. Lighter colors represent higher density materials, and darker colors represent lower density materials (small white spots are rock masses, shades of gray are the soil matrix, and black areas are empty pores or root channels). Gray-scale is variable across scans due to beam-hardening.

Table 1.3. Soil root and bulk density. Average density calculated from soil core samples of known volume (2.5-cm radius x 15-cm height cylinder, $n=3$ from each plot) collected from plots dominated by Kentucky bluegrass (KBG, $n=4$) and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Statistical significance was assessed at $p \leq 0.05$.

Treatment	Root density mg/cm ³	Bulk density g/cm ³
'17 burn	3.97	0.91
KBG	5.86	0.90
'18 burn	10.3	0.84

Average values of soil volumetric water content (VWC) and temperature (°C) are detailed in Table 1.4. VWC was greatest in the KBG treatment at both 5- and 15-cm for every month measured during the 2018 growing season. Statistically significant differences in VWC were observed at 5-cm in May, July, and September. Results also showed that the KBG treatment maintained consistently cooler average temperatures at both 5- and 15-cm for every month of the 2018 growing season. Statistically significant differences in temperature were observed at both 5- and 15-cm in every month except September. There were no statistically significant differences between the '17 burn and '18 burn treatments, but trends showed the '17 burn treatment (which had a year to recover post-fire when measured in 2018) tended to have greater VWC at 5-cm, and lower VWC at 15-cm.

Table 1.4. Average monthly volumetric water content and temperature characteristics for the 2018 growing season. *In situ* sensor data records of average (A) volumetric water content (VWC), and (B) temperature (°C) at 5- and 15-cm from plots dominated by Kentucky bluegrass (KBG, $n=4$) and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Letters (a, b) represent statistically significant differences across treatments, at an individual depth ($p \leq 0.05$).

A. 2018 VWC (m³/m³)					
5-cm					
	May	June	July	Aug	Sept
KBG	0.35 (0.06) ^a	0.33 (0.05) ^a	0.34 (0.05) ^a	0.24 (0.03) ^a	0.25 (0.03) ^a
'17 burn	0.27 (0.01) ^b	0.26 (0.02) ^a	0.26 (0.03) ^{ab}	0.17 (0.05) ^a	0.19 (0.02) ^{ab}
'18 burn	-	0.25 (0.04) ^a	0.22 (0.05) ^b	0.13 (0.08) ^a	0.12 (0.07) ^b
15-cm					
	May	June	July	Aug	Sept
KBG	0.29 (0.07) ^a	0.27 (0.08) ^a	0.27 (0.07) ^a	0.20 (0.08) ^a	0.18 (0.04) ^a
'17 burn	0.21 (0.04) ^a	0.20 (0.01) ^a	0.19 (0.03) ^a	0.11 (0.03) ^a	0.12 (0.02) ^a
'18 burn	-	0.25 (0.06) ^a	0.26 (0.07) ^a	0.13 (0.04) ^a	0.11 (0.02) ^a
B. 2018 Temp (°C)					
5-cm					
	May	June	July	Aug	Sept
KBG	11.4 (1.7) ^a	17.7 (1.3) ^a	19.3 (1.3) ^a	18.6 (0.9) ^a	15.4 (0.3) ^a
'17 burn	15.2 (0.7) ^b	20.4 (0.2) ^b	21.8 (0.3) ^b	20.2 (0.4) ^b	16.4 (0.7) ^a
'18 burn	-	19.9 (0.6) ^b	21.1 (0.2) ^b	19.3 (0.2) ^{ab}	16.2 (1.2) ^a
15-cm					
	May	June	July	Aug	Sept
KBG	9.7 (1.6) ^a	16.6 (1.3) ^a	18.8 (1.2) ^a	17.8 (0.8) ^a	15.3 (0.3) ^a
'17 burn	13.5 (0.4) ^b	19.3 (0.2) ^b	21.0 (0.3) ^b	19.6 (0.4) ^b	16.4 (0.6) ^a
'18 burn	-	19.0 (0.4) ^b	20.6 (0.3) ^b	19.1 (0.3) ^b	16.3 (0.9) ^a

Graphics depicting these VWC and temperature trends (by plot) are presented in Figures 1.4 and 1.5, respectively. Additional results from sensors installed at KBG study plots, but with thatch manually removed (KBG-NT), are displayed in these figures. Results from KBG-NT sensors showed that soil tended to be drier at 5-cm, and wetter at 15-cm than the KBG treatment.

These results suggest that the bluegrass thatch layer reduces evapotranspiration from the surface, and may play a role in limiting water infiltration into deeper soil. However, both bluegrass treatment conditions tended to maintain higher VWC at 5- and 15-cm than either of the burned and grazed treatments throughout the season, suggesting that if thatch is limiting infiltration to deeper depths, the effect is overall trivial. Relative temperature in the KBG-NT treatment was higher and more variable than the KBG treatment, indicating that bluegrass litter on top of the soil surface moderates temperature fluctuations, and maintains cooler soil conditions during the growing season. The analysis of sensor VWC responses to rainfall revealed no significant statistical differences in relative VWC following rainfall events.

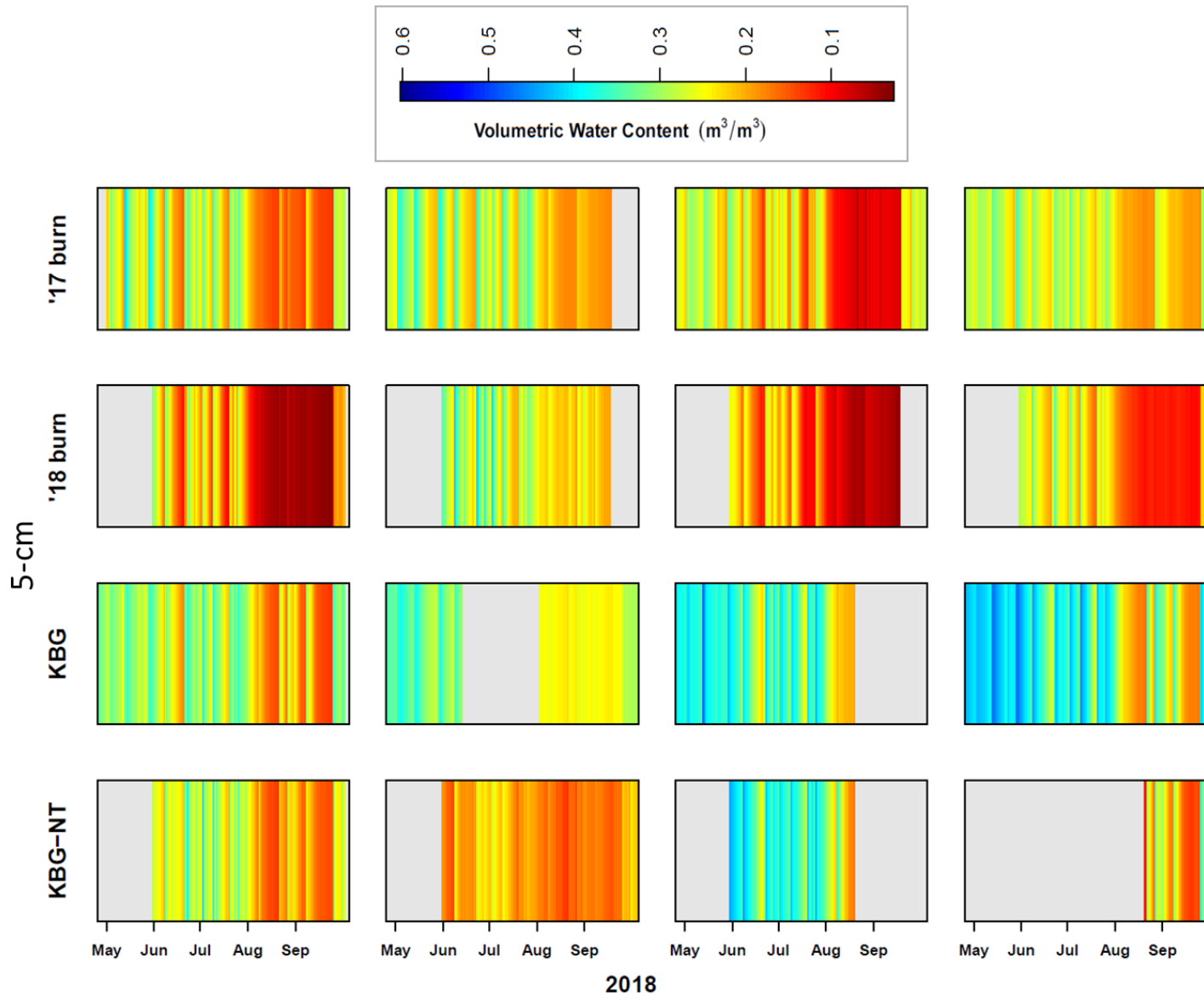


Figure 1.4. Heat map summary of volumetric water content (VWC) by month for the 2018 growing season.

