Vegetation response to fall wildfire in the mixedgrass prairie of western Canada

by

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

Master of Science

in

Rangeland and Wildlife Resources

Department of Agricultural, Food and Nutritional Science

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ABSTRACT

Post-fire rangeland management is typically a period of rest on the Great Plains of North America, but recent research has questioned if long periods of rest are necessary. This study was designed to test different intervals of post-fire rest from defoliation during the first growing season after wildfire. Further, this study monitored the length of time needed and variables affecting vegetation recovery on ungrazed native rangeland in the mixedgrass prairie of Alberta and Saskatchewan, Canada. While post-fire plant production and subsequent June defoliations did not differ from the non-clipped control, July defoliation negatively influenced plant production on burned and non-burned subplots. However, these differences may be explained by the high intensity of defoliation (2 cm) imposing more stress on plants during July defoliation due to a greater relative amount of vegetation removed. Although post-fire June defoliation did not have negative impacts to plant production, there was very little vegetation present at this point in the growing season. Further, litter mass was reduced by wildfire and all defoliation treatments. Percent crude protein increased on burned areas in June of the first post-fire growing season, but this difference disappeared by July. When monitoring vegetation on grasslands in the absence of grazing, by the third post-fire growing season total plant production on burned subplots was reduced by 25% compared to non-burned and litter remained 65% lower on burned subplots. I found a positive relationship between total plant production and litter mass, which indicates that litter is important for plant growth. Importantly, I found a positive relationship between total plant production and pre-burned range health scores, which may indicate that pastures that are managed to maintain or improve rangeland health will improve the rate of post-fire plant recovery. Overall, this research has added to the current body of knowledge on grassland wildfires and it should help better inform post-fire grazing management practices.

ACKNOWLEDGEMENTS

The Government of Alberta, the Government of Saskatchewan, the Saskatchewan Cattlemen's Association, and Environment and Climate Change Canada funded this research. I would like to thank all sources of funding for giving me the opportunity to do my graduate studies on this important topic.

I would like to thank my committee members, Cameron Carlyle, Ph.D., Eric Lamb, Ph.D., and Edward Bork, Ph.D., for helping me through this process. I would especially like to thank my supervisor, Cameron Carlyle, Ph.D., for providing support throughout my degree and enabling me to finish my thesis.

I would like to thank all the ranchers who took time out of their schedules to meet and work with me following this devastating event to their livelihoods and allowing me to do fieldwork on the lands they manage. I would also like to thank all my field assistants, Anne Van Loon, Megan Johns, and Jenna Hlewka, from the University of Alberta, as well as Jacey Bell, Marcus Comfort, Oskar Wolf, Scott Barron, Sera Grover, Laura MacMilan, and Ella Hydomako, from the University of Saskatchewan.

Finally, I would like to thank my family and friends for supporting me through this long and arduous process. Thank you to my parents, brothers, friends, and colleagues. Most of all I would thank my girlfriend and best friend, Rosheen.

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Chapter 1: Fire and Grazing on the Great Plains of North America

On October 17th, 2017, two devastating wildfires spread across the mixedgrass prairie near Acadia Valley and Hilda, Alberta, Canada. Both fires started in Alberta, before crossing the provincial border into Saskatchewan. Dry fall conditions with wind gusting up to 120 km/h contributed to these fires spreading rapidly. Combined, the fires burned approximately 28,000 hectares, including at least 8,100 hectares of grassland. The fires had immediate short-term impacts on cow-calf operations, which included the death of approximately 780 animals, as well as the loss of stockpiled livestock forage for the ensuing winter.

Other potential effects on the ecosystem could include reduced moisture retention, increased soil erosion, changes in plant species composition, the introduction of non-native plant species, and a long-term reduction in plant productivity (Wright and Bailey 1982; Vermiere and Russell 2018). Effects could be long lasting depending on how quickly the ecosystem recovers. Recovery depends on a variety of factors including the environmental conditions at time of the fire (Bailey and Anderson 1980), the weather in years following the fire (Bork et al. 2002), and management decisions regarding how soon and how intensively cattle graze affected areas (Augustine et al. 2010). For decades, the recommended post-fire rangeland management strategy has been to follow a precautionary principle of providing burned grasslands with extended rest to allow sufficient time for rangeland recovery; however, recent research has suggested this may not be necessary. Few studies have examined post-fire grazing management in the mixedgrass region of Alberta and Saskatchewan, and questions remain regarding how long burned pastures should be rested, how to assess post-fire impacts in order to determine stocking rate, and for how long forage quality is affected. This thesis will address these questions by studying the response of vegetation to these wildfires.

1.1 History of Fire on the Great Plains of North America

Grasslands of the Great Plains of North America evolved with disturbances including drought, fire, and grazing (Ford and McPherson 1996; Anderson 2006). Low rainfall and years of drought maintain grassland by restricting shrub and tree encroachment (Ford and McPherson 1996). Common wild herbivores included bison, elk, pronghorn, deer, and moose. Across the Great Plains, lightning-caused fires were a common occurrence and fires had the ability to travel over large areas (Gleason 1913; Gleason 1922; Haley 1929; Wright and Bailey 1982). For example, in the fall of 1885, a fire in Kansas travelled 282 km (Haley 1929). In 1910, a fire in Nebraska travelled 200 km in one day (Rannie 2001) and in 1893 and 1894, fires in southern Saskatchewan burned 1.3M and 1.55M ha, respectively (Rannie 2001). Depending on location and topography in the Great Plains, the historical fire frequency ranged from five to thirty years (Wright and Bailey 1982). Newly burned grasslands attract bison and other ungulates because fire creates fresh plant growth with higher palatability and quality (Ford and McPherson 1996; Rannie 2001; Dufek et al. 2014; Roos et al. 2018). This created an environment where grass fires would be followed by immigration of ungulates to graze new plant growth. Vegetation in the Great Plains of North America is adapted to a disturbance regime that involves fire closely followed by grazing as well as intermittent droughts (Fuhlendorf et al. 2009).

While fire is natural on the Great Plains, humans have had an impact on fire frequency. There is evidence that Indigenous peoples played a role in fire frequency (Sauer 1950; Borchert 1950; Wright and Bailey 1982; Rannie 2001; Roos et al. 2018). Indigenous people deliberately set fires for various reasons including warfare tactics, insect control, and bison hunting (Nelson and England 1971; Rannie 2001). Indigenous people recognized that bison and other ungulates travel to recently burned grasslands and used this knowledge to facilitate their hunting by starting fires to control animal movement (Rannie 2001). After the near elimination of bison from the Great Plains at the end of the 19th century, there were fewer fires because Indigenous people were not starting them for hunting. Conversion of grasslands to agriculture also reduced large-scale wildfire spread. Where wildfire continued to pose a risk, fire suppression policies were introduced (Rannie 2001). As a result, fire frequency and spatial extent have markedly decreased across the Great Plains since European settlement (Wright and Bailey 1982; Rannie 2001).

1.2 Fire Research

Early explorers and natural historians observed how the extent of grasslands was decreasing from encroachment of trees and shrubs due to a reduction in fire frequency (Cook 1908; Gleason 1913; Gleason 1922; Rannie 2001). In the early 20th century, scientists understood that the maintenance of grasslands depended on drought, grazing, and fire to collectively limit woody plant growth and forest expansion (Transeau 1935; Albertson and Weaver 1945). Despite benefits of fire on grassland ecosystems, fear and suppression of fire persisted throughout much of the Great Plains in the early 20th century (Green 1931; Heyward 1936; Heyward 1937; Heyward 1939). Research on post-fire effects helped to shape this perception into best management practices. Early research found that grassland production declined following fire and that the extent of reduction was dependant on burn seasonality. From a study in the mixedgrass prairie of southern Canada, spring prescribed burns led to a reduction in plant production by approximately 50% that year, and another 15% decline the following year, while fall prescribed burns reduced production by 30% the following year (Clarke et al. 1947). Further, if grazing was continued on burned areas they recommended three to five years had to lapse following fire for productivity to become comparable to non-burned (Clarke et al. 1947). In a Kansas study conducted in the short grass prairie, similar results were found, with a 75% reduction in productivity from a spring burn, and a 30% reduction with fall burning (Hopkins et al. 1948). From the same study, burning of a tall grass prairie habitat led to a productivity reduction of 50% with spring burning, and a 35% decline with fall burning (Hopkins et al. 1948). They noted that burn severity was greater in areas that had historically low stocking rates with more litter accumulation and recommended more than two years of rest for recovery (Hopkins et al. 1948). The variation in fire severity results from differences in fuel load (litter) among sites, which influences fire intensity and vegetation to a greater extent (Erichsen-Arychuk et al. 2002). Both studies indicated that more than two years were required for productivity to recover in mixedgrass prairie, short grass prairie, and tall grass prairie. A potential mechanism that caused reduced productivity is the loss of the litter layer that leads to increased soil temperatures and evaporation (Clarke et al. 1947; Hopkins et al. 1948). These studies also found a difference in productivity reduction between spring and fall fires, which could be explained by greater damage to plants during active spring growth, rather than when plants have gone through senescence in fall and may have been more resistant to damage. These findings indicated that a period of rest following fire was required for the recovery in plant productivity, which in turn guided the development of best management practices.

Major increases in research on grassland response to fire began in the 1960s across the Great Plains, especially on forage production following burn events (Anderson 2006). Findings showed similar results to early research, which confirmed that forage production and litter took up to three years to recover to pre-burn conditions (Dix 1960; Launchbaugh 1964; Heirman and

Wright 1973; Wright 1974; DeJong and MacDonald 1975; Wright and Bailey 1982). Similar results were found in short grass prairie, mixed grass prairie, dry mixed grass prairie, and fescue prairie (Figure 1.1). There was also evidence that some plant species were more palatable and productive following fire (Heirman and Wright 1973; Wright 1974). Some scientists began recommending a return to the natural disturbance regime through use of prescribed fire because grasslands are inherently resilient and resistant to fire (Wright and Bailey 1982). Following a disturbance event, resilience is the ability to recover, while resistance is an ecosystem's ability to maintain components (Tilman and Downing 1994). Although plant productivity decreases following fire, it is resilient and has the ability to recover. Wheatgrass species are typically considered more resistant to fire because they have less flammable litter material concentrated near meristematic tissues, at plant bases, and have the ability to regrow from rhizomes protected below the soil surface (Wright and Bailey 1982). Further, there was evidence that higher than average post-fire precipitation resulted in faster recovery of plant production (Dwyer and Pieper 1967; Trlica and Schuster 1969; Heirman and Wright 1973; Wright 1974; Bork et al. 2002). Another positive benefit of fire was that it suppressed woody vegetation that was susceptible to fire (Wright and Bailey 1982). Although scientists recommended prescribed fires, post-fire research in the 20th century gave scientific support for fire suppression policy and this resulted in recommended and required post-fire rest periods to allow plant productivity to recover.