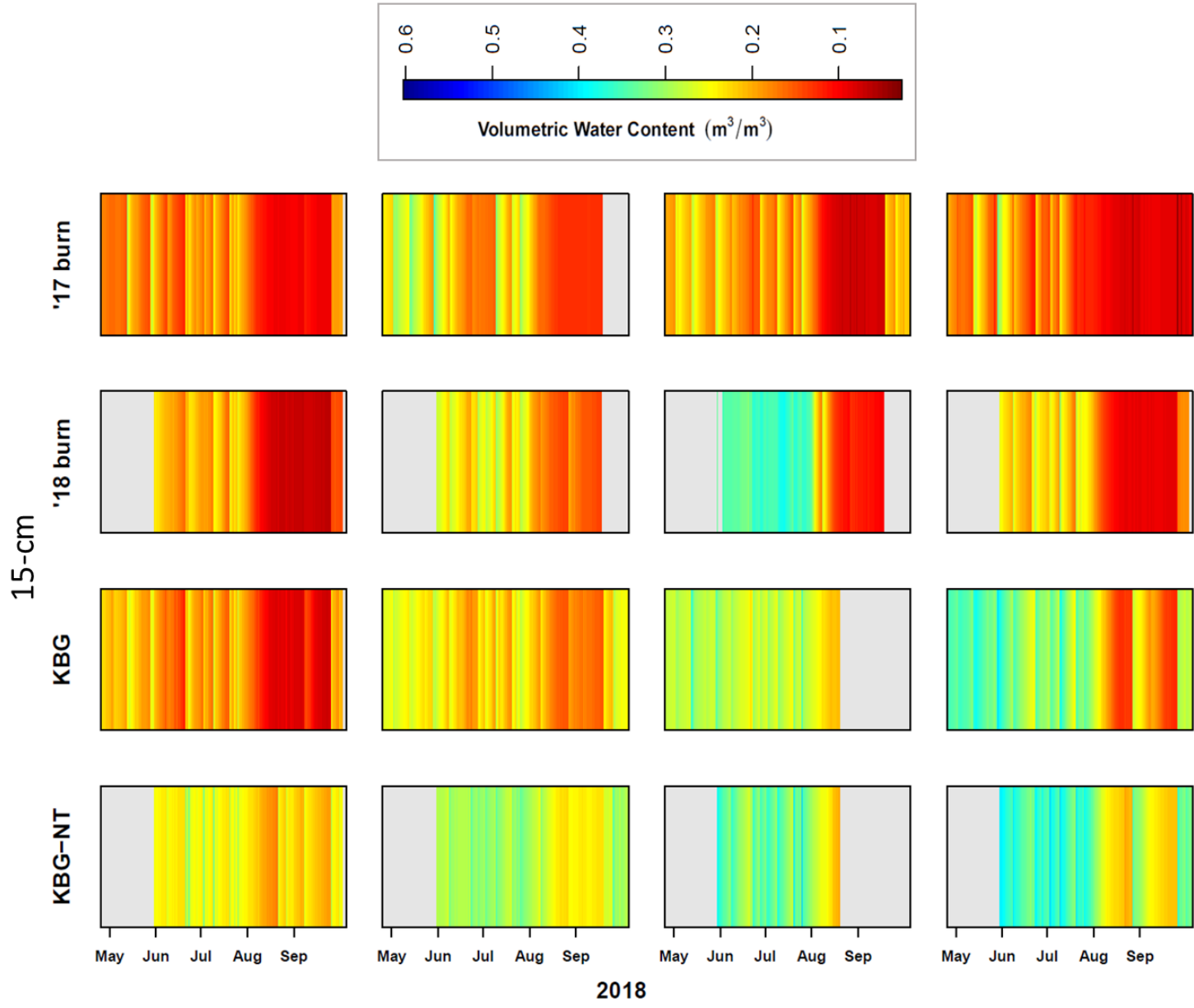


Figure 1.4. Heat map summary of volumetric water content (VWC) by month for the 2018 growing season (continued). *In situ* sensor data records of VWC at 5-cm (A), and 15-cm (B) from plots dominated by Kentucky bluegrass with thatch (KBG, $n=4$) and with thatch manually removed (KBG-NT, $n=4$), and at plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Warm and cool colors represent drier and wetter conditions, respectively, while gray represents periods of missing sensor data. Each individual box represents a treatment replicate.

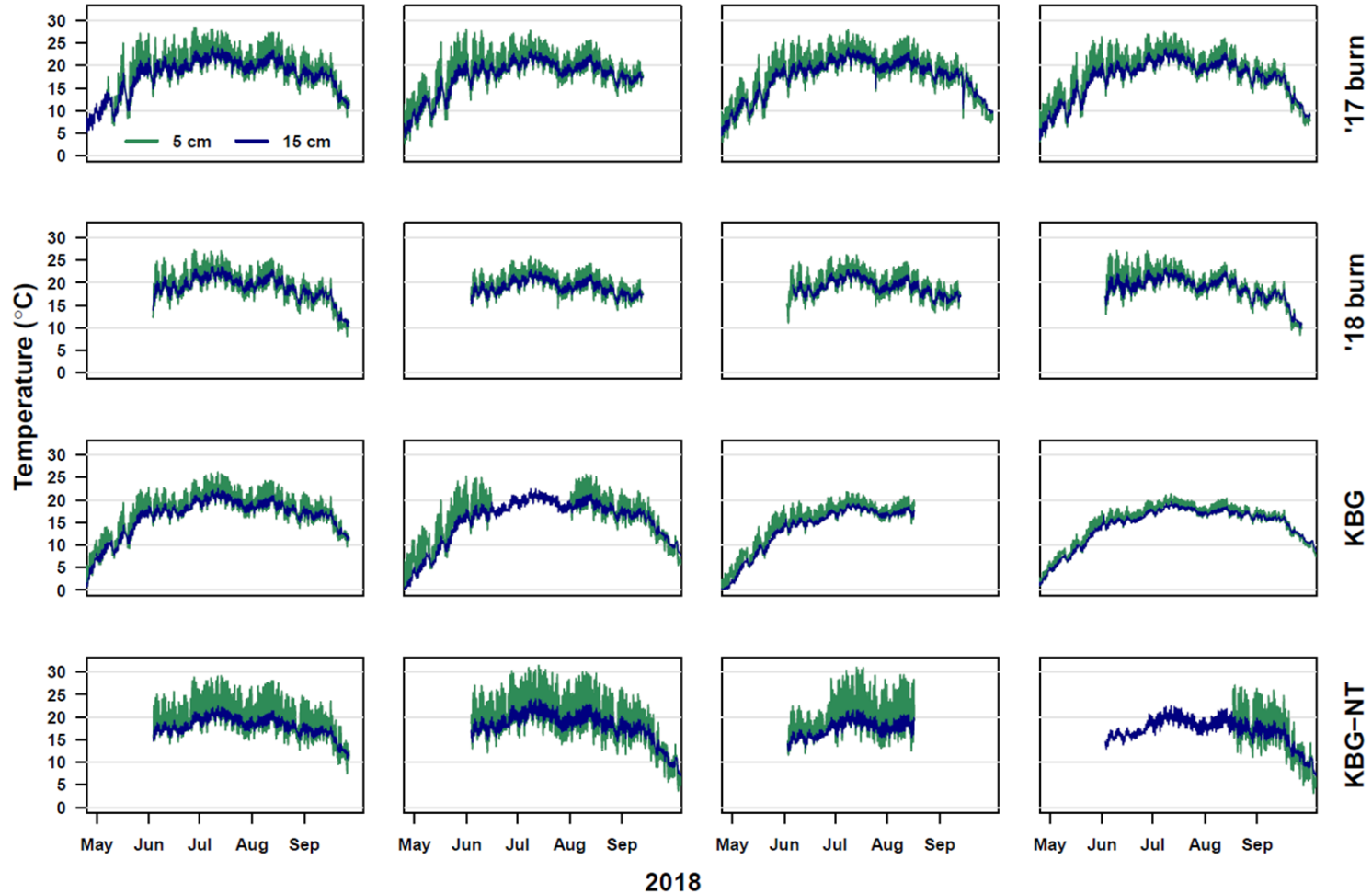


Figure 1.5. Temperature summary by month for the 2018 growing season. *In situ* sensor data records of temperature ($^{\circ}\text{C}$) at 5- and 15-cm from plots dominated by Kentucky bluegrass with thatch (KBG, $n=4$) and with thatch manually removed (KBG-NT, $n=4$), and at plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Each individual box represents a treatment replicate.

Discussion

Our results suggest that bluegrass, and the development of a thatch layer, plays an important role in shaping the abiotic soil environment. During the 2018 growing season, soil moisture was consistently higher and temperature was consistently lower under bluegrass dominated conditions compared to burned and grazed plots with greater native species composition and little to no thatch. Soil conditions under bluegrass monocultures maintained more consistency in water content and temperature, leading to more homogenous conditions overall. When above-ground vegetation and litter was removed from bluegrass plots, the soil moisture decreased in shallow soil, increased in deeper soil, and the temperature at both depths increased and was more variable. This response to the removal of the bluegrass thatch layer suggests that it was the direct cause of these shifts.

In general, lower temperatures tend to favor C₃ species such as bluegrass. Williams III (1974) found that temperature increases at increments of 5 °C resulted in the marked reduction of net photosynthesis in C₃ grass species. Additionally, a study by Christie and Delting (1982) showed that at lower temperatures, C₃ grass species had enhanced competitive ability and greater uptake and utilization of soil nitrogen resources than C₄ grass species. This trend reversed at higher temperatures. While these studies compared C₃ to C₄ grass species, and C₄ grass species make up a smaller percent of grasses in the northern Great Plains (Teeri and Stowe 1976), the results are still in line with the idea that bluegrass engineers soil conditions that are favorable to its' own growth.

The negative implications of decreased soil temperature are evident for the establishment of native prairie plant species, but in theory increased soil moisture should be beneficial considering that rainfall and soil moisture are often the most limiting factors for vegetation

growth in semi-arid grasslands (USDA NRCS ESD 2017). However, the dense rhizomatous rooting structure of bluegrass makes the species extremely competitive for water (Printz and Hendrickson 2015). Additionally, the thatch associated with bluegrass suppresses native seedling emergence (Bosy and Reader 1995). As a result, conditions associated with bluegrass will suppress the emergence of new native plant species, and bluegrass will out-compete native plant species that are already present for water resources. In essence, bluegrass will benefit from the increased soil moisture while native species will not, further contributing to the competitive advantage of bluegrass.