1.3 Current Management

Currently, the predominant post-fire management recommendation or requirement on arid rangelands is to give a period of rest from grazing to allow plant regrowth and litter accumulation (Bureau of Land Management 2007; Government of Alberta 2018). In Alberta, Canada there is a recommended one-year rest period (Government of Alberta 2018). In other jurisdictions, such as public lands in the United States there is a minimum required rest period of two years (Bureau of Land Management 2007). The use of rest is thought to ensure the long-term maintenance and sustainability of the forage resource because the required length of post-fire recovery is variable in grasslands. Although rest is a common management practice across the Great Plains, this option may also be impractical for ranchers, especially if all their land has burned and no other forage is available, and it is also possible that rest may not be necessary to ensure recovery (Augustine et al. 2010; Roselle et al. 2010; Bates and Davies 2014).

1.4 Recent Fire Research

While more recent research continues to support the notion of providing post-fire rest by affirming that up to 3 years may be necessary for forage productivity to return to pre-burn conditions (Erichsen-Arychuk et al. 2002), there are recent studies suggesting that rest may not be necessary. Pastures that were moderately grazed by sheep following summer fire were not different from non-grazed areas and had recovered by the following growing season (Roselle et al. 2010; Vermiere et al. 2014). Similarly, following summer wildfire in the Nebraska sandhills, an ecosystem that can easily be destabilized, pastures were lightly stocked with bison during drought conditions, yet plant productivity recovered within two growing seasons (Arterburn et al. 2018). Despite livestock grazing, these rangelands recovered within two years or less. This research indicates grazing following fire did not always negatively affect plant production when compared to non-burned areas, but environmental conditions can influence speed of recovery.

Consideration of precipitation following wildfire is integral to understanding the recovery of grassland plant productivity. In an Alberta mixedgrass prairie, an August wildfire reduced plant production in uplands until the third growing season following burning, while lowlands were affected more severely, potentially due to greater fuel load from litter accumulation, and recovered in the fourth growing season (Erichsen-Arychuk et al. 2002). Prolonged post-fire drought was a major factor delaying the recovery in plant production (Erichsen-Arychuk et al. 2002). Conversely, by the second growing season after a spring wildfire in the mixedgrass prairie of North Dakota, burned plant productivity recovered compared to non-burned (Gates et al. 2017). In this study, precipitation was 190% of the long-term average in the first year following fire, and slightly above average (114%) the second year after fire (Gates et al. 2017). This exemplifies how above average precipitation following burn events will reduce recovery time of grassland productivity, which has been found in previous studies (Dwyer and Pieper 1967; Trlica and Schuster 1969; Heirman and Wright 1973; Wright 1974; Bork et al. 2002).

Research throughout the 20th century has indicated that rest is the best post-fire management strategy (Wright and Bailey 1982). As a result, this is the predominant recommendation or requirement on burned grasslands across the Great Plains of North America (Bureau of Land Management 2007; Government of Alberta 2018); however, some recent literature disputes the necessity for long periods of rest following burn events (Vermiere et al. 2014; Roselle et al. 2010; Gates et al. 2017). This inconsistency in the literature demonstrates that more research is needed to better inform land managers and governments on appropriate post-fire rangeland management practices. I conducted two quantitative experiments to examine the recovery of mixedgrass prairie following the October 2017 wildfires that spread across the Alberta/Saskatchewan border. In Chapter 2, I conducted an experiment that manipulated the timing of defoliation in the first year following fire in order to test the effects of different rest periods immediately following fire on subsequent plant recovery. In Chapter 3, I measured plant communities at a series of paired ungrazed, burned/non-burned locations, over three consecutive years, to examine the temporal recovery of plant communities to wildfire. Chapter 4 provides a synthesis of results and a recommendation for post-fire grazing management.

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1.6 Figures

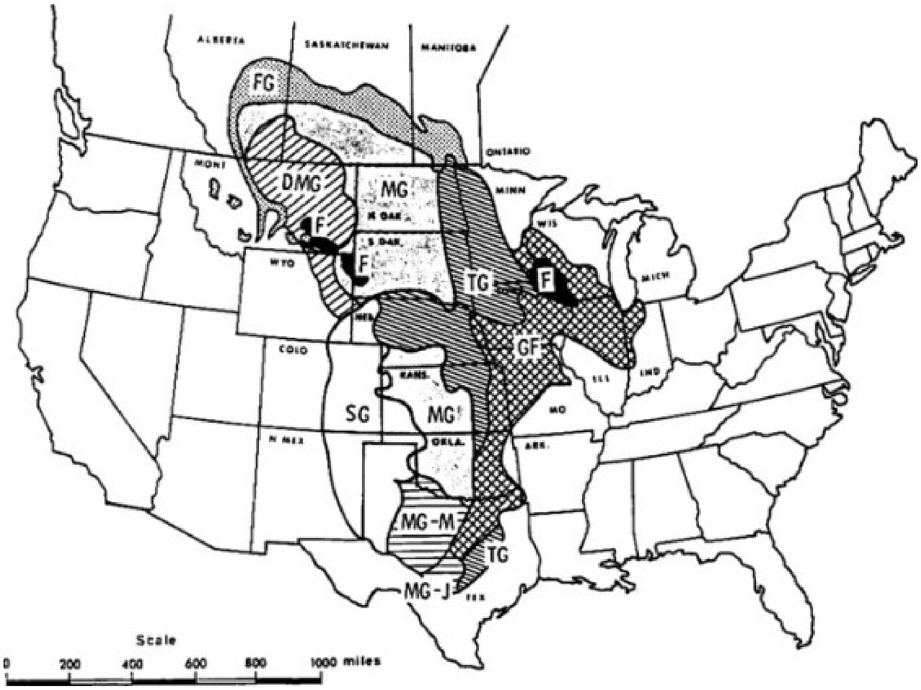

Figure 1.1. Natural Vegetation of the Great Plains grasslands (Wright and Bailey 1982). Modified from Kuchler (1965) and Rowe (1972). DMG – Dry mixed grass; F – Forest; FG – Fescue Grass; GF – Grassland forest; MG – Mixed grass; MG-J – Mixed grass-juniper-oak; MG-M – Mixed grass-mesquite; SG – Short grass; TG – Tall grass.

Chapter 2: Effects of defoliation timing during the first post-fire growing season on forage quantity, forage quality, and litter accumulation in burned and non-burned mixedgrass prairie

2.1 Abstract

The common management recommendation on grasslands following fire is a period of rest from grazing for at least a year or longer. Rest following grassland fire is thought to allow time for vegetative regrowth and litter accumulation. Despite this, there is debate on the required length of rest, if any, for the maintenance of plant production. Using a paired experimental design, we examined the recovery of burned mixedgrass prairie by comparing production in burned and adjacent non-burned mixedgrass prairie (n=13) following different post-fire defoliation treatment times. Treatments were conducted along the perimeter of two wildfires that occurred in October 2017 across the Alberta-Saskatchewan border, Canada. To assess effects of early defoliation following fire, I clipped vegetation at 3 different times in the year following fire (early June, late June, July, with a non-clip control for comparison). In the first growing season, burned subplots had reduced plant production compared to non-burned grassland. By the second growing season, June defoliations did not compound the negative effect of fire on plant production, with no difference between burned and non-burned defoliation treatments. However, July defoliation led to the lowest total plant biomass in the second growing season. Litter quantity was reduced by all defoliation treatments and wildfire, which could be expected to decrease plant production over the long-term. Fire effects on forage quality were evident at the beginning of the first post-fire growing season, but disappeared by the end. Increased forage quality the first growing season following fire may or may not be beneficial depending on rancher management goals.

Information from this study will provide grazing managers with advice to aid in post-fire management decisions.

2.2 Introduction

The Great Plains of North American formed through a disturbance regime of drought, fire, and herbivory (Ford and McPherson 1996; Anderson 2006). While fire is natural, it may remove litter material that supports plant growth, and thereby reduce plant production (Willms et al. 1986). Grasslands provide forage for livestock; therefore, a decline in productivity and associated forage availability can reduce economic gains for ranchers. Litter enables grassland plant communities to retain more moisture, especially in arid regions, because litter reflects solar radiation, which decreases soil temperatures and evaporation (Clarke et al. 1947; Hopkins et al. 1948; DeJong and MacDonald 1975). Litter also reduces wind exposure, which further limits evaporation. To allow time for litter layer accumulation and the recovery of plant growth, a period of rest from grazing can be used (Wright and Bailey 1982). An appropriate length of post-fire rest is considered to be important for vegetation recovery and thereby ensure sustainable management of forage resources.

Even in the absence of grazing, the length of time required for plant productivity to recover in grasslands following wildfire is variable and can take a number of years. After spring wildfire on short grass prairie in Kansas, plant productivity did not recover until the third growing season (Launchbaugh 1964). Similarly, an August wildfire on a mixedgrass prairie in Alberta reduced plant production in uplands until the third growing season following burning, while lowlands took longer, eventually recovering by the fourth growing season (Erichsen-Arychuk et al. 2002); these results highlight spatial variation in fire effects on plant production and recovery across rangeland landscapes. Timing of fire has also been shown to have variable impacts on vegetation. When fires occur in spring during active plant growth there is more damage to vegetation compared to if fires occur in fall when plants have senesced (Clarke et al. 1947; Hopkins et al. 1948). Further, studies have shown that if post-fire precipitation is higher than average, the time required for grassland productivity to approach those of non-burned levels is reduced (Dwyer and Pieper 1967; Trlica and Schuster 1969; Heirman and Wright 1973; Wright 1974; Bork et al. 2002). Vegetative recovery may take a number of years, but environmental variables and post-fire grazing management may influence the rate of this recovery.

Post-fire grazing may prolong recovery periods or further reduce plant productivity below reductions induced by fire alone; therefore, the fire and defoliation disturbance interaction is generally avoided in rangeland management. This has resulted in recommendations for extended rest from livestock grazing following wildfire. For example, in Alberta, Canada, provincial rangeland managers recommend a one-year rest period (Government of Alberta 2018), while on some public lands in the United States, there is a required minimum rest period of two years (Bureau of Land Management 2007). A study in an Alberta mixedgrass prairie on fire effects suggested one to three years is needed for plant productivity to recover in the absence of grazing; however, if cattle grazing continued, then three to five years may be required (Clarke et al. 1947). Further, spring, summer, and fall defoliation have all been shown to decrease plant productivity following grassland burning, so questions remain about the proper timing of post-fire grazing in the mixedgrass prairie necessary to ensure recovery (Bunting et al. 1998; Bogen et al. 2003). There are a number of possible mechanisms underlying prolonged recovery of vegetation due to grazing following fire. Cattle and wildlife show a preference for burned grasslands due to improved forage quality arising from increased nitrogen and digestibility (i.e., reduced fiber content) (Willms et al. 1980; Ford and McPherson 1996; Erichsen-Arychuk et al. 2002; Dufek et al. 2014). Increased quality, in turn, can result in overgrazing with preferential use of burned areas. Litter, which helps to retain soil moisture and enhance plant growth (Willms et al. 1986), is removed by burning and accumulates more slowly when grazed (Bates et al. 2009; Vermiere et al. 2014). Consequently, post-fire rest has been recommended in North America through much of the 20th century due to concerns of reduced plant production, delayed rangeland recovery from post-fire grazing, and prolonged decreases in economic gains.

In contrast to the general recommendations or requirements for rest, a few studies have suggested that only one year of rest from grazing is needed, if at all, (Roselle et al. 2010; Vermiere et al. 2014; Gates et al. 2017) because grazing did not extend recovery times or further reduce plant productivity. In other words, defoliation impacts did not compound the negative effects of fire on plant production. After an early spring wildfire with spring, summer, and fall defoliation times in the mixedgrass prairie of North Dakota, the subsequent growing season showed that the spring defoliation had 22 % higher current year plant production compared to control, summer, and fall defoliations, which led to a recommendation that rest was not necessary in this area (Gates et al. 2017). In a similar experiment, following an Idaho spring prescribed fire on a forest understory composed of *Festuca idahoensis* and *Agropyron spicatum*, defoliation in early summer resulted in a reduction in plant productivity and an increase in plant mortality; therefore, one year of rest was recommended (Bunting et al. 1998). Similar results have also been found with summer fire in the sagebrush-steppe and in the mixedgrass prairie followed by moderate sheep grazing where plant production recovered the year following fire (Roselle et al. 2010; Vermiere et al. 2014). Consequently, there are a range of possible rest periods that may be appropriate following fire in order to maintain plant productivity.

In light of the varied responses of grassland vegetation to post-fire grazing, and consequent divergence in post-fire management recommendations, I evaluated the effects of timing of defoliation at various points during the first growing season following a wildfire occurring the previous fall in the mixedgrass prairie. The objective of this study was to assess effects of varied timing of defoliation, ranging from rest (no-clip control), spring, and summer treatments, on mixedgrass prairie following a fall wildfire. Responses examined were forage quantity, forage quality, and litter accumulation. I hypothesized that 1) plant production will be most negatively impacted by the burn and early June defoliation interaction because plants are in active growth and stressed following a disturbance event, 2) forage quality will be greater on burned subplots than non-burned, specifically in June, because all plant material has been removed by fire, and 3) litter mass will be more negatively impacted by the wildfire and early June defoliation interaction because lower plant biomass will be available to contribute to its accumulation.

2.3 Methods

2.3.1 Study Sites

On October 17, 2017, two separate wildfires burned through a region of the mixedgrass prairie in southeastern Alberta and southwestern Saskatchewan, Canada. Dry conditions prior to the fires and wind speeds of up to 120 km/h on the day of the fires contributed to rapid spread (Government of Alberta, 2020). Combined, the fires burned approximately 28,000 hectares, including at least 8,100 hectares of pastureland. The study sites are in the Dry Mixedgrass Natural Subregion of Alberta and Dry Mixed Grassland ecoregion of Saskatchewan (Natural Regions Committee 2006; Thorpe 2014). Despite different classification systems between the provinces, the study region is continuous. Long-term average annual precipitation from the three closest Alberta Agriculture and Forestry weather stations is 317.3 mm (Government of Alberta 2020). The dominant vegetation of the region is composed of drought tolerant species such as needle-and-thread (*Hesperostipa comata* (Trin. & Rupr.) Barkworth), blue grama (*Bouteloua gracilis* (Kunth) Lag. ex Griffiths), Junegrass (*Koeleria macrantha* (Ledeb.) Schult.), western wheatgrass (*Pascopyrum smithii* (Rydb.) Barkworth & D.R. Dewey), and northern wheatgrass (*Elymus lanceolatus* (Scribn. & Smith) Gould). Soils in the region are predominately Orthic Brown Chernozems with some Solonetzic, Vertisolic, and Regosolic soil types (Natural Regions Committee 2006; Thorpe 2014). Historically, grazing practices differed between individual pastures, with some grazed rotationally (spring, summer, or fall) and others grazed continuously through the growing season. Stocking rates during the growing season ranged from 0.24 to 0.95 AUM/ha, while dormant season grazing ranged from 1.74 to 2.53 AUM/ha.

Precipitation data were collected from the three nearest Alberta Agriculture and Forestry weather stations (Acadia Valley, Schuler, and Social Plains) to study sites in order to describe weather conditions during the monitoring period (Figure 2.6). Long-term average precipitation data accumulated from 1961-2020 was used to describe the sites (Government of Alberta 2020).

2.3.2 Experimental Design

The wildfires provided an opportunity to evaluate burned and non-burned subplots immediately adjacent to each other on either side of the burn perimeter (Appendix A). Sites were selected by walking along burn edges to find locations where the same ecological site was present on both sides of the fire perimeter and where the wildfire pathing did not appear to have been impacted by unusual natural land features, such as hills or wetlands. It was expected that fire intensity would be higher along the fire periphery due to slower wildfire movement speed. In total, 13 sites in eight different pastures were selected to create a paired burned/non-burned sampling design. Throughout the duration of the study, sampled areas were excluded from grazing using either fences or range cages (~1 m^2 in size).

2.3.3 Plant Biomass

To test the effect of different times of defoliation the first growing season following wildfire, I conducted a small plot experiment that controlled the timing of clipping in 2018, the first growing season following the wildfires. At each site, four 1 m^2 quadrats were established in each of the burned and non-burned subplots and randomly assigned to one of four clipping treatments: early June defoliation (June 1st-3rd), late June defoliation (June 26th-28th), July defoliation (July 10th-20th), and a non-clipped control. These quadrats were hand-clipped to 2 cm above the soil surface at their respective defoliation times. In 2019, the second full growing season following wildfire, all quadrats were clipped by hand to 2 cm above the soil surface at estimated peak biomass, in July (10th-20th), to assess the vegetative response of grassland to the defoliation treatments imposed in 2018. The central 50 x 50 cm (0.25 m^2) area of the quadrat was used for data collection, in 2018 and 2019, which created a 25 cm treated buffer to reduce edge effects. Clipped vegetation was collected and sorted into morphological groups: grasses, forbs, and shrubs, and weighed to provide a measure of biomass removal. Shrubs were uncommon and were not included in analysis. Vegetation clipped from the buffer area was also removed from the quadrats. In 2018, litter was not collected, while in 2019, litter was hand raked from the 50 x 50 cm quadrat before clipping by hand. All vegetation samples were dried at 55 °C until the loss in weight was no longer observed, and then weighed. Further, forb to grass ratios were assessed in 2018 and 2019 to characterize impacts of these morphological groups in relation to each other following wildfire and defoliation time. Forb to grass ratio was calculated by dividing forb biomass by grass biomass. Clipping by hand was used as a substitute for livestock grazing, however this defoliation treatment (down to 2 cm) would be similar to a heavy defoliation event (ex. High intensity-low frequency).

2.3.4 Forage Quality Analysis

Following drying, a Wiley Mill was used to grind vegetation samples before nitrogen content analysis. All samples were analyzed for nitrogen content using a FOSS NIR (FOSS 2020) and a subset of samples were analyzed using a Thermo Scientific Flash 1112 analyzer for model development (Thermo Scientific 2007). Further, the calibration curve was developed using my samples as well as samples previously collected in Saskatchewan (Appendix B, unpublished data). Percent crude protein was used as an indicator of forage quality and was calculated by multiplying nitrogen content by 6.25. Forbs and grasses were analyzed separately, however, there were very few shrub samples and they were not analyzed. Crude protein yield was calculated by multiplying forb or grass percent crude protein by grass or forb biomass found within burned or non-burned plots. Total crude protein yield was calculated as the sum of grass and forb available protein.

2.3.5 Statistical Analysis

All statistical analyses were conducted in R (R Core Team 2018) using the *1me4* package and *lmer* function (Bates et al. 2015). A split plot design with defoliation time nested within burn/non-burn subplots and burn treatment nested within site was used to examine effects of wildfire on forage quantity, forage quality, and litter quantity. For all analyses burn/non-burn and defoliation time were included as fixed effects with their interaction, and site was a random effect. In 2018, for total, grass and forb biomass data, and forb to grass ratio, I expected biomass to increase through the growing season due to advancing plant phenology; thus, Tukey's honestly significant difference (HSD) test was used to compare burn treatment within individual defoliation times. Forage quality in 2018 was assessed using Tukey's HSD test for all burn and defoliation time interactions because I wanted to determine differences throughout the first growing season in relation to both fixed effects. In 2019, total, grass and forb biomass, forb to grass ratio, litter mass, and forage quality, were analyzed using Tukey's HSD test for all burn and defoliation time interactions in order to assess effects of the burn treatments that occurred in 2017 and the defoliation treatments applied in 2018.

Response variables were transformed when necessary to conform to the assumption of normality of the residuals and homogeneity of variances. For 2018, grass, forb, and total biomass were log (x+1) transformed, while the forb to grass ratio was log (2x+1) transformed. For 2019, grass and total biomass were log (x+1) transformed, while forb biomass, litter mass, and the forb to grass ratio were log (2x+1) transformed. Grass and forb percent crude protein in 2018 and 2019 did not need to be transformed. Total, grass, and forb crude protein yields in 2018 and 2019 were log (x+1) transformed. Significance for all ANOVA and Tukey's HSD tests were assessed at alpha = 0.05.

2.4 Results

2.4.1 Plant Production

In 2018, the first growing season following wildfire, total and grass biomass were affected by burn, defoliation time, and burn*defoliation time, while forb biomass was only affected by defoliation time (Table 2.1). Total and grass biomass increased throughout the growing season (Figure 2.1). The total biomass in early June burned subplots was 64% lower compared to non-burned subplots, but by July, burned subplots were only 27% lower than non-burned (Figure 2.1). Forb to grass ratio in 2018 was affected by burn, but not defoliation time or burn*defoliation time (Table 2.1). Forb to grass ratios were different between burned and non-burned areas in early June and late June defoliation, but did not differ between burn treatments during the July defoliation (Figure 2.1).

In 2019, the second growing season following wildfire, burned and non-burned subplots did not differ for total, grass, or forb biomass; however, the timing of defoliation in 2018 did affect subsequent biomass (Table 2.2, Figure 2.2). Total biomass was lowest following the July 2018 defoliation treatment, but remained equal among the non-clipped control, early June, and late June defoliations (Figure 2.2). Grass biomass was also lowest following the July defoliation treatment, greatest in the control, and the two June defoliations were intermediary (Figure 2.2). Forb biomass did not differ as a result of treatment timing (Figure 2.2). In 2019, there was no effect on the forb to grass ratio from burn or defoliation time (Table 2.2, Figure 2.2). For litter mass in 2019, there was an effect from burn, defoliation time, and burn*defoliation time (Table 2.2). In the absence of clipping (non-clipped control), litter on burned subplots were 71% lower than non-burned, while all defoliation treatments reduced litter with and without fire (Figure 2.3). Additionally, the later defoliation treatments reduced litter to a greater extent, to the point that litter was similar in quadrats defoliated in late June or July, regardless of fire exposure.