Data collected following both real and simulated rainfall events was not as revealing as the absolute values of VWC over the growing season. Analysis of data following rainfall events across treatments revealed no significant differences in calculated parameters (infiltration rate, runoff rate, early or steady-state infiltration, relative peak VWC magnitude, and the rate of soil drying following rainfall). The lack of significant differences, despite comprehensive analysis of a wide range of response variables, suggests that perhaps bluegrass does not significantly impact how large quantities of water enter and exit the soil profile. The differences in soil water are more likely expressed as water is retained over a greater period of time and on a smaller scale, during periods of time when evapotranspiration, as opposed to drainage, has more influence on soil moisture.

Measuring the VWC response parameters in the immediate time window following rainfall captures soil drainage processes, rather than the subtle drying effects of evapotranspiration where the differences between experimental treatments may have a more pronounced effect. Teuling et al. (2006) found that vegetation plays a significant role in the dry-down process of soil moisture through evapotranspiration, and that the largest differences in root

water uptake occur on a timescale of multiple weeks. Analysis of soil drying rate on this timescale was not possible in our study due to the sporadic nature of successive confounding rainfall events in our data sets. A more controlled experiment may be better adept at measuring this property.

Infiltration and runoff responses measured in burned and grazed study plots within the same season as the application of a prescribed burn, and one year following a prescribed burn, showed infiltration significantly increased, and runoff significantly decreased one year following management. Without pre-fire infiltration/runoff data, it is difficult to conclude the prescribed fire caused the reduction of infiltration the first year following fire, but we can definitively conclude within a year following the burn, infiltration rates recovered, and runoff rates reduced significantly as a result. These results parallel those from a study conducted by Morales et al. (2000), which found prescribed burning increased surface runoff, but this effect diminished one year after the application of the prescribed burn. This variation in response to rainfall following management with burning and grazing could potentially aid in creating heterogeneity in the plant community. These results reinforce the idea patch-burn grazing is a practical management strategy to promote biological diversity, and management effects on soil conditions from burning and grazing are short-lived.

The response of infiltration and runoff rates following fire tend to be variable (Alcañiz et al. 2018), as seen in our results. The variable tendency of infiltration and runoff measurements is likely in part due to the inherent spatial heterogeneity of infiltration characteristics, as well as to the methodology utilized in measuring this property (Paige and Stone 1997). For example, Nielsen et al. (1973) found steady state infiltration measurements in a 150-ha field taken from 20 randomly established plots varied from 0.5 mm/hr to 50 mm/hr, with a CV of 91%. Similarly,

Sharma et al. (1980) found no obvious pattern in the distribution of infiltration parameters in terms of soil type or landscape position over 26 tests carried out on a 9.6-ha watershed. Due to this inherent variation with infiltration testing methodology, it is important to also characterize spatial variability, soil structural properties, and cover characteristics simultaneously with infiltration measurements (Paige and Stone 1997). In our study, we characterized cover characteristics via vegetation surveys, as well as structural properties of our study plots through three-dimensional computed tomography scans.

Imagery assessment of internal pore structure revealed pore network patterns to be variable across study plots. In general, greater porosity was evident in bluegrass dominated treatments, as well as burned and grazed systems burned within the same year as sampling, than in burned and grazed systems burned the year prior to sampling. The trend was surprising, considering we originally expected both burned and grazed treatments to be similar, regardless of time since burning, and the bluegrass treatment would be more divergent. One important caveat that may explain these results is the study plots burned the year prior to sampling were established after significant regrowth following prescribed burns, allowing us the opportunity to select plots with greater expression of native plant diversity and minimal bluegrass expression. The burned and grazed plots sampled the same year as the prescribed burn were selected prior to burning and were heavily invaded by bluegrass. In these plots, the degree to which bluegrass was reduced and the expression of native plant communities following fire was more variable.

Ultimately, soil structure in the treatment burned and grazed the same year as sampling appeared to have soil structural characteristics more similar to the bluegrass treatment, than to the treatment burned and grazed the year prior to sampling. We interpret these results to have several possible ecological implications: 1) the short-term effects of management with burning

and grazing likely play a negligible role in soil structure and porosity, 2) bluegrass has a tendency to invade more porous soil, rather than creating soil porosity after establishment, and 3) there are certain soil types where bluegrass struggles to establish, and native species thrive in these pockets regardless of management.

While this study was not designed to test the validity or likelihood of any of these ecological implications, there is literature that supports the possibility of each. For example, to support the negligible role of fire in shaping soil structure, Scharenbroch et al. (2012) found that after twenty-three years of low-severity prescribed burns applied once every one to three years in a Midwestern oak forest, no effect was measured on soil aggregate stability. The idea that bluegrass may tend to invade more porous soil, rather than creating said porosity, is consistent with the ‘back-seat driver’ model described by Bauer (2012). In the back-seat driver model, an invasive species initially establishes in a given area due to favorable underlying environmental conditions, such as soil porosity, but with time the invasive species begins to drive change in native plant communities. In support of the theory there are certain soil types where bluegrass struggles to establish, and native species thrive, Cully et al. (2003) describes ecological pockets comprised of rocky outcrops and shallow soils, where fragments of the remaining 1% of the original native tallgrass prairie exist despite pressures from invasive species. In the case of bluegrass invasions, a combination of these three scenarios are likely acting simultaneously.

Considering the current extent of bluegrass invasion in the northern Great Plains, if there are soil types or environmental conditions that inhibit bluegrass establishment, these conditions likely make up only small discrete areas of land scattered discontinuously across the landscape, which would not provide adequate provisioning ecosystem services long-term. For this reason, land management practices should be adopted that control bluegrass, and promote heterogeneity

and biodiversity on a large-scale. The results of this study suggest that periodically applying fire to patches of the landscape, in combination with grazing, is an effective and efficient management tool that can reduce bluegrass cover and shift abiotic soil properties, such as moisture and temperature, away from conditions that favor bluegrass dominance.

Whether soil type does or does not influence the likelihood that bluegrass will invade a given area, once fully established, bluegrass does appear to play a significant role in shaping soil microclimate. As the soil environment shifts towards wetter and cooler conditions under bluegrass monocultures, this novel environment could have implications on other soil properties, such as microbial populations, nutrient pools, and decomposition. Potentially, the widespread invasion of bluegrass could trigger large-scale impacts to the ecosystem, and maintain a steady-state that is different from historical conditions.

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CHAPTER 2: SOIL CHEMISTRY, MICROBIAL COMMUNITIES, AND BIOLOGICAL PROCESSES ASSOCIATED WITH INVASIVE PERENNIAL GRASS AND ALTERNATIVE LAND MANAGEMENT TECHNIQUES

Abstract

Kentucky bluegrass (*Poa pratensis* L.; hereafter referred to as bluegrass) is an invasive perennial grass that has become naturalized throughout the entire United States. This species is an effective competitor and produces abundant litter, which in turn creates a thick thatch layer. In the northern Great Plains, disturbances such as fire and grazing are being explored as tools to reduce this thatch layer and promote native grass and forb diversity. The overall objective of this research is to examine below-ground characteristics under accumulated thatch and in response to management with fire and grazing. We hypothesized that the dominance of bluegrass and development of thatch strongly influences the soil microclimate and the nature of organic matter inputs, which, in turn, affects nutrient and microbial dynamics. Soil samples were collected from plots dominated by bluegrass with thatch, and from plots one year after management with fire and season-long grazing, where native plant communities had recovered. To understand immediate and in-season effects of fire and grazing on soil properties, we also sampled soil immediately before, and at increasing time steps after spring fires and exposure to grazing aimed to reduce the thatch layer. Soils were analyzed for carbon and nitrogen fractions, microbial abundance and community structure, and decomposition rates. Results show that below-ground pools of carbon, nitrogen, and microbial communities are resilient to changes in management practices, and the composition of expressed vegetative cover. However, prescribed burning caused a brief decline of labile carbon, as well as decomposition in shallow soil, but both properties rebounded by one year after burning. Our observations indicate that the combination

of fire and grazing has minimal impact on the soil, and is effective at removing thatch and reducing bluegrass, though the response of the vegetation community following management is variable.

Introduction

Natural resource management practices in grassland systems have been based on a developing progression of human needs and values since settlement began in the Great Plains region. Historically, in the northern Great Plains ungulate herbivory shifted over the landscape in response to fire, and this process played an important role in determining plant communities (Samson et al. 2004). Fire and grazing predominantly drove vegetation patterns on the landscape for centuries, until the arrival of Euro-Americans. In the late 19th century, a series of federal acts passed in the United States that led to the mass conversion of native grasslands to agricultural land (Samson et al. 2004). This conversion of land, coupled concurrently with anthropogenic disruption to fire and grazing patterns, led to the exploitation and degradation of rangelands on a large scale (Briske 2017). Today, these grassland systems are among the most threatened in the United States (Samson et al. 2004).

While agricultural expansion continues in the northern Great Plains (Wright and Wimberly 2013; Johnston 2014), anthropogenic influence in this region has shifted marginally from exploitation towards stewardship in recent years (Chapin et al. 2009). While land owners historically perceived rangelands as a limitless resource for grazing (Briske 2017), and fire as a potential for liability that must be suppressed (Wonkka et al. 2015), many modern land owners have a better understanding of the importance of natural disturbances and heterogeneity in these grassland systems (Briske 2017). Unfortunately, we must still cope with unforeseen ecological repercussions from historical, and continued, mismanagement.

One such repercussion is the invasion of non-native species such as Kentucky bluegrass (*Poa pratensis* L.; hereafter referred to as bluegrass). Often, when there are shifts in fire regime, conditions develop that facilitate the invasion of non-native species such as bluegrass (D'Antonio 2000). In addition, bluegrass is a grazing-tolerant species, so historically over-grazing grasslands may have displaced native species and facilitated the invasion of bluegrass (Toledo et al. 2014). Bluegrass has rapidly become one of the predominant species in the Great Plains (USDA 2014), threatening biodiversity of native plant species, and ecosystem provisioning services (such as forage production).