2.4.2 Percent Crude Protein and Crude Protein Yield

In 2018, grass percent crude protein was affected by burn, defoliation time, and burn*defoliation time (Table 2.3). Forb percent crude protein was only affected by defoliation time and burn* defoliation time (Table 2.3). Grass percent crude protein was higher on burned subplots than non-burned when defoliated in early and late June, while forb percent crude protein was only greater on burned subplots when defoliated in early June (Figure 2.4). By July 2018, percent crude protein did not differ between burned and non-burned subplots for either grasses or forbs. In 2018, total and grass crude protein yield was affected by burn and defoliation time, but not burn*defoliation time (Table 2.3). Forb crude protein yield was only affected by time (Table 2.3). Total, grass, and forb crude protein yields were lowest in burned subplots defoliated in early June, but by late June, there was no difference between burned and non-burned biomass (Figure 2.5).

In 2019, grass and forb percent crude protein were not affected by burn or defoliation treatments (Table 2.3). Total and grass crude protein yields in 2019 showed no difference between burned and non-burned subplots, but timing of defoliation had an effect with the July defoliation having the lowest available protein in both cases (Table 2.3). Forb crude protein yield did not differ across burn or defoliation time treatments (Table 2.3). Total, grass, and forb crude protein yields follow patterns based on their relative biomass abundance similar to Figure 2.2.

2.4.3 Precipitation

I summarized precipitation patterns during the growing season to interpret biomass responses. During the growing season before the October 17, 2017 wildfires, accumulated growing season precipitation (April to July) was comparable to long-term average precipitation by the end of July (Figure 2.6). In the few months leading up to the wildfires, there was an extended drought in the study area, with only 59% of long-term average precipitation in August, 51% in September, and 87% from October 1st to October 16th. In 2018, accumulated growing season precipitation was approximately the historical average with 95% of long-term average precipitation. Finally, 2019 only received 85% of long-term average precipitation; however, the greatest decline occurred in July with precipitation approximating long-term average from April to June (Government of Alberta 2020).

2.5 Discussion

My results indicate that defoliation in the first growing season following wildfire did not reduce plant biomass the subsequent year in the mixedgrass prairie. Similar results were found in a North Dakota mixedgrass prairie, where a spring wildfire did not interact with post-fire defoliation in the first growing season after burning (Gates et al. 2017). In fact, my experiment showed that by the second growing season after fire there was no difference in plant biomass between burned and non-burned subplots. Other studies have also found that only one year of rest is necessary for plant production to recover following rangeland burning and defoliation (Bunting et al. 1998; Gates et al. 2017). Typically, low precipitation delays the recovery of plant production on burned areas (Erichsen-Arychuk et al. 2002). However, during the two years following wildfire in my experiment, accumulated growing season precipitation approximated the historical average in the study area and there was no difference in plant biomass between burned and non-burned subplots. The Great Plains of North America evolved with fire, drought, and grazing, therefore these grasslands may have a high degree of resilience to these disturbances (Fuhlendorf et al. 2009). These results demonstrate that mixedgrass prairie plant productivity may be resilient to a combination of concurrent disturbances and that, in some cases, long periods of post-fire rest from grazing may not be necessary for biomass growth.

The July 2018 defoliation led to less plant biomass in the following year in both burned and non-burned subplots compared to the non-clipped control and both June defoliations, suggesting that defoliation during peak growth can reduce plant vigour in the mixedgrass prairie. This is concerning because the July defoliation was applied in the middle of the month, which is at the start of the recommended grazing period for native grasses in this region (Bailey et al. 2010; Adams et al. 2016). Avoidance of early season grazing is recommended because native plants grow slowly, but retain quality later into the growing season (Bailey et al. 2010; Adams et al. 2016). However, these defoliation effects must be assessed by considering the relative amount of stress imposed on plants from this intense defoliation event (2 cm) and from the fixed defoliation height at various times through the growing season. Plants defoliated in June had less biomass removed and may have been less affected relative to those defoliated in July, at peak vegetative biomass (Bogen et al. 2003). Previous studies (Bogen et al. 2003) found an interaction between fire and defoliation timing, but this was not found in my experiment. The lack of interaction in my experiment could be because defoliation had such a large effect. Future experimental work should assess the effects of fixed relative defoliation heights (i.e. 50% removal) and include spring, summer, and fall defoliation times to determine differences in plant stress. Although June defoliations did not negatively impact burned areas in the following year, I would not recommend grazing at this time because of low forage availability throughout the month of June.

Above ground plant biomass was lower in burned subplots the year after the fire, but was not different from non-burned subplots in the second year suggesting resilience to fall fire. The relatively quick recovery of biomass production may result from the fact that this was a fall fire. Following prescribed fires in an Alberta mixedgrass prairie, plant production was reduced from a spring fire by approximately 50% that year and 15% the following year, while a fall burn reduced production by 30% the following year (Clarke et al. 1947). Similar results were found in a Kansas short grass prairie, where there was a 75% reduction in productivity with spring prescribed burn and 30% reduction with fall prescribed burn (Hopkins et al. 1948). This difference may be the result of perennial plants already being senesced and dormant in the fall with their energy reserves below ground (Clarke et al. 1947; Hopkins et al. 1948). However, in my experiment during 2018, there was a higher forb to grass ratio on burned subplots compared to non-burned, which indicates forbs increased following burning. Although there was no significant difference between burn or defoliation treatments, the average forb to grass ratio in 2019 was also higher in early June and late June on burned subplots, which would have affected total biomass. Fast recovery in plant biomass between burned and non-burned subplots may be explained by fall wildfire, however increased forb biomass must also be considered, as this is may not be a favorable forage replacement depending on species.

Of note, clipping by hand is not a direct substitute for livestock grazing and my results could have been different if livestock had been grazing the study locations instead of clipping by hand. Many factors can contribute to impacts on vegetation such as selectivity of animals, intensity and season of grazing, as well as frequency of defoliation. A comparison between livestock grazing and mowing, which is similar to clipping by hand, was conducted in a recent meta-analysis and they concluded that, while there are some differences, the effect sizes are small (Tälle et al. 2016). In my experiment, the clipping by hand treatment would most closely resemble a high intensity-low frequency grazing system.

Litter quantity was reduced by all defoliation treatments and the wildfire, which may affect long-term plant productivity. At a location near to our study sites in the mixedgrass prairie, three years of litter removal led to a 57% reduction in plant production due to increased soil temperature and evaporation (Willms et al. 1986). This highlights the importance of the litter layer for rangeland plant production, especially in arid ecosystems. By the second growing season in my experiment, there was no reduction in total biomass between burned and non-burned in the control, early June, or late June, but this could be because low rainfall in the year following defoliation eliminated any litter benefits. Further, a reduction in litter could result in increased soil erosion from livestock grazing and trampling (Naeth et al. 1991). While plant biomass was not different between treatments, litter was reduced and this could have long-term consequences on plant production or other ecosystem functions, especially in drought conditions when water may be limiting.

In my experiment, wildfire effects on forage quality were noticeable at the beginning of the first post-fire growing season, but disappeared by the end. Similarly, in a Montana study it was reported that the crude protein of *Aristida purpurea* increased following summer and fall wildfires when compared to non-burned areas, but were not different by the end of the first growing season (Dufek et al. 2014). Other studies have reported increased plant percent crude protein on burned areas in the first year following prescribed burning (Willms et al. 1980; Bork et al. 2002; Augustine et al. 2010). Despite increased percent crude protein on burned subplots in the first year of my experiment, there was a reduction in total crude protein yield on burned subplots compared to non-burned due to a reduction in overall plant biomass. Higher forage quality helped to partially compensate for reduce forage availability, which may be beneficial because increased forage quality leads to increased weigh gains by livestock (Smith and Owensby 1972). However, burned subplots that have low plant biomass and high forage quality is probably not sufficient for grazing. Further, the negative impacts of defoliation on the landscape will outweigh any short-term positive effects of livestock weight gains. My results show that total available protein was always lower in burned subplots, but by late June in the first post-fire growing season there was no difference between burned and non-burned subplots.

Livestock utilization has also been shown to be greater on burned areas the first year after fire with no difference between burned and non-burned the following year (Erichsen-Aryhcuk 2002). This has important implication for ranchers who graze partially burned pastures in the first year after fire because there may be some benefits to livestock from grazing these areas, but livestock may overuse the burned portion of the pasture.

2.6 Summary

Studies showing reduced plant production following fire has resulted in conservative post-fire rangeland management practices across the Great Plains of North America (Bureau of Land Management 2007; Alberta Government 2018). My results show that the effects of wildfire on plant production and quality were most apparent in the first year following burning, but disappeared by the second year. There was no difference between burned and non-burned plant production in 2019, however July 2018 clipping showed a larger reduction in plant production than other defoliation treatments. This could be explained by the relatively intense defoliation treatment (2cm) at peak biomass. Early season defoliation following wildfire did not affect grassland biomass production, but there may have been a lack of effect due to low biomass removal. Although this study did not show any impacts of June defoliation following wildfire, the low amount of forage availability at this point of the growing season should be considered. Further, my study did not assess potential impacts that could result from livestock grazing such as trampling and erosion.

My results indicated an increase in forb and grass percent crude protein in early June and late June in the year immediately following wildfire, which contributed to total available protein on burned and non-burned subplots not differing by late June. Whether increased forage quality following wildfire compensates for reduced forage quantity for livestock will likely depend on producer management needs and objectives (Government of Alberta 2018). Regardless, post-fire grazing management will still require care to avoid overgrazing due to livestock preference to graze in burned areas, avoid undue erosion from trampling, and rebuild litter levels to enhance soil moisture and long-term plant productivity. This experiment showed no difference in plant productivity between rested control plots and June clipped plots, which indicates that rest may not be required in the first growing season following burning; however, given the known critical importance of litter, management should allow litter to accumulate to ensure the long-term maintenance and sustainability of the forage resource.

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Tables

Table 2.1. Results of generalised linear model testing effects of 2017 fall wildfire and of defoliation timing treatment (early June, late June, and July) during the 2018 growing season on total biomass, grass biomass, forb biomass, and the ratio of forb to grass biomass. P-values < 0.05 are indicated in bold font.

Variable	***df***	***F***	***p***
		2018	
Total Biomass (g / 0.25 m^2)			
Burn	1	39.8	**<0.001**
Defoliation Time	2	54.0	**<0.001**
Burn*Defoliation Time	2	5.25	**0.0079**
Grass Biomass (g / 0.25 m^2)			
Burn	1	56.8	**<0.001**
Defoliation Time	2	39.2	**<0.001**
Burn*Defoliation Time	2	5.52	**0.0063**
Forb Biomass (g / 0.25 m^2)			
Burn	1	0.60	0.44
Defoliation Time	2	15.1	**<0.001**
Burn*Defoliation Time	2	0.74	0.48
Forb:Grass			
Burn	1	6.91	**0.011**
Defoliation Time	2	3.73	**0.030**
Burn*Defoliation Time	2	0.14	0.87

Table 2.2. Results of generalized linear model testing effects of 2017 fall wildfire and timing of defoliation during the 2018 growing season (non-defoliated control, early June, late June, and July) on total biomass, grass biomass, forb biomass, the ratio of forb to grass biomass, and litter mass collected in July of 2019. P-values < 0.05 are indicated in bold font.

Variable	***df***	***F***	***p***
		2019	
Total Biomass (g / 0.25 m^2)			
Burn	1	0.099	0.75
Defoliation Time	3	10.5	**<0.001**
Burn*Defoliation Time	3	0.32	0.81
Grass Biomass (g / 0.25 m^2)			
Burn	1	0.42	0.52
Defoliation Time	3	8.47	**<0.001**
Burn*Defoliation Time	3	0.51	0.67
Forb Biomass (g / 0.25 m^2)			
Burn	1	0.74	0.39
Defoliation Time	3	2.63	0.056
Burn*Defoliation Time	3	0.31	0.82
Forb:Grass			
Burn	1	1.06	0.31
Defoliation Time	3	0.93	0.43
Burn*Defoliation Time	3	0.19	0.90
Litter Mass (g / 0.25m^2)			
Burn	1	46.2	**<0.001**
Defoliation Time	3	32.8	**<0.001**
Burn*Defoliation Time	3	2.40	0.073

Table 2.3. Results of generalised linear model testing effects of 2017 fall wildfire and timing of defoliation during the 2018 growing season on percent crude protein and crude protein yield in 2018 and 2019. P-values < 0.05 are indicated in bold font.

Variable	*df*	*F*	*p*	*df*	*F*	*p*
	2018			**2019**		
Grass Percent Crude Protein						
Burn	1	21.6	**<0.001**	1	0.72	0.40
Defoliation Time	2	8.33	**<0.001**	3	0.51	0.68
Burn*Defoliation Time	2	6.13	**<0.01**	3	0.45	0.72
Forb Percent Crude Protein						
Burn	1	3.20	0.081	1	0.029	0.87
Defoliation Time	2	6.20	**<0.01**	3	0.23	0.88
Burn*Defoliation Time	2	3.76	**0.032**	3	0.35	0.79
Grass Crude Protein Yield (g / 0.25 m^2)						
Burn	1	23.1	**<0.001**	1	0.12	0.73
Defoliation Time	2	23.8	**<0.001**	3	6.82	**<0.001**
Burn*Defoliation Time	2	2.84	0.066	3	0.84	0.48
Forb Crude Protein Yield (g / 0.25 m^2)						
Burn	1	0.67	0.42	1	0.045	0.83
Defoliation Time	2	20.2	**<0.001**	3	2.39	0.079
Burn*Defoliation Time	2	1.28	0.29	3	0.43	0.73
Total Crude Protein Yield (g / 0.25 m^2)						
Burn	1	18.7	**<0.001**	1	0.65	0.42
Defoliation Time	2	23.7	**<0.001**	3	9.93	**<0.001**
Burn*Defoliation Time	2	1.17	0.32	3	0.38	0.77

Figures

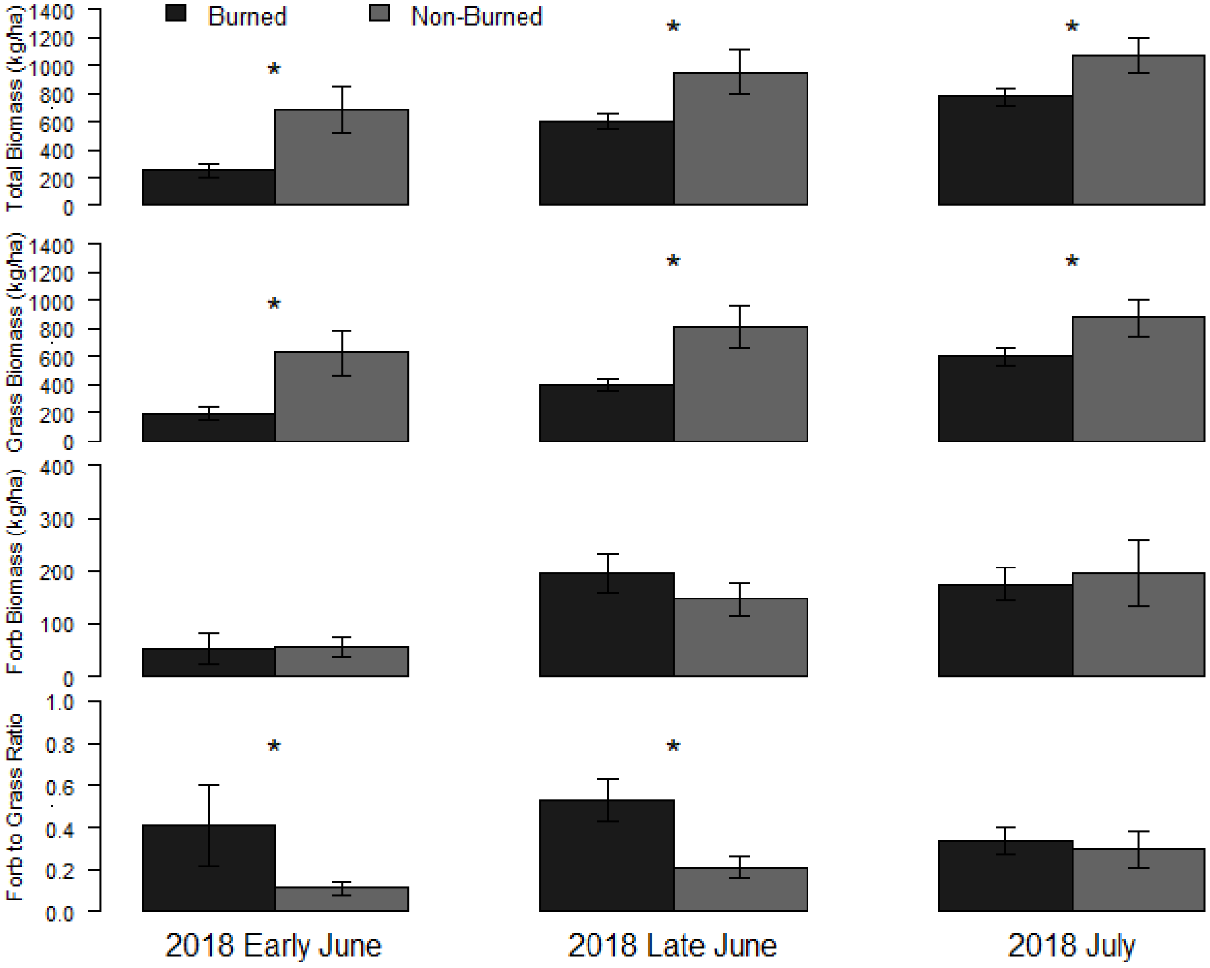

Figure 2.1. Total, grass, and forb biomass in kg/ha (SE) removed by clipping the vegetation to 2 cm at three different times of defoliation in 2018 on burned and non-burned subplots, the first growing season following fall 2017 wildfire. Forb to grass ratios at different initial defoliation treatment times in 2018. A * above bars indicates a significant difference within a given defoliation time (Tukey HSD test, p < 0. 05).

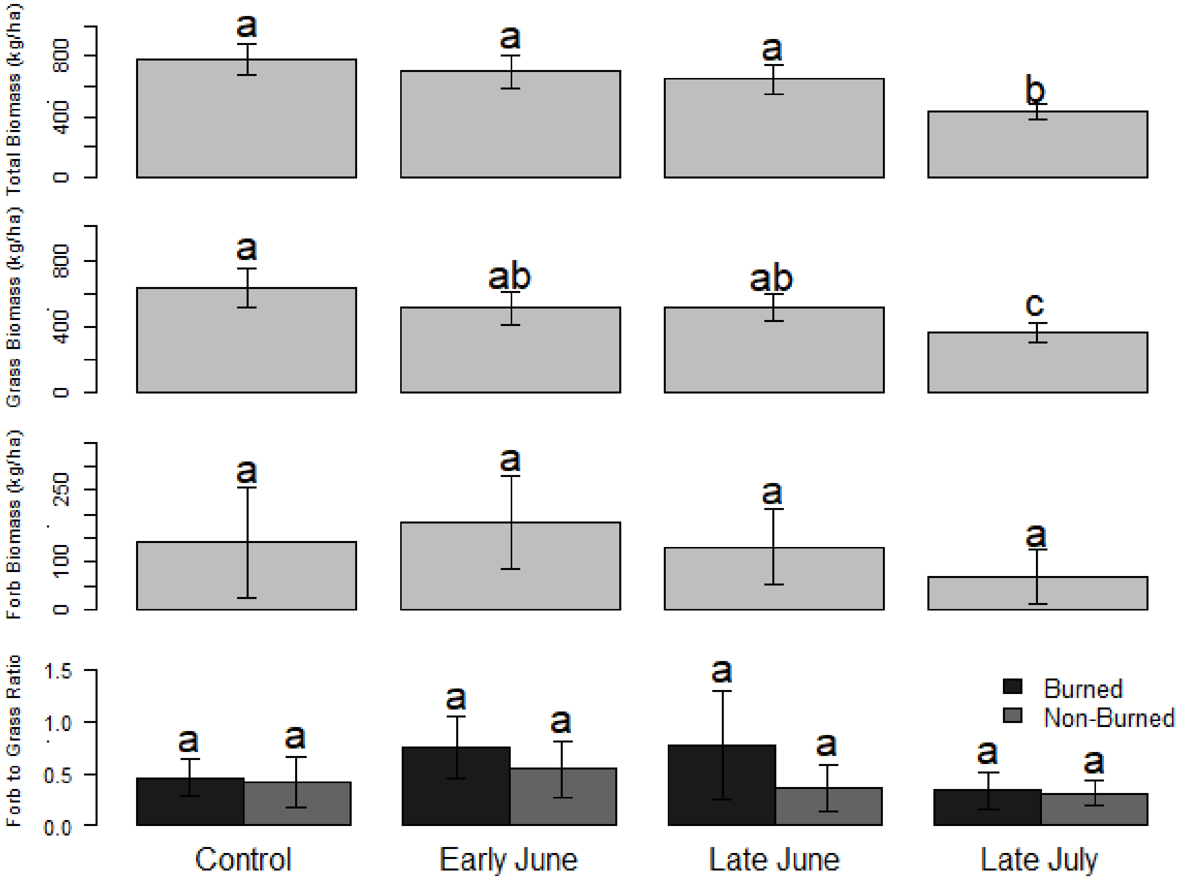

Figure 2.2. Total, grass, and forb biomass in kg/ha (SE) removed in July 2019, following the 2018 defoliation treatment. There was no difference between burned and non-burned subplots for total, grass, or forb biomass (Table 2.2). Forb to grass ratios in 2019 following different defoliation treatments in the previous year. Bars sharing the same letters are not significantly different (Tukey HSD test, $p < 0.05$).