Bluegrass also has the potential to alter soil properties, in part due to the formation of a thickened thatch layer. Thatch refers to the accumulation of litter and root biomass on top of, and intermingled with, the soil surface. In the case of bluegrass, this excess litter may alter the microclimate of the soil by acting as a buffer between environmental moisture and temperature (Bossy and Reader 1995; Pierson et al. 2002), which can have an extending effect on microbial activity and nutrient cycling. Monocultures of bluegrass have been found to cause lower soil C:N ratios than native C₄ grass species (Wedin and Pastor 1993). As the C:N ratio of residue inputs and a soil shift, subsequent soil processes may also be impacted. Bradley et al. (2006) found that increased nitrogen levels in grassland systems led to significant changes in the composition and relative abundance of soil microbial communities by increasing bacterial abundance, and decreasing fungal abundance. Increased nitrogen is also generally associated with increased decomposition rates, however, Knorr et al. (2005) suggests that increased nitrogen can have an inhibitory effect on certain decomposers, which may in turn dampen the stimulating effect of increased nitrogen.

Competition for nutrients and biological associations plays an important role in structuring vegetation. Excess nitrogen may shift the competitive advantage away from native plant species and towards species that have adapted to nitrogen-enhanced systems, such as bluegrass (Wedin and Tilman 1990; Dornbusch et al. 2018a). This may lead to a synergistic feedback cycle perpetuating the dominance of bluegrass, and further facilitating associated changes in soil properties that favor bluegrass. Overall, bluegrass dominance has the potential to impact a wide array of soil properties and ecological processes, due to its invasion extent and dominance, as well as its morphological and phenological differences from the historic native plant community.

To control bluegrass, and minimize any impacts that the species may have on the soil, we can guide our land management choices to incorporate adaptive grazing and fire techniques, which can diminish the competitive advantage of bluegrass. When applied to a landscape individually, grazing and fire have varied impacts on bluegrass. In general, studies show that grazing alone has a limited impact on bluegrass control (Biondini et al. 1998; Rogers et al. 2005; Otfinowski, et al. 2017). Overgrazing and the exclusion of grazing have also been known to increase bluegrass levels (Grant et al. 2009). Fire, on the other hand, has been found to consistently reduce bluegrass cover, although the degree of control depends on fire regime as well as initial bluegrass invasion level (McMurphy and Anderson 1965; Towne and Kemp 2008; Ereth et al. 2017; Kral et al. 2018). Kral et al. (2018) found that the decline in bluegrass cover following late-growing season and dormant season burns persists at least three years post-fire. A combination of grazing and burning has been shown to increase plant species diversity more than either process alone (Harnett et al. 1996).

Fire and grazing have varied impacts on soil. A 2018 review paper found that the effects of prescribed fires on soil properties depended on parameters such as fire regime, soil type, seasonality, residence time, and periodicity (Alcañiz et al. 2018). This review evaluated the impact of prescribed fire on a variety of soil properties, including chemical and biological properties. Soil nitrogen generally increased after fire, though excessively high fire temperatures (>200 °C) may lead to N volatilization (Alcañiz et al. 2018). Likewise, carbon stocks in the soil often increase after low-intensity fires due to the incorporation of partially burned organic matter, while high-intensity fires typically decreased carbon stocks (Alcañiz et al. 2018). The effect of fire on soil organisms was extremely variable. Results ranged from complete elimination of soil organisms (Choromanska and DeLuca 2001; Switzer et al. 2012; Williams et al. 2012; Barreiro et al. 2015), to increased microbial activity resulting from the influx of ash materials (Blankenship and Arthur 1999; Gray and Dighton 2009; Fultz et al. 2016).

Grazing may also play a role in shaping soil microbial populations, based on the results from a 2017 meta-analysis on the subject (Zhao et al. 2017). Zhao et al. (2017) found that heavy grazing significantly decreased total microbial, bacterial, and fungal community size, yet brief, light, and moderate grazing had no effect on these properties. Furthermore, grazing intensity may also impact soil carbon and nitrogen pools. In a grazing intensity trial in the mixed-grass prairie of North Dakota, Frank et al. (1995) found that soils in sites excluded from grazing had higher nitrogen content than in moderately or heavily grazed pastures. In this study, moderate grazing also decreased carbon content yet heavy grazing did not, likely due to changes in species composition resulting from the disturbance (Frank et al. 1995).

Since fire and grazing often occur simultaneously on a landscape, we must also consider the combined impact that these two disturbances may have on soil properties. A study in the tall

grass prairie of Kansas found that grazing increased net N mineralization, while annual burning decreased that same property (Johnson and Matchett 2001). These results suggest that fire and grazing may have contrasting effects on soil properties, and also that these disturbances may impact the soil processes that transform nutrients between pools. There is currently limited research on the coupled effect that fire and grazing (and subsequent shifts in plant species assemblages) may have on soil properties in the grasslands of the northern Great Plains.

One adaptive management approach that combines grazing and fire is known as patch-burn grazing. Patch-burn grazing is a management model based on the concept of pyric herbivory, i.e. grazing patterns that are influenced by fire (Fuhlendorf et al. 2009). In these systems, spatially and temporally discrete fires are applied to the landscape, subsequently increasing the likelihood that the burned areas will be grazed (Fuhlendorf et al. 2009). While this management approach mimics natural pyric herbivory patterns, there are inherent differences between these managed systems and pre-settlement disturbance regimes. Prescribed burns often vary from wildfires in seasonality, intensity, extent, frequency, and type, in order to ensure safe and effective prescribed burning practices. Historically, grazing was dominated by bison with free range to roam and graze across the landscape, while patch-burn grazing management generally utilizes cattle, confined to the fenced boundaries of a pasture. Despite these differences, over time, patch-burn grazing creates a mosaic of patches shifting in disturbance level across the landscape, which promotes heterogeneity in successional diversity.

While the eradication of bluegrass from grasslands in the northern Great Plains is not a realistic option due to the extent of the invasion, Dornbusch et al. (2018b) found that patch-burn grazing reduces the cover of bluegrass, as well as the influence of bluegrass on plant community composition over time compared to traditional management. With bluegrass established as a

permanent fixture in these ecosystems, and the need for adaptive management (such as patch-burn grazing) required on a greater scale, we must evaluate ecosystem effects, including any potential impacts to below-ground soil properties, resulting from both novel bluegrass ecosystems, as well as land management practices. Just as bluegrass has the potential to affect nutrient cycling and biological properties of the soil, so does patch-burn grazing.

The goal of our research was to measure the effect that bluegrass, and management with fire and grazing, may have on soil nutrient pools, microbial communities, and biological processes. We designed an experiment to identify differences in these soil properties between burned and grazed systems managed in the context of a larger patch-burn grazing management model, and systems dominated by bluegrass. Between the idle (bluegrass dominated) and managed systems, we compared carbon and nitrogen fractions in both labile and stable soil pools, as well as microbial abundance and community structure. These measurements were stratified through time to understand the impacts of fire and grazing on soil properties before and immediately after fire, as well as one year after fire. Additionally, we conducted in-field litterbag incubations to determine if any potential shifts in soil properties affect litter decomposition. Results will help us to understand how both bluegrass and land management techniques may impact soil properties, and ultimately provide insight into key determinants of subsequent plant growth.

Methods

Study Area

This study was conducted during the summer of 2018 in the mixed-grass prairie at North Dakota State University's Central Grasslands Research Extension Center (CGREC) in south-central North Dakota. The research center is located near Streeter, North Dakota, USA, which

lies within the Missouri Coteau region. Prairies in this region are characterized by the dominance of cool season, perennial grasses (Toledo et al. 2014). We conducted our study in thin loamy ecological sites, which have a historical climax plant community including green needlegrass (*Nassella viridula*), little bluestem (*Schizachyrium scoparium*), and western wheatgrass (*Pascopyrum smithii*) (USDA NRCS ESD 2019). However, following extended periods of idle management, excessive litter will accumulate resulting in conditions that favor bluegrass (USDA NRCS ESD 2019).

Extreme temperatures and light rainfall are characteristic in this region. Rainfall is typically the limiting factor for vegetation, with annual precipitation ranging from approximately 38-51-cm per year (USDA NRCS ESD 2017), the majority of which falls during the growing season months. During the 2018 growing season the study area received above average rainfall, with precipitation totals reaching 44.2-cm between April and September (NDAWN 2019). Soils in the study area are typically well drained and have low runoff depending on the slope and the vegetative cover (USDA NRCS ESD 2017). Experimental plots were selected from map units identified as the Zahl-Williams-Zahill complex by the USDA NRCS online Web Soil Survey (2017). Typical soil profiles for these units include a thin layer of loam (0-15-cm), on top of clay loam extending through the substrata (USDA NRCS WSS 2017). For the 2018 growing season, soil temperature under turf vegetation averaged 14 °C (NDAWN 2019). Specific study plots were analyzed for general soil properties, summarized in the *Results* section of this chapter.

Experimental Design

In 2017, we established eight original sampling plots for this study distributed across the research center. The plots were selected based on the historical management of the land, and a visual assessment of expressed plant community which we validated through plant community

surveys. Half of the plots were composed of predominantly native vegetation, representing the reference plant community, and the remaining half were dominated by bluegrass.

The plots established in 2017 with predominantly native plant species were managed with fire and grazing, in the context of a larger patch-burn grazing management system (hereafter referred to as '17 burn, $n=4$). We burned these plots in the spring of 2017, in conjunction with season-long, moderate grazing between mid-May and October. The pastures are stocked at a rate so that approximately 50% of the vegetation produced during the growing season remains after grazing. In contrast, the bluegrass dominated plots (hereafter referred to as KBG, $n=4$) were excluded from grazing, and have not been burned in recent decades. These conditions have resulted in bluegrass dominance and the accumulation of a thick thatch layer. Additional burned and grazed plots were added in 2018 (hereafter referred to as '18 burn, $n=4$) following the spring burns in order to compare to those burned in 2017 which had a year to recover post-disturbance when sampled in 2018.

At the beginning of the 2018 growing season, immediately following the application of prescribed burns to the '18 burn treatment, soil samples were collected from all treatments for analysis of general, chemical, and biological properties. This data helped us to determine if soil properties varied across treatments. Additional soil samples were collected from the '18 burn treatment throughout the 2018 growing season, which provided insight as to how soil properties changed over time in response to management. These plots were sampled immediately before and after the spring burn occurred, and then one- and three-months after spring fires and exposure to grazing.

For each sampling event throughout the season, we collected six samples (each divided into depths of 0-5 and 5-15-cm) to form one composite sample at each depth for each study plot,

resulting in a total of 144 samples throughout the 2018 season. The six samples per plot were collected randomly from a 5-meter radius around a GPS recorded center-point. Additionally, we measured decomposition through year-long litterbag incubations, the methodology of which is described in greater detail within the *Data Collection* section of this paper.

Data Collection

General soil properties

We measured a range of general soil properties from our study plots including texture, pH, electrical conductivity (EC), bulk density (ρ_b), and water content. Soil texture was determined based on the USDA soil particle size classification system, and we used the hydrometer method outlined by Gee and Or (2002) to analyze particle size. We determined EC and pH using the 1:1 soil to water ratio methods outlined respectively by Rhoades (1996), and Thomas (1996). We measured bulk density by averaging three replicate soil samples of known volume (2.5-cm radius x 15-cm height cylinder) from 0-15-cm depth, collected via hammer corer from each study plot. Bulk density samples were collected and processed according to protocol established in Blake and Hartge (1986). Lastly, water content (θ_g) was determined on a gravimetric dry-mass basis through mass loss (g) of field moist soil samples oven dried to constant weight, as outlined in Gardner (1986). Volumetric water content (θ_v) was calculated by multiplying gravimetric water content by the bulk density of the soil ($\theta_v = \theta_g * \rho_b$).

Carbon

Carbon properties measured include total carbon, organic carbon, inorganic carbon, permanganate oxidizable carbon (POXC), and microbial biomass carbon (MBC). For total, organic, and inorganic carbon fractions, soil samples were air dried and then ground to pass a 250-micron sieve, as recommended in Nelson and Sommers (1996). Total carbon was

determined by dry-combustion through Elementar analysis, while inorganic carbon was determined on a Skalar analyzer system. We then calculated organic carbon as the difference between total and inorganic carbon.

The remaining carbon pools, POX C and MBC represent labile soil carbon fractions. POX C was determined following the procedure from Weil et al. (2003). We measured MBC concentrations through the chloroform fumigation-extraction method (Beck, et al. 1997; Joergensen, et al. 1996; Vance et al. 1987).

Nitrogen

Nitrogen pools measured include total nitrogen, inorganic nitrogen (ammonium and nitrate), and potentially mineralizable nitrogen (PMN). Soil samples for these analyses were air-dried rapidly following collection, and then stored in air-tight containers to limit microbial processes that may alter nitrogen levels. Air-drying soils has become a widely adopted method in preserving soil for the analysis of inorganic nitrogen forms, however some research suggests that analysis of field-moist soils, frozen in air-tight containers, is a more effective method (Nelson and Bremner 1972). However, due to the low initial levels of inorganic nitrogen in our study plots, we determined the air-drying method to be sufficient. Samples for analysis of total nitrogen were ground to pass a 250-micron sieve, while the samples for inorganic nitrogen were ground to pass a 2.00-millimeter sieve. Total nitrogen was determined by Elementar analysis. We performed potassium chloride extractions on samples prior to inorganic nitrogen analysis (on an auto-analyzer), following procedures outlined in Robertson et al. (1999).

Lastly, we measured PMN, an indicator of the nitrogen mineralization capacity of the soil's microbial community. We determined PMN using the 7-day anaerobic incubation method

outlined by Drinkwater et al. (1996). The difference between inorganic soil nitrogen measured in incubated and control samples represents PMN.

Phospholipid fatty acid analysis (PLFA)

We used phospholipid fatty acid (PLFA) analysis to determine microbial community structure. This method provides estimates of the number and relative abundance of microbial populations at a broad taxonomic level. Samples were stored on ice during transport and then frozen until processing. Upon processing, frozen samples were lyophilized and then ground to pass a 2.00-millimeter sieve. PLFA analysis was conducted by Microbial Identification (MIDI) Labs, Inc. (Newark, DE). The MIDI lab follows lipid extraction procedures described by Buyer and Sasser (2012), quantitative analysis with gas chromatography (HP6890, Hewlett Packard, Palo Alto, CA), peak identification using Sherlock software version 6.2 (MIDI, Inc., Newark, DE) and the PLFAD2 version 2.0 peak naming table. We analyzed microbial groups in terms of absolute abundance (nmol fatty acid / g soil), relative abundance (group / total abundance, as percent), and Fungal:Bacterial ratio across treatments.

Decomposition

To measure decomposition across treatments we installed litterbags, stratified by depth, at each plot. Litterbags were fitted on top of surface vegetation, and at depths of 5- and 15-cm in the soil profile for a year-long incubation period (three replicate litterbags per depth, per plot). At each bluegrass plot, an additional set of triplicate litterbags were installed within the bluegrass thatch layer, positioned above the soil surface but below accumulated vegetative biomass. We filled the litterbags with bluegrass litter collected from our study area. Standing and fallen bluegrass litter was included, while bluegrass thatch that was intermingled with the soil surface was excluded in order to minimize the inclusion of mineral soil. Chemical analysis of standing

and fallen bluegrass biomass revealed that the litter had an average C:N ratio of 29:1. Protocol for the preparation, installation, retrieval, and processing of the litterbags was modified from methods described by Bocoock and Gilbert (1957). Decomposition was estimated from mass loss (%) of litter after the in-field incubation period.

Data Analysis

Statistical differences across treatments were analyzed using a one-way analysis of variance (ANOVA) test with Tukey's honest significance test used for mean comparisons. To explore how variables change through time in the '18 burn treatment, we used a repeated-measures ANOVA. Additional analyses include: 1) computation of pairwise correlation coefficients to better understand relationships between selected properties, 2) Principal Component Analysis of shifts in microbial communities, and 3) a paired *t*-test to assess how decomposition may vary within a study plot between the bluegrass thatch canopy and the exposed surface. All analysis was performed in JMP[®], Version 13. SAS Institute Inc., Cary, NC, 1989-2019, and statistically significant differences were determined at a $p \leq 0.05$.

One-way ANOVAs assume that a population is normally distributed, has uniform variances, and samples are independent. We have no reason to believe that that populations in this study are not normally distributed, or that the variances are not equal, and we are limited in our ability to assess these characteristics across $n = 4$ samples. While the unique pasture histories across experimental plots in this study could possibly affect the independence of samples, many studies show that soil properties become independent within 5-10 meters – across many different ecosystems, including grasslands (summarized by Ettema and Wardle 2002; Ritz et al. 2004). Additional differences in topography, land use history, and soils all increased the spatial

independence of experimental units. Statistical analyses prior to publication efforts may re-evaluate data using non-parametric, linear, and/or generalized mixed-effect models.

Results

Since soil properties selected for measurement in this study vary with soil depth, we mainly assessed treatment differences within depth, rather than between depths. Results for general soil properties are reported in Table 2.1. Measured properties were fairly variable, despite best efforts to minimize inherent soil disparities across plots during experimental design.

Table 2.1. General soil properties across study sites at the North Dakota State University Central Grasslands Research Extension Center. Mean values (with ranges reported in parentheses) of soil pH, electrical conductivity (EC), sand (%), and clay (%), within depths of 0-5 and 5-15-cm at North Dakota State University Central Grasslands Research Extension Center study plots during the 2018 growing season ($n=12$ plots). Bulk density of the soil was measured for the depth 0-15-cm ($n=36$, 3 cores at 12 plots).

Depth (cm)	pH	EC ($\mu\text{S}/\text{cm}$)	Sand (%)	Clay (%)	Bulk density (g/cm^3)
0-5	6.7 (6.15-7.23)	387 (257-535)	36 (25-53)	26 (11-35)	0.9 (0.78-1.05)
5-15	6.9 (6.28-7.72)	400 (198-805)	40 (21-68)	27 (14-39)	

Overall, there were few significant differences in carbon and nitrogen pools across treatments. However, we did find significant differences in POX C, one of the most highly labile carbon pools. POX C was significantly lower in the '18 burn treatment immediately following fire, at both 0-5 and 5-15-cm depths, when compared to the KBG and '17 burn treatments (Table 2.2). Although no other significant differences were measured, an interesting trend was observed in shallow soils (0-5-cm) between the KBG and '17 burn treatments. The KBG treatment mean values were consistently lower than those of the '17 burn treatment in shallow soils, with the

singular exception of NH_4 being higher in the KBG treatment. While not as uniform, this trend was also present at 5-15-cm, with exceptions being the PMN, NH_4 , and NO_3 pools.

Our analysis revealed no significant changes over time from seasonal sampling of the '18 burn treatment (Table 2.3). To better understand if there were any meaningful trends, or relationships driving the fluctuations in these pools, we analyzed correlations between variables, and included volumetric soil moisture as an additional variable in the analysis. The results of the correlation analysis are reported in Figure 2.1. We found significant correlations between volumetric water content and PMN and NH_4 , as well as between total N and several carbon pools.

Table 2.2. Nutrient pool mean values from sites dominated by invasive Kentucky bluegrass, and sites managed with patch-burn grazing in 2017 and 2018. Mean absolute values (with standard deviations reported in parentheses) of soil carbon and nitrogen fractions (mg/kg) at depths of 0-5-cm (top) and 5-15-cm (bottom), from plots dominated by Kentucky bluegrass (KBG, $n=4$), and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Carbon fractions measured include total carbon (Total C), organic carbon (Organic C), permanganate oxidizable carbon (POX C), and microbial biomass carbon (MBC). Nitrogen fractions measured include total nitrogen (Total N), potentially mineralizable nitrogen (PMN), ammonium (NH₄), and nitrate (NO₃). Measurements were taken in April, 2018 at the North Dakota State University Central Grasslands Research Extension Center. Letters (a, b) represent statistically significant differences across treatments, within a nutrient pool at an individual depth ($p \leq 0.05$).