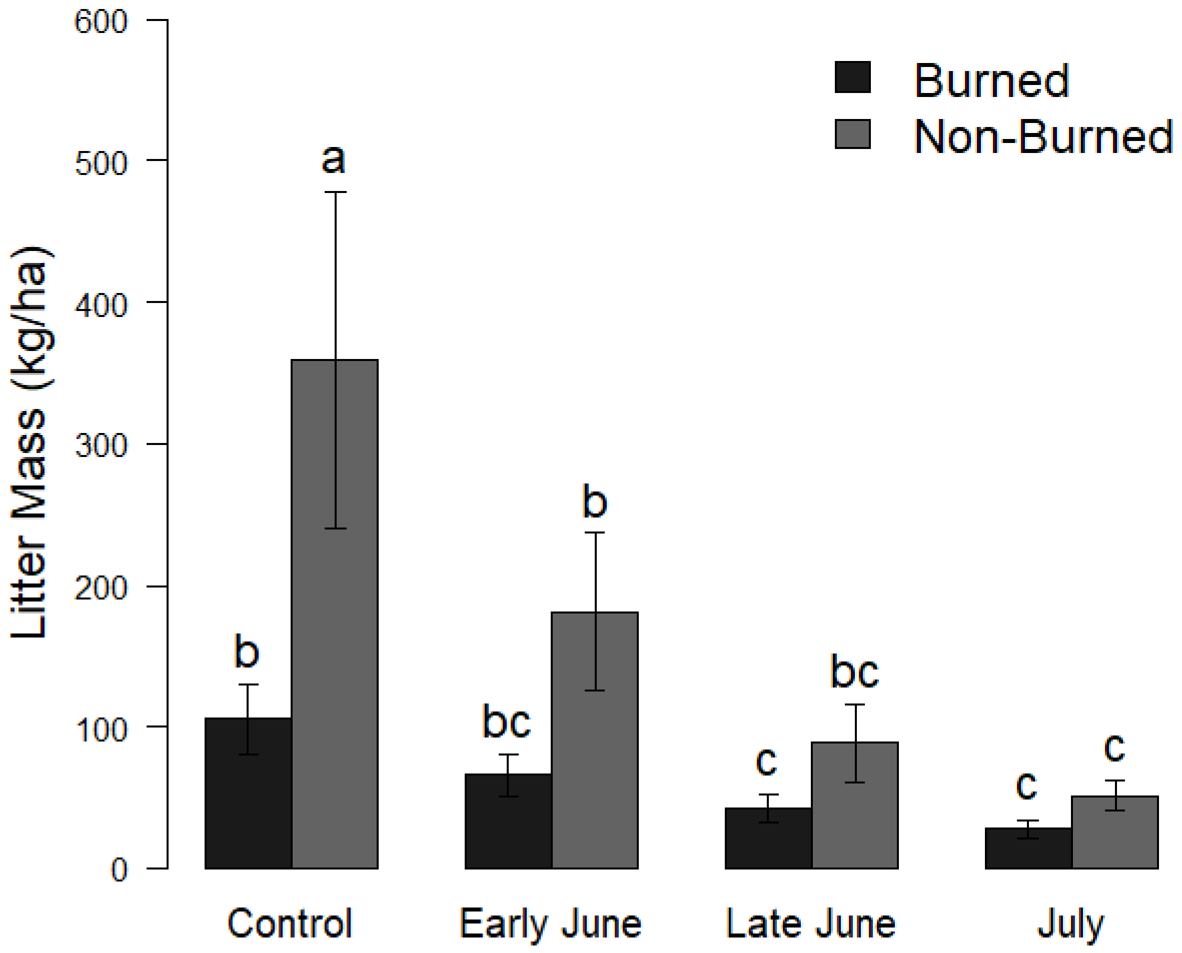

Figure 2.3. Litter mass (kg/ha SE) in July 2019, following the 2018 defoliation treatment. Bars sharing the same letters are not significantly different (Tukey HSD test, $p < 0.05$).

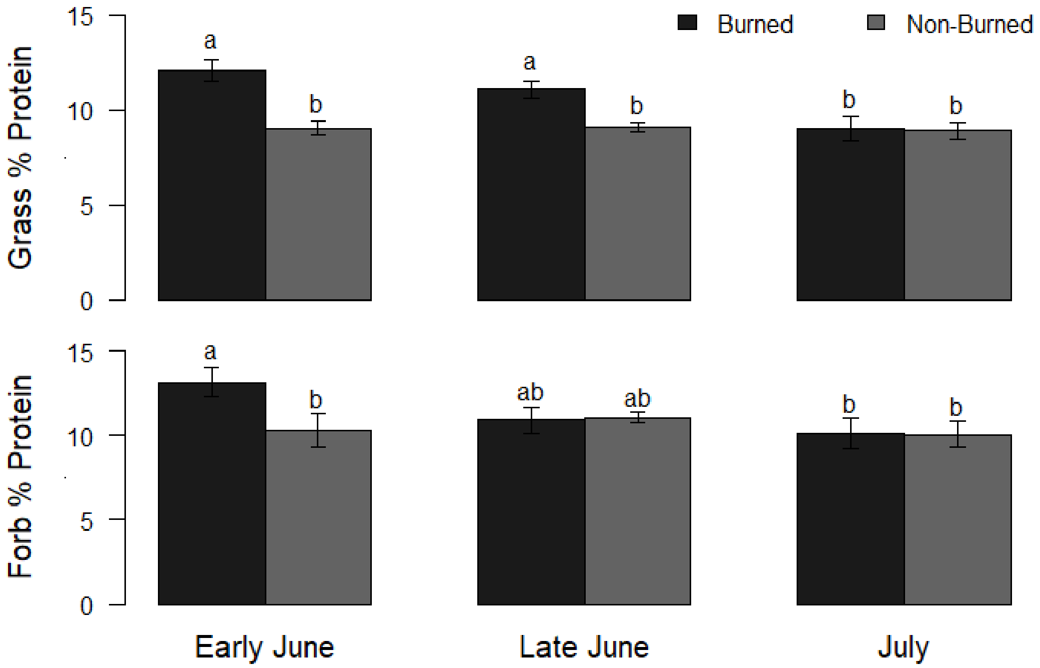

Figure 2.4. Grass and forb percent crude protein (SE) in 2018, the first growing season following wildfire, on burned and non-burned subplots at different defoliation times. Bars sharing the same letters are not significantly different (Tukey HSD test, $p < 0.05$).

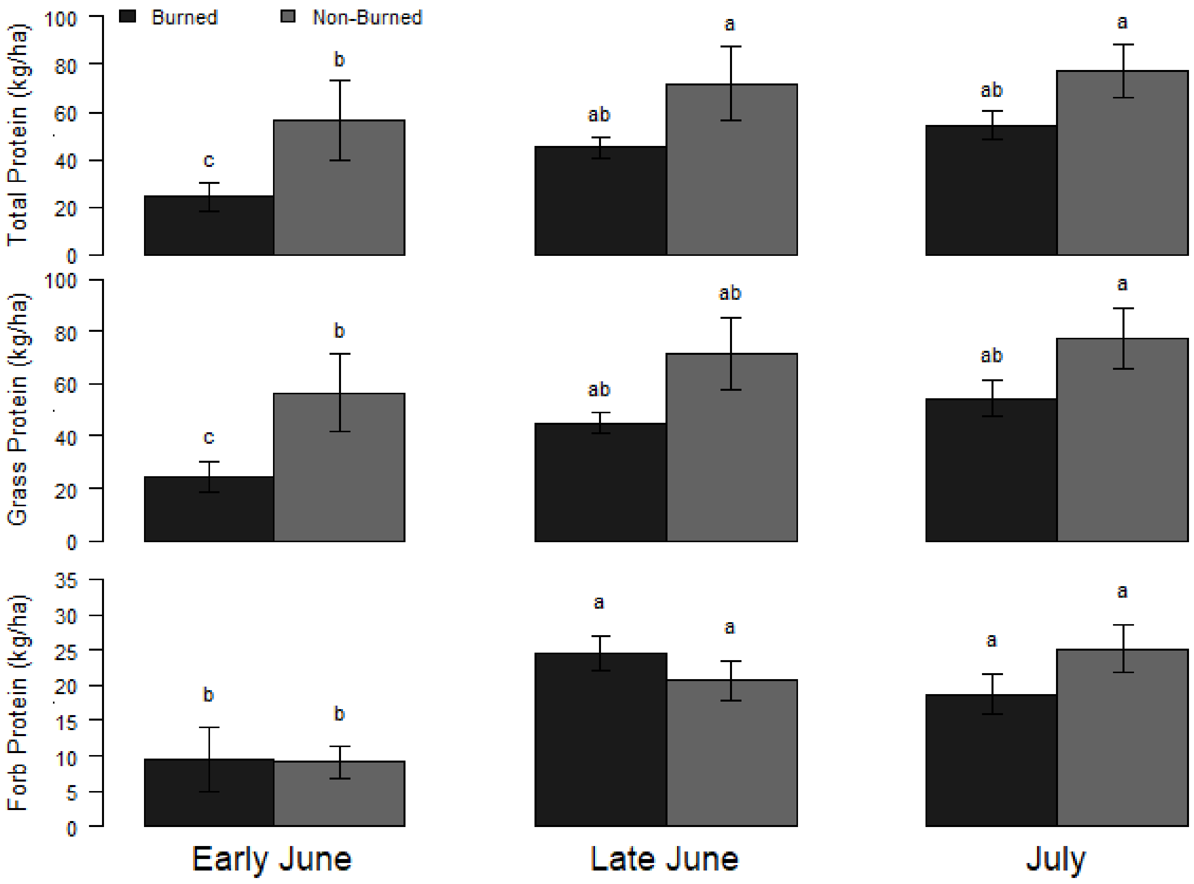

Figure 2.5. Total, grass, and forb crude protein yield (kg/ha SE) in 2018, the first growing season following wildfire, on burned and non-burned subplots at different defoliation times. Bars sharing the same letters are not significantly different (Tukey HSD test, $p < 0.05$).

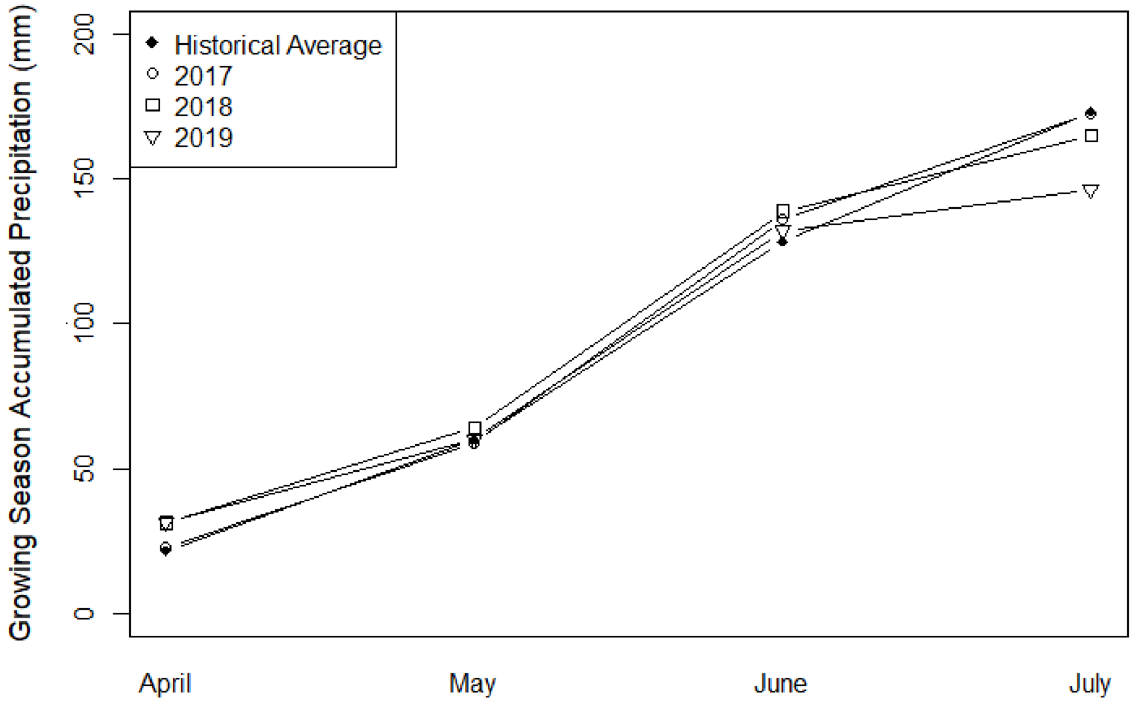

Figure 2.6. Mean accumulated growing season precipitation (April to July) from 2017 to 2019 in comparison to historical average precipitation. Precipitation data are from the three nearest Alberta Agriculture and Forestry weather stations (Acadia Valley, Schuler, and Social Plains) to the October 2017 wildfires (Government of Alberta 2020). Long-term average precipitation data are from 1961-2020.

Chapter 3: Recovery of forage quantity and quality, litter accumulation, and plant species composition following fall wildfire in the mixedgrass prairie

3.1 Abstract

Wildfire can alter grassland ecosystem function and post-fire grassland recovery is critical for sustained livestock forage production. A common management recommendation to ranchers following grassland fire is to give the pasture a period of rest from grazing. Rest allows time for the recovery of plant production through plant regrowth and litter accumulation, but there is debate on the required length of recovery time needed for grasslands following wildfire. Understanding factors that contribute to grassland recovery may explain some of this variation and can aid land managers in decision making following wildfire. Using a paired experimental design, I monitored vegetative recovery of burned subplots and compared them to non-burned subplots (n=25) along the perimeter of two wildfires that occurred in October 2017 across the Alberta-Saskatchewan border within the mixedgrass prairie. To assess recovery, I evaluated forage quantity, forage quality, plant species composition, and rangeland health over three successive growing seasons. Measured at peak biomass in July of 2018, 2019, and 2020 following the wildfire, burned subplots had 54%, 74%, and 75%, each year respectively, of total vegetative biomass compared to non-burned subplots. However, litter on burned subplots was only 35% of non-burned grassland by the third growing season following fire, which given its importance for soil moisture retention, may be contributing to reduced annual biomass production. Importantly, I found that sites with higher pre-burn range health scores produced more biomass in the years following the wildfire. No differences were found in forage quality between burned and non-burned subplots by July in the first growing season following wildfire. Compositional differences between plant communities were initially absent for 2 growing

seasons, but appeared 3 years after burning. As the occurrence of prairie wildfire is increasing, this information will provide grazing managers with information to aid in future post-fire management decisions.

3.2 Introduction

Vegetation on the Great Plains of North America formed under a disturbance regime of drought, fire, and grazing (Ford and McPherson 1998; Anderson 2006); however, wildfire affects livestock production through the destruction of vegetation, which may have lasting effects. The recovery of plant communities following wildfire is critical for continued sustainable livestock production. Typically, a period of rest from grazing is used following grassland fires to allow time for litter layer accumulation and plant production recovery (Wright and Bailey 1982), though the actual recovery of these systems remains inconsistent. A more detailed understanding of the factors that enhance grassland recovery from wildfire can aid managers in making better post-fire management decisions.

Length of post-fire plant production recovery can take several years and is variable due to pasture management and environmental conditions. Although these grasslands are adapted to fire, plant production may be reduced and litter material, which supports plant productivity by conserving soil moisture (Willms et al. 1986), will be lost. Litter facilitates moisture retention in grassland plant communities, especially in arid regions, because litter reflects solar radiation, thereby minimizing soil temperatures and limiting evaporation (Clarke et al. 1947; Hopkins et al. 1948; DeJong and MacDonald 1975). There may be spatial variation in fire effects on plant production and recovery, such as in an Alberta mixedgrass prairie following an August wildfire, where plant production was reduced in uplands until the third growing season following burning, while lowlands did not recover until the fourth growing season (Erichsen-Arychuk et al. 2002). This variation in fire severity was hypothesized to arise from differences in fuel load (litter) among sites, with lowlands having more fuel and therefore greater fire intensities, thereby influencing the vegetation to a greater extent.

Timing of fire has also been shown to have variable impacts to vegetation. On average, there is more vegetative damage when fires occur during active plant growth, such as in spring or early summer, than if fires occur in fall when plants are dormant (Clarke et al. 1947; Hopkins et al. 1948). Additionally, plant species have different responses to fire; for example, wheatgrass species are normally resistant to fire because they are rhizomatous (Wright and Bailey 1982). Environmental conditions surrounding fire also influence the speed of vegetative recovery. If post-fire precipitation is higher than average, plant production may recover faster (Dwyer and Pieper 1967; Trlica and Schuster 1969; Heirman and Wright 1973; Wright 1974; Bork et al. 2002). Further, pre-fire pasture management could affect the rate of recovery because different intensities of grazing influence plant vigour (Adams et al. 2016). Rangeland health is an indicator of plant vigour (Adams et al. 2016) that may be used for pre-fire condition assessment and to determine plant species composition responses to fire. Post-fire grazing may also delay the recovery of plant production and there are a number of explanations for this (Clarke et al. 1947). Fire results in improved forage quality from increased nitrogen content; therefore, livestock and wildlife prefer grazing on burned rangeland, which could increase the risk of overgrazing (Clarke et al. 1947; Willms et al. 1980; Ford and McPherson 1996; Erichsen-Arychuk 2002; Dufek et al. 2014). Another reason for delayed plant production recovery is that grazing results in slower litter accumulation (Bates et al. 2009; Vermiere et al. 2014), which may reduce plant production by lengthening the period of impairment to hydrologic function. Overall, the length of post-fire recovery is often unpredictable due to the many different variables that contribute to rangeland recovery.

Consequently, concerns of reduced plant production, delayed rangeland recovery from post-fire grazing, and prolonged decreases in economic gains have resulted in the recommendation for post-fire rest across the Great Plains of North America for much of the 20th century. In Alberta, Canada, there is currently a recommended one-year rest period for grassland (Government of Alberta 2018), while in other jurisdictions such as public lands in the United States, there is a minimum required rest period of two years (Bureau of Land Management 2007). However, a few studies have suggested that rest is only necessary for a single year, or not at all. In the mixedgrass prairie of North Dakota, following spring fires with subsequent defoliation treatments, plant production was not impacted by vegetation removal after wildfire and thus rest was not recommended in this area (Gates et al. 2017). Results have also been found with summer fires followed by moderate sheep grazing in the sagebrush-steppe and in the mixedgrass prairie where plant production recovered by the second growing season following fire (Roselle et al. 2010; Vermiere et al. 2014). These studies show that, despite the combined effects of grazing and fire, in some cases only one year of rest or less may be required for the recovery of plant production. Thus, understanding the underlying factors creating variation in recovery times following fire could help generate more site-specific recommendations for post-fire grazing management.

Vegetation responses to fire are variable across studies and long-periods of rest may or may not be necessary; therefore, I monitored the effects of a fall (October) wildfire on native mixedgrass prairie pastures in the absence of post-fire grazing for three successive growing seasons to assess the length of time required for recovery. The first objective of this study was to identify the effects of burning on temporal dynamics of forage quantity, forage quality, litter accumulation, and species composition. The second objective was to identify factors that regulate the rate of recovery, specifically litter accumulation and range health. I hypothesized that 1) forage quantity on burned subplots will increase as litter increases, 2) forage quality would be greater on burned than non-burned subplots, 3) species composition will not differ between burned and non-burned subplots, and 4) sites with higher pre-burned Alberta Rangeland Health Assessment scores will have faster recovery on burned subplots.

3.3 Methods

3.3.1 Study Sites

Two wildfires burned through a region of the mixedgrass prairie in southeastern Alberta and southwestern Saskatchewan, Canada on October 17, 2017. Wind gusting up to 120 km/h in combination with dry conditions resulted in fast moving fires (Government of Alberta 2020). Including both fires, approximately 28,000 ha of land was burned, with at least 8,100 hectares of rangelands affected, most of which are subject to cattle grazing. Sites were in the Dry Mixed Grassland ecoregion of Saskatchewan and the Dry Mixedgrass Natural Subregion of Alberta (Natural Regions Committee 2006; Thorpe 2014). Although classification systems are different between the provinces, the study region is continuous. The three nearest Alberta Agriculture and Forestry weather stations show that the yearly long-term average precipitation is 317.3 mm (Government of Alberta 2020). Dominant grass species are composed of drought tolerant species such as needle-and-thread (*Hesperostipa comata* (Trin. & Rupr.) Barkworth), blue grama (*Bouteloua gracilis* (Kunth) Lag. ex Griffiths), Junegrass (*Koeleria macrantha* (Ledeb.) Schult.), western wheatgrass (*Pascopyrum smithii* (Rydb.) Barkworth & D.R. Dewey), and northern wheatgrass (*Elymus lanceolatus* (Scribn. & Smith) Gould). Predominant soils in the region are Orthic Brown Chernozems with other soil types such as Solonetzic, Vertisolic, and Regosolic (Natural Regions Committee 2006; Thorpe 2014). Historically, grazing practices differed between individual pastures, with some grazed rotationally (spring, summer, or fall) and others grazed continuously. Stocking rates during the growing period ranged 0.24 to 0.95 AUM/ha, while dormant season grazing ranged from 1.74 to 2.53 AUM/h (rancher personal communications, June 2018). Following the wildfires only one study site was grazed by cattle in the year immediately following the wildfires, while in the second and third years more sites were grazed but grazing management was not typical of ongoing practices prior to the fires (lower stocking rates).

Precipitation data were collected from the three nearest Alberta Agriculture and Forestry weather stations (Acadia Valley, Schuler, and Social Plains) in order to describe weather conditions during the monitoring period (Government of Alberta 2020). Long-term average precipitation data accumulated from 1961-2020 was used to describe the sites.