Depth: 0-5-cm			
	KBG	'17 burn	'18 burn
Total C (mg/kg)	56800 (5239) ^a	59625 (6523) ^a	56600 (13227) ^a
Organic C (mg/kg)	56400 (5114) ^a	58800 (6908) ^a	56475 (13118) ^a
POX C (mg/kg)	1420 (79) ^a	1431 (111) ^a	1134 (191) ^b
MBC (mg/kg)	1117 (169) ^a	1228 (170) ^a	1120 (234) ^a
Total N (mg/kg)	5175 (465) ^a	5525 (287) ^a	5225 (981) ^a
PMN (mg/kg)	160 (32) ^a	165 (19) ^a	169 (5.00) ^a
NH ₄ (mg/kg)	22 (6.04) ^a	17 (5.32) ^a	20 (6.57) ^a
NO ₃ (mg/kg)	2.15 (0.01) ^a	3.73 (2.00) ^a	2.67 (1.06) ^a
Depth: 5-15-cm			
	KBG	'17 burn	'18 burn
Total C (mg/kg)	36350 (3426) ^a	38425 (4424) ^a	35475 (7871) ^a
Organic C (mg/kg)	34625 (1338) ^a	35775 (2170) ^a	35350 (8093) ^a
POX C (mg/kg)	967 (82) ^a	1007 (75) ^a	768 (109) ^b
MBC (mg/kg)	663 (21) ^a	720 (189) ^a	677 (82) ^a
Total N (mg/kg)	3675 (206) ^a	3725 (222) ^a	3525 (680) ^a
PMN (mg/kg)	74 (8.71) ^a	72 (15) ^a	84 (14) ^a
NH ₄ (mg/kg)	12 (3.08) ^a	12 (5.81) ^a	13 (1.39) ^a
NO ₃ (mg/kg)	2.13 (0.03) ^a	2.13 (0.04) ^a	2.66 (1.06) ^a

Table 2.3. Nutrient pool mean values over time from sites managed with patch-burn grazing in 2018. Mean absolute values (with standard deviations reported in parentheses) of soil carbon and nitrogen fractions (mg/kg) at depths of 0-5-cm (top) and 5-15-cm (bottom), from plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2018 ($n=4$). Values were measured immediately before (pre-burn) and after (post-burn) the prescribed burn performed in the spring of 2018, and at increasing time steps throughout the 2018 season (1 mo. and 3 mo.). Carbon fractions measured include total carbon (Total C), organic carbon (Organic C), permanganate oxidizable carbon (POX C), and microbial biomass carbon (MBC). Nitrogen fractions measured include total nitrogen (Total N), potentially mineralizable nitrogen (PMN), ammonium (NH₄-N), and nitrate (NO₃-N). Measurements were taken between April and August, 2018 at the North Dakota State University Central Grasslands Research Extension Center. Statistically significant differences were assessed at $p \leq 0.05$.

Depth: 0-5-cm				
	pre-burn	post-burn	1 mo.	3 mo.
Total C (mg/kg)	54300 (14266) ^a	56600 (13227) ^a	60075 (16592) ^a	58675 (11067) ^a
Organic C (mg/kg)	53975 (13812) ^a	56475 (13118) ^a	59800 (16476) ^a	58550 (11173) ^a
POX C (mg/kg)	1141 (362) ^a	1135 (191) ^a	1271 (157) ^a	1121 (120) ^a
MBC (mg/kg)	1252 (175) ^a	1120 (234) ^a	1193 (220) ^a	1004 (126) ^a
Total N (mg/kg)	5100 (1036) ^a	5225 (981) ^a	5575 (1417) ^a	5375 (885) ^a
PMN (mg/kg)	144 (31) ^a	169 (5.00) ^a	165 (12) ^a	171 (29) ^a
NH ₄ (mg/kg)	25 (15) ^a	21 (6.57) ^a	36 (9.70) ^a	9.14 (1.10) ^a
NO ₃ (mg/kg)	13 (18) ^a	2.67 (1.06) ^a	4.42 (2.60) ^a	2.15 (0.01) ^a
Depth: 5-15-cm				
	pre-burn	post-burn	1 mo.	3 mo.
Total C (mg/kg)	34275 (7730) ^a	35475 (7871) ^a	34675 (8894) ^a	35475 (8159) ^a
Organic C (mg/kg)	33625 (7762) ^a	35350 (8093) ^a	34300 (8765) ^a	35050 (8545) ^a
POX C (mg/kg)	626 (74) ^a	768 (109) ^a	834 (122) ^a	694 (157) ^a
MBC (mg/kg)	650 (80) ^a	677 (82) ^a	652 (79) ^a	574 (57) ^a
Total N (mg/kg)	3350 (580) ^a	3525 (680) ^a	3450 (819) ^a	3600 (796) ^a
PMN (mg/kg)	63 (28) ^a	84 (14) ^a	74 (15) ^a	76 (15) ^a
NH ₄ (mg/kg)	7.54 (3.62) ^a	13 (1.39) ^a	22 (7.86) ^a	7.50 (2.29) ^a
NO ₃ (mg/kg)	5.64 (5.56) ^a	2.66 (1.06) ^a	2.68 (1.07) ^a	2.14 (0.02) ^a

Depth: 0-5-cm

Depth: 5-15-cm	Total C	r=0.99***	r=0.44*	r=0.16	r=0.98***	r=-0.001	r=-0.09	r=0.03	r=-0.14
	r=0.95***	Organic C	r=0.43*	r=0.15	r=0.98***	r=0.01	r=-0.09	r=0.03	r=-0.14
	r=0.51*	r=0.47*	POX C	r=0.28	r=0.41*	r=-0.17	r=0.19	r=-0.13	r=0.18
	r=0.14	r=-0.01	r=0.16	MBC	r=0.26	r=0.04	r=0.26	r=0.15	r=-0.06
	r=0.96***	r=0.97***	r=0.55**	r=0.05	Total N	r=0.05	r=-0.06	r=0.03	r=-0.18
	r=0.36	r=0.38	r=0.12	r=0.02	r=0.31	PMN	r=-0.04	r=-0.17	r=-0.02
	r=-0.04	r=-0.03	r=0.16	r=0.27	r=-0.03	r=0.16	NH₄	r=-0.31	r=0.63***
	r=-0.25	r=-0.28	r=-0.34	r=0.04	r=-0.27	r=-0.62**	r=-0.26	NO₃	r=0.07
	r=-0.05	r=-0.06	r=-0.05	r=0.02	r=-0.11	r=0.53**	r=0.47*	r=-0.18	Vol. H₂O

Figure 2.1. Correlation between measured variables. Pairwise correlation coefficients were computed for soil data collected at the North Dakota State University Central Grasslands Research Extension Center in Streeter, ND throughout the 2018 growing season. Statistically significant correlations are indicated by *, **, and *** ($p \leq 0.05$, $p \leq 0.01$, and $p \leq 0.001$ respectively). Correlation coefficients greater than 0.40 are displayed in bold red.

Analysis of the absolute abundance of microbial groups and the Fungal:Bacterial ratios across treatments revealed no significant differences in total microbial abundance, individual groups across treatments, or in response to fire (Fig. 2.2). Principal Component Analysis did not reveal any distinction between treatments, based on community shifts. Means and standard deviations of microbial taxonomic groups are reported in the appendix.

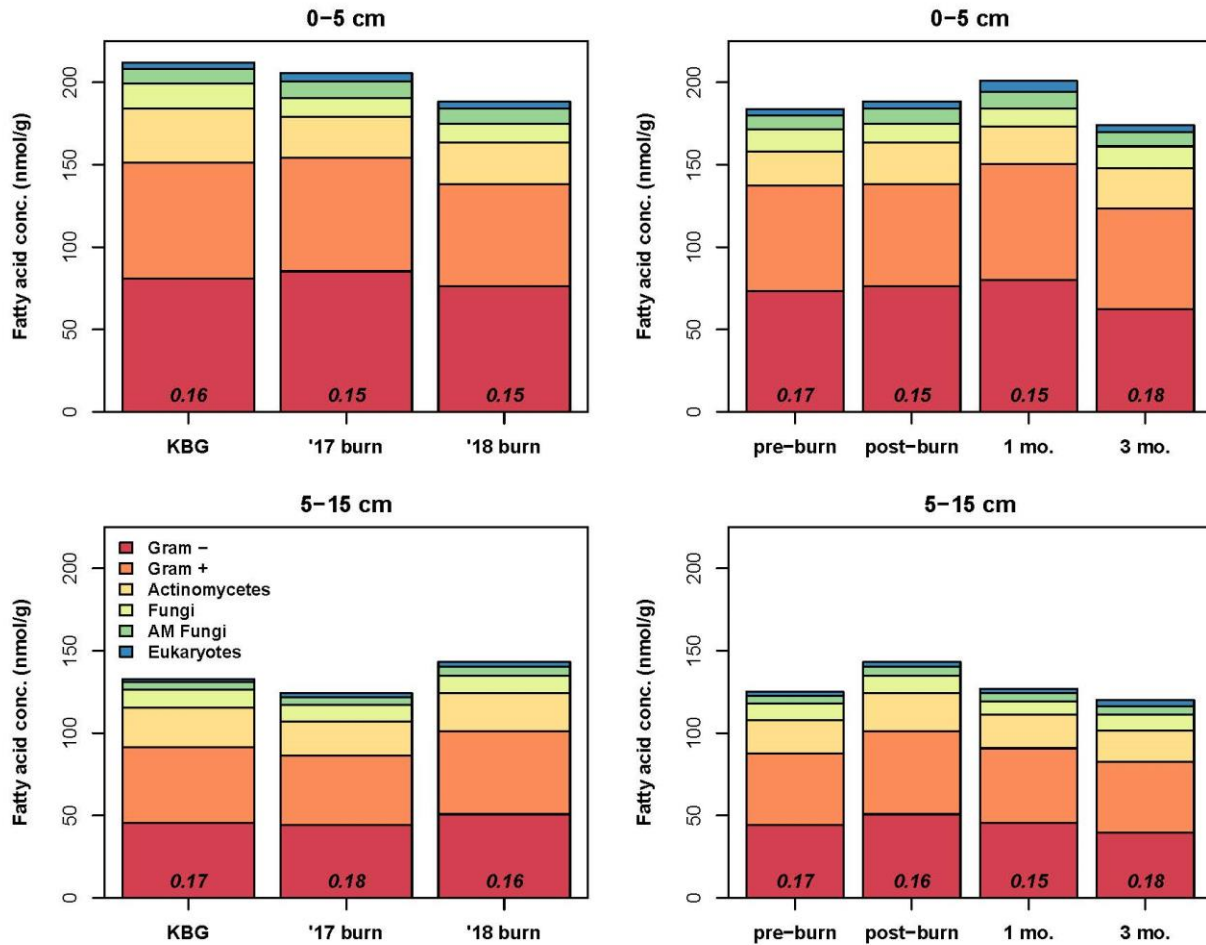


Figure 2.2. Absolute abundance of soil microbial community distribution, and Fungal:Bacterial ratios, across treatments dominated by invasive Kentucky bluegrass, and sites managed with patch-burn grazing in 2017 and 2018. Distribution of soil microbial communities by broad taxonomic groups in terms of absolute abundance (nmol/g) at depths of 0-5-cm and 5-15-cm. Graphs on the left represent values from plots dominated by Kentucky bluegrass (KBG, $n=4$), and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Graphs on the right represent mean values measured immediately before (pre-burn) and after (post-burn) the prescribed burn performed in the spring of 2018, and at increasing time steps throughout the 2018 season (1 mo. and 3 mo.). Microbial groups analyzed include Gram-negative and Gram-positive bacteria (Gram - and Gram +), actinomycetes, fungi, arbuscular mycorrhizal fungi (AM Fungi), and eukaryotes (protists). The numbers at the base of the bars represent Fungal:Bacterial ratios. Statistical differences across treatments, within an individual depth, were assessed at $p \leq 0.05$.

Although we did not observe significant changes among microbial populations, litterbag incubation results showed that decomposition of bluegrass litter varied across treatments, within

depth (Fig. 2.3). Significantly less decomposition occurred in the '18 burn treatment at 5-cm than in the KBG treatment at the same depth. However, no statistically significant differences were found across treatments at the surface, or at 15-cm. Overall, decomposition was fairly high in all study plots, ranging from 25-43% at the surface, to 34-83% at 5-cm, and to 38-82% at 15-cm. Litterbags installed within the thatch layer of the KBG treatment lost on average between 27-58%, however a *t*-test comparison to decomposition occurring at the surface of those same plots revealed no significant differences.

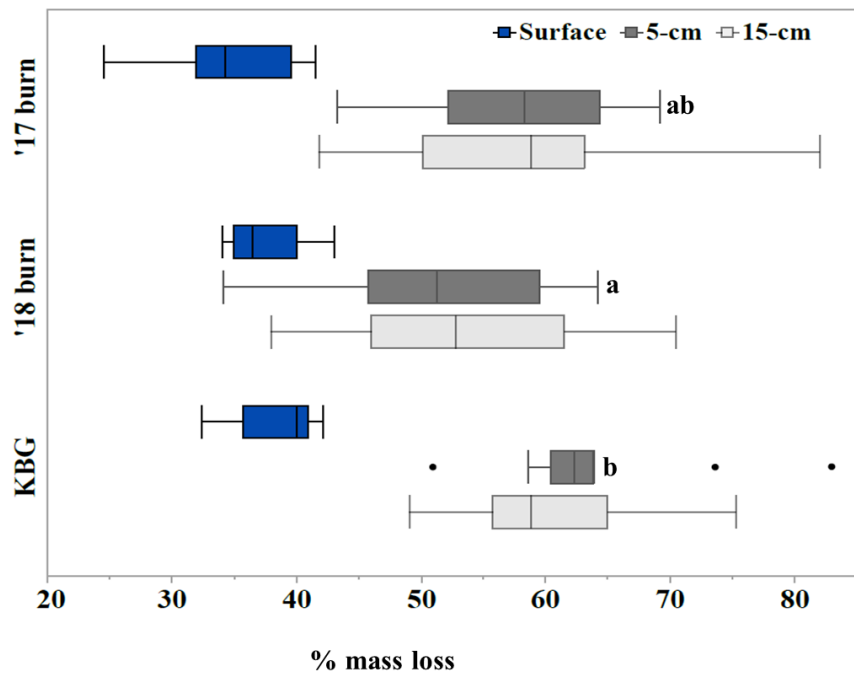


Figure 2.3. Decomposition of Kentucky bluegrass litter by depth. Average decomposition (calculated by litter mass loss (%)) following field incubations) of bluegrass litter at the soil surface ($n=3$ per plot), 5-cm ($n=3$ per plot), and 15-cm ($n=3$ per plot) across plots dominated by Kentucky bluegrass (KBG, $n=4$), and plots managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Decomposition was measured over a year long incubation period from 2018 to 2019 at the North Dakota State University Central Grasslands Research Extension Center. Letters (a, b) represent statistically significant treatment differences among means, within depth ($p \leq 0.05$).

Discussion

Overall, below-ground pools of carbon, nitrogen, and microbial communities proved to be fairly resilient to changes in management practices, and the composition of expressed vegetative cover and surface litters. The majority of carbon and nitrogen fractions, as well as measures of microbial groups, abundance, and activity, showed no significant differences in soils across bluegrass dominated idle-management areas, or under burned and grazed plots with greater native plant species diversity. There were no significant changes in response to management across time steps throughout the season either. The data did not support the original hypothesis that shifts in vegetation patterns, namely the change in quantity of bluegrass and associated litter production resulting from burning and grazing, would alter soil chemical and biological properties. These results are surprising considering that exotic plant invasions, as well as fire and grazing, are known to have significant impacts on soil properties (Ehrenfeld 2003; Zhao et al. 2017; Alcañiz et al. 2018).

We did find soil POX C to significantly differ across treatments, with lower values in managed areas immediately following the application of a prescribed burn than in plots given a year to recover after the application of a burn, and in plots with no active management dominated by bluegrass. This suggests that the prescribed burns in this experiment could have caused the decline in POX C. These results are in contrast to many studies assessing the response of total soil carbon to prescribed burns, which generally find increased values following fire (Úbeda et al. 2005; Brye 2006; Scharenbroch et al. 2012).

The measurement of POX C does differ from total carbon in the sense that it is a highly active, labile form of carbon. Our results are in line with those of a study conducted by Muqaddas et al. (2015), which found decreasing values of soil POX C as the frequency of burns

increased. This form of carbon likely decreases following fire due to the above-ground removal of vegetation, and thus the decreased carbon inputs below-ground from root exudates. However, POX C measured in our study a year after prescribed burning had the highest mean values, suggesting that any decline in this pool as a result of fire is temporary, and prescribed fire may actually increase this pool with time as plant communities above-ground recover. This deviation between low and high levels of labile carbon can act as a source of heterogeneity in the ecosystem, in contrast to more steady and homogenous nutrient pools below idle management areas.

While not statistically significant, bluegrass dominated plots had overall lower carbon and nitrogen values at 0-5-cm for nearly all measured pools than plots managed with fire and grazing one year after burning. The only exception to this trend was greater NH_4 values in bluegrass dominated areas. Since this trend was most pronounced in shallow soils, where bluegrass litter and the effects of above-ground management such as fire and grazing would have the greatest influence, this suggests that over time the removal of bluegrass and its associated thatch through fire and grazing may increase carbon and nitrogen levels in the soil, with the exception of NH_4 , which may decrease.

Further testing would be required to determine if changes in NH_4 concentrations were the result of bluegrass stimulating this pool, or fire and grazing depleting it. Often, fire is thought of as a disturbance that reduces soil nitrogen through volatilization, however, several studies suggest otherwise. Kennard and Gholz (2001) and Gundale et al. (2005) found extractable ions of NH_4 to increase following applications of prescribed burns, which both studies attributed to post-fire mineralization. Higher levels of NH_4 in bluegrass dominated areas compared to burned areas suggest that litter processing rates may be greater in bluegrass conditions, although it is

hard to capture this dynamic process just by measuring the abundance of NH_4 , as this pool is very small, and changes rapidly over short time scales. If bluegrass monocultures are causing increased levels of NH_4 , this would promote a synergistic positive feedback cycle. However, since these values were not statistically significant in our study, this narrative is mainly speculative.

Correlation analysis between variables revealed many of the measured carbon pools were strongly tied with total nitrogen. This relationship is logical considering nitrogen is often the limiting nutrient for plant growth and microbial processes, which take up carbon and nitrogen together at a relatively consistent ratio. Less strong correlations were observed between the volumetric water content of the soil and both NH_4 and PMN. Soil water may be altered as a result of bluegrass dominance and the accumulation of thatch on top of the soil surface (Toledo et al. 2014), which could in turn affect these correlated labile nitrogen pools. However, if soil water in this study was affected by the bluegrass monocultures, or management with fire and grazing, our results suggest the treatment effects were not significant enough to result in differences in either of these nutrient pools.

Although there were few changes in the measured soil nutrient pools, and no changes observed in microbial populations, we did find decomposition to be significantly lower at 5-cm in soils that had been exposed to a prescribed burn within the same year than in soils that were dominated by bluegrass monocultures. A combination of factors contribute to decomposition, including abiotic conditions of the physical soil environment, microbial communities, and substrate availability. Differences in the combination of factors that drive decomposition were great enough between these two treatments to result in significant differences in the breakdown of litter. Since the composition and availability of litter was constant in the litterbag incubations,

and no changes were observed in microbial communities across treatments, it is likely the soil abiotic conditions in the burned and grazed plots are driving differences in decomposition observed at 5-cm.