3.3.2 Experimental Design

Numerous burns lines were created by the wildfires across the study region with adjacent burned and non-burned areas (Appendix A). A paired burned/non-burned sampling design was established on pastures with native vegetation. Site selection involved walking along burn lines to find locations where natural land features did not appear to influence the wildfire pathing, such as hills or wetlands, and where ecological site was similar on both sides of the fire perimeter. Sampling locations were placed more than 100 m away from one another if they were within the same pasture and there was a maximum of 4 sites in one pasture. From 2018 to 2020, I collected data from 25 sites on 13 different pastures.

3.3.3 Plant Biomass

At each site, four randomly placed quadrats on both burned and non-burned subplots were clipped by hand to a stubble height of 2 cm in July from 2018 to 2020 for measurement of peak biomass. Quadrats were 50 x 50 cm and quadrat layout ensured a different area was clipped every year at each site. Before clipping vegetative biomass by hand, litter material was collected by hand raking quadrats. Vegetation harvested was separated into morphological groups: grasses, forbs, and shrubs. As there were very few shrubs, analysis was only conducted on grasses and forbs. Total biomass was the addition of grass and forb biomass. If a pasture was to be grazed, I randomly placed three 1 m diameter range cages prior to the grazing period, on both burned and non-burned areas, in order to obtain an ungrazed measure of plant production. All vegetation samples were dried at 55 °C until loss in weight was no longer observed and then weighed.

3.3.4 Forage Quality Analysis

Following drying, a Wiley Mill was used to grind vegetation samples before nitrogen content analysis. All samples were analyzed for nitrogen content using a FOSS NIR (FOSS 2020) and a subset of samples were analyzed using a Thermo Scientific Flash 1112 analyzer for model development (Thermo Scientific 2007). Further, the calibration curve was developed using my samples as well as samples previously collected in Saskatchewan (Appendix B, unpublished data). Percent crude protein was used as an indicator of forage quality and was calculated by multiplying nitrogen content by 6.25. Forbs and grasses were analyzed separately, however, there were very few shrub samples and they were not analyzed. Crude protein yield was calculated by multiplying forb or grass percent crude protein by their respective vegetative biomass found within burned or non-burned subplots. Total crude protein yield was calculated as the sum of grass and forb available protein.

3.3.5 Species Composition

To assess potential shifts in plant species composition, permanent plant identification transects were installed adjacent to the yearly biomass sampling locations. Along these transects, percent cover estimates were used to monitored plant species, litter, and bare ground. Transects were 8 m in length with ten 20 x 50 cm quadrats evaluated for percent cover. Starting at the northernmost point of the transect, cover was estimated every 0.5 m. Cattle trails and other well travelled areas were avoided by approximately 10 m to limit the influence of overuse by livestock in sampling locations. Due to difficulties differentiating between *Pascopyrum smithii* and *Elymus lanceolatus*, especially in the first growing season following wildfire (when grasses were small and primarily vegetative), these species were grouped together as *Elymus spp.* for analysis. A list of plant species found at study locations can be found in Appendix B.

Average species composition at individual sites was calculated by dividing plant species cover by quadrat number. The measures of plant species composition used in this experiment were species richness, Shannon's Diversity Index, permutational analysis of variance (PERMANOVA), and NMDS ordination. Species richness is calculated by counting the number of plant species in burned and non-burned subplots. The Shannon's Diversity Index is calculated by the sum of the proportion of species multiplied by the natural log of the proportion of species (Shannon 1948). PERMANOVA and NMDS ordinations were run using R (R Core Team 2018).

3.3.6 Rangeland Health Assessment

During 2019, the second year after fire, rangeland health assessments were conducted at each site on burned and non-burned areas using the Alberta Rangeland Health Assessment (Adams et al. 2016). The Rangeland Health Assessment scores on non-burned areas were assumed to be the same on both sides of the burn perimeter before fire and were used as an indicator of pre-fire pasture management, so that pre-fire range health scores could be used as a predictor of post-fire plant production recovery. These scores were also used as a predictor to assess differences between burned and non-burned plant communities in NMDS ordinations. Briefly, the Alberta Rangeland Health Assessment score is based on the following components: plant species community, plant community structure (vegetation layers), hydrologic function and nutrient cycling, site stability and erosion, and noxious weeds (Adams et al. 2016). Scores are out of 100% and indicate different rangeland health categories where greater than 75% is healthy, 50 to 74% is healthy with problems, and below 50% is unhealthy. Species composition and community structure is compared relative to long-term reference plant communities. These reference plant communities are based off range sites that are lightly grazed or ungrazed and represent the 'potential plant community' for any given ecological site. Hydrologic function is measured through litter quantity. Stability is assessed through visual evidence of erosion and human-caused bare soil. Noxious weeds score is evaluated on their presence, cover, and density (Adams et al. 2016).

3.3.7 Statistical Analysis

Statistical analyses were conducted using R (R Core Team 2018). Significance for all tests was assessed at alpha = 0.05. All mixed effects models used the *1me4* package and *lmer* function

(Bates et al. 2015) and Tukey's honestly significant difference (HSD) test to assess post-hoc differences. A mixed effects model was used to examine effects of fall fire on forage quantity and litter quantity for 2018, 2019, and 2020. Grass, forb, and total biomass were log (x+1) transformed, while litter mass was log (2x+1) transformed. In these analyses, the main effects were burn, year, and their interaction, and site was nested within pasture as a random effect. Total biomass was tested separately on burned and non-burned subplots using a mixed model with litter mass, year, and the interaction to assess if the amount of litter material had an effect on total biomass. Total biomass was also assessed in mixed model using pre-burned range health score, burn, year, and interactions to determine if pre-burned range health scores influenced post-fire total biomass. Total biomass in the burned/non-burned litter mixed models as well as the pre-burned range health mixed model was log (x+1) transformed.

Forage quality, assessed as percent crude protein and crude protein yield, was also analyzed using a mixed effects model for 2018 and 2019. Grass and forb percent crude protein were not transformed, while grass, forb, and total crude protein yield were all log (x+1) transformed. The main model effects for percent crude protein and crude protein yield were burn, year, and their interaction with site nested within pasture as a random effect. Percent crude protein and crude protein yield post-hoc tests were calculated within year.

Species richness, Shannon's Diversity Index, and permutational analysis of variance (PERMANOVA) were run separately in 2018, 2019, and 2020, with the main model effect as burn and site as a random effect to identify if the wildfire treatment had an influence on species composition within year. Indicator species analysis was also run for each year. Further, NMDS ordinations were run separately in 2018, 2019, and 2020 with significant environmental variables and indicator species overlaid.

3.4 Results

3.4.1 Plant Production

Total biomass was affected by burn, year, and burn*year (Table 3.1). In 2018, 2019, and 2020, non-burned total biomass was greater than burned, by 46, 26, and 25%, respectively, representing slow recovery over time within the burned subplots towards that of the non-burned (Figure 3.1). Grass and forb biomass were affected by burn, but not by year or burn*year, although there were some marginally significant effects on grass biomass ($p = 0.078$) so I have examined these patterns (Table 3.1). Grass biomass was lower in burned subplots in 2018 and 2019, but by 2020, burned and non-burned did not differ (Figure 3.1). Although forb biomass did not differ between years, it was plotted to illustrate effects to total biomass (Figure 3.1). The average forb biomass in burned subplots across all years was 100±18 kg/ha, which was lower ($p < 0.05$) than that within the non-burned plots across all years at 125±14 kg/ha. Litter mass was affected by burn, year, and burn*year (Table 3.1). Litter mass in burned subplots was lower than non-burned subplots having been eliminated in 2018, but only accumulating to 35% of that of non-burned subplots by 2020 (Figure 3.1). When examining site factors that could affect plant biomass, I found that litter mass on burned and non-burned subplots was correlated with total biomass (Table 3.2). Further, pre-burned range health scores were correlated with total biomass (Table 3.2). More litter mass resulted in greater total biomass in burned and non-burned subplots, however increases in litter mass on non-burned areas appear to have slight negative influences on total biomass after approximately 1000 kg/ha (Figure 3.2). Higher pre-burned range health also had positive effects on post-fire total biomass, yet there was no difference between burn treatments (Table 3.2, Figure 3.3).

3.4.2 Percent Crude Protein and Crude Protein Yield

In July 2018 and 2019, there was no difference in percent crude protein for grasses or forbs between burned and non-burned subplots, although percent crude protein for both morphological groups were greater in 2019 than 2018 (Table 3.3, Figure 3.4). Total and grass crude protein yield was greater in non-burned subplots in both 2018 and 2019. Forb crude protein yield responses to burning were marginally significant ($p<0.06$); forb crude protein yield was lower in burned subplots in 2018, but did not differ in 2019 (Figure 3.4).

3.4.3 Species Composition

Species richness and Shannon's Diversity Index showed no difference between burned and non-burned areas in 2018, 2019, or 2020 (Table 3.4). PERMANOVAs showed no difference between burn treatments in 2018 or 2019, but there was a difference in 2020 (Table 3.4). Environmental variables tested for effects on species composition included bare ground cover, litter cover, ground cover (*Selaginella densa* (Rydb.), cow mature, lichen, and moss), total biomass, grass biomass, forb biomass, and range health score. Indicator species analysis showed that *Chenopodium spp.* (CHEN_SPP) was more common on burned subplots in 2018, while in 2019 and 2020 there were no indicator species. The ordination of species composition in 2018 shows a lack of burn effect, however the overlaid environmental variables that were significantly correlated with the ordination overlaid: grass biomass ($R^2 = 0.112$, p-value < 0.05), forb biomass ($R^2 = 0.233$, p-value < 0.05), and total biomass ($R^2 = 0.171$, p-value < 0.05) (Figure 3.5). An ordination of species composition in 2020 illustrates the difference between burned and non-burned subplots with environmental variables that were significantly correlated with the ordination overlaid: bare ground cover ($R^2 = 0.184$, p-value < 0.05), litter cover ($R^2 = 0.332$, p-value < 0.01), and range health score ($R^2 = 0.149$, p-value < 0.05) (Figure 3.6). Although the indicator species analysis did not identify any species associated with burned or non-burned subplots in 2020, the four most common species across all study sites were plotted on the NMDS ordination: *Bouteloua gracilis*, *Carex spp*. (Linn.), *Elymus spp*., and *Hesperostipa comata* (Figure 3.6). All common species were close to the center of burned and non-burned species composition ellipses (Figure 3.6).

3.4.4 Precipitation

To interpret biomass responses, I summarized precipitation patterns during the growing season (April to July) (Figure 3.7). Accumulated growing season precipitation in the months leading up to the October 17, 2017 wildfires was comparable to historical precipitation levels (Figure 3.7). In the few months leading up to the wildfires, there was an extended drought in the study area, with only 59% of long-term average precipitation in August, 51% in September, and 87% from October 1st to October 16th. By the end of July in 2018 and 2019, accumulated growing season precipitation approximated the historical average, at 95% in 2018 and 85% in 2019. By the end of July 2020, precipitation was 154% of the historical average (Government of Alberta 2020).

3.5 Discussion

My results indicate that by the third growing season following these wildfires, average total production on burned subplots remained lower than those on the non-burned, which suggests