Warmer and dryer soil conditions at shallow depths, characteristic of recently burned areas, may have played a role in the decreased decomposition we observed. Typically higher temperatures correspond to higher rates of decomposition, provided enough moisture is also available (Sierra et al. 2017). The link between temperature and moisture in relation to decomposition suggests since temperature was not limiting in our study, reduced soil moisture (averaging $0.18 \text{ m}^3/\text{m}^3$ in recently burned plots compared to $0.30 \text{ m}^3/\text{m}^3$ in bluegrass dominated plots at 5-cm throughout the 2018 growing season) was likely the factor hindering decomposition in recently burned areas.

In contrast to sites that had been burned the same year as the study, no differences in decomposition were observed between soils dominated by bluegrass monocultures and areas burned the year previously, suggesting decomposing conditions rebound relatively quickly from fire. The decomposition study revealed another interesting trend, which was decomposition of bluegrass litter was fairly high across all study areas, regardless of treatment. This was surprising considering bluegrass is known to accumulate litter and develop a thick thatch layer, and decreased levels of decomposition are often implicated as the cause. The high levels of decomposition in our study are consistent with findings from Hendrickson et al. (2001), which showed monocultures of improved grass cultivars had significantly greater rates of decomposition than native grass species occurring in undisturbed rangeland. Additionally chemical analysis of standing and fallen bluegrass biomass from our study revealed the litter had an average C:N ratio of 29:1, which would not be in a range prohibitive to decomposition. The

results suggest decreased decomposition is not the reason for the characteristic accumulation of bluegrass litter, but the cause is more likely due to disproportionately high levels of bluegrass biomass production.

The observed differences in decomposition suggest soil processes in this study may be affected by vegetative characteristics, yet our analysis of nutrient and biochemical pools found abundances to remain fairly stable across treatments. While the total abundance of soil nutrient and biochemical pools may not change significantly, it is possible the processes driving the turnover rate of these pools is different. This idea is supported by findings from Wedin and Tilman (1990), and Wedin and Pastor (1993), which showed monocultures of perennial grasses could affect annual net N mineralization up to 10-fold after three years (Wedin and Tilman 1990), yet monocultures of the same species did not change total soil C or N after four years (Wedin and Pastor 1993). Sanderson et al. (2017) found bluegrass also changed the isotopic composition of soil C compared to native prairie species. Future studies on the subject should focus on quantifying the cycling rates, rather than the sizes of these pools.

Overall, our study found neither the difference between bluegrass and native plant composition, nor grazing, had a significant impact on the soil properties studied in this research. The prescribed burn may have caused a brief decline of labile POX C, as well as decomposition in shallow soil, but both properties rebounded by one year after burning. Soil properties measured in this study repeatedly over time did not reveal any seasonal shifts within the same season following the application of prescribed burns. Future research should characterize long-term soil response to management with fire and grazing, as the patch-burn grazing system in this study was newly implemented.

Results from this study revealed below-ground properties in the study area were strikingly uniform, suggesting significant resilience to shifts in above-ground characteristics. While marginal differences were noted between areas recently burned and grazed, and areas dominated by bluegrass under idle management, these differences were short-lived. The plasticity of the measured soil conditions in response to management, and tendency to return to stable conditions, suggests one of two possibilities: 1) prior to bluegrass invasion the soil conditions in this area were already similar to conditions measured in this study, and have not shifted significantly in response to changes in above-ground plant assemblages, or 2) the soil properties studied in this experiment may have reached a threshold point where they are no longer responsive to management, and they have been fundamentally changed as a result of bluegrass. Under either of these circumstances, we have hope management strategies for a more desired plant community are not going to be challenged or delayed because of any differences in the soil.

Ultimately, the consequences of bluegrass invasions on below-ground properties remains largely unknown, and is a topic that requires additional research. However, we do know bluegrass leads to the widespread loss of above-ground biodiversity, and without active and adaptive land management, bluegrass will continue to spread across the northern Great Plains, and beyond, potentially impacting ecosystem services detrimentally. The use of patch-burn grazing as a management practice to reduce bluegrass and promote native plant diversity appears to be a promising strategy, as this study found no long-term negative impacts on the studied soil properties in areas under this management regime.

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GENERAL CONCLUSION

Kentucky bluegrass (*Poa pratensis* L.; hereafter referred to as bluegrass) has rapidly invaded the northern Great Plains over the past three decades (DeKeyser et al. 2009), with detrimental effects on native plant diversity, and ecosystem services. There is evidence that the reintroduction of fire and natural grazing patterns may be a practical option for reducing bluegrass and restoring biodiversity to grassland ecosystems in the northern Great Plains (Dornbusch et al. 2018). While the above-ground impact of bluegrass is prominent, a knowledge gap still exists regarding the below-ground impacts on soil properties (Toledo et al. 2014). Similarly, despite extensive research on the effects of fire and grazing on soil properties, there is very little information published on the interaction of these two management practices in bluegrass dominated systems, and even less information on the potential effects on soil properties resulting from these conditions.

At the North Dakota State University Central Grasslands Research Center near Streeter, ND, we conducted a research experiment to examine below-ground characteristics under bluegrass monocultures, and in response to management with fire and grazing. We collected data measuring soil microclimate, hydrology, biochemical pools, microbial abundance, and decomposition. Overall, findings from our study indicate that bluegrass dominance and the presence of thatch significantly impacts soil microclimate, resulting in cooler and wetter soil conditions throughout the growing season. These shifts in soil temperature and moisture did not appear to have extending effects on to other soil properties, such as biochemical pools, or microbial abundance. In contrasting burned and grazed systems, we found that within the same season as the application of a prescribed burn, there was a decline of labile soil carbon, as well as

decomposition in shallow soil, but both properties had rebounded when measured one year after burning.

As bluegrass prevalence in the northern Great Plains continues to increase, the need for land management techniques that both control bluegrass and promote diversity of native plant species also increases. Understanding soil response to both bluegrass invasions, and land management practices, will be crucial in developing functional rangeland systems, capable of providing valuable ecosystem services. Our observations indicate that bluegrass dominance results in a novel soil microclimate, deviating from growing conditions that favor native plant species, while the combination of fire and grazing has minimal impact on the soil, and is effective at removing thatch and reducing bluegrass. Our findings support that patch-burn grazing is a practical management strategy to reduce bluegrass cover and restore biodiversity to grassland ecosystems.

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**APPENDIX. SOIL MICROBIAL COMMUNITY TAXANOMIC GROUPS AND
FUNGAL:BACTERIAL RATIOS**

A. 0-5-cm							
	KBG	'17 burn	'18 burn	pre-burn	post-burn	1 mo.	3 mo.
AMF	9.1 ± 1.1	10.2 ± 0.8	9.2 ± 1.0	8.7 ± 1.2	9.2 ± 1.0	10 ± 1.5	8.8 ± 1.6
Euk	3.7 ± 0.3	4.9 ± 1.5	4.2 ± 0.6	3.7 ± 0.4	4.2 ± 0.6	6.4 ± 3.2	4.2 ± 0.9
Fungi	14.8 ± 5.1	11.6 ± 3.5	11.4 ± 5.0	13.2 ± 3.9	11.4 ± 5.0	11.1 ± 2.7	13.3 ± 8.5
Actino	32.9 ± 4.3	24.7 ± 11.0	25.2 ± 4.3	20.8 ± 10.6	25.2 ± 4.3	22.7 ± 8.9	24.3 ± 6.5
F:B	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0
Bac	151.2 ± 13.5	154 ± 19.9	138.2 ± 13.7	137.2 ± 13.1	138.2 ± 13.7	150.3 ± 16.9	123.4 ± 28.8

B. 5-15-cm							
	KBG	'17 burn	'18 burn	pre-burn	post-burn	1 mo.	3 mo.
AMF	4.6 ± 0.7	4.7 ± 0.7	5.4 ± 1.1	4.7 ± 1.0	5.4 ± 1.1	5.0 ± 0.7	4.9 ± 0.6
Euk	1.8 ± 0.2	2.4 ± 0.4	2.8 ± 0.3	2.3 ± 0.3	2.8 ± 0.3	2.6 ± 0.6	3.8 ± 2.7
Fungi	10.6 ± 2.4	10.1 ± 1.5	10.3 ± 3.2	9.9 ± 2.3	10.3 ± 3.2	7.8 ± 3.0	9.6 ± 3.9
Actino	24.2 ± 3.4	20.5 ± 3.0	23.3 ± 3.4	20.3 ± 2.5	23.3 ± 3.4	20.5 ± 2.9	19 ± 3.5
F:B	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0	0.1 ± 0.0
Bac	91.3 ± 8.2	86.2 ± 10.6	100.9 ± 14.6	87.6 ± 4.3	100.9 ± 14.6	90.7 ± 8.5	82.5 ± 9.5

Abundance of soil microbial communities by broad taxonomic groups in terms of mean values (nmol/g), followed by standard deviations at depths of 0-5-cm (A) and 5-15-cm (B). Treatments include Kentucky bluegrass monocultures (KBG, $n=4$), and areas managed under a patch-burn grazing regime with a prescribed burn performed in the spring of 2017 ('17 burn, $n=4$), and in the spring of 2018 ('18 burn, $n=4$). Mean values were also measured immediately before (pre-burn) and after (post-burn) the prescribed burn performed in the spring of 2018, and at increasing time steps throughout the 2018 season (1 mo. and 3 mo.). Microbial taxonomic groups analyzed include Gram-negative and Gram-positive bacteria (bac), actinomycetes (actino), fungi, arbuscular mycorrhizal fungi (AMF), eukaryotes (euk), and fungal:bacterial ratios (F:B). Statistical differences across treatments, within an individual depth, were assessed at $p \leq 0.05$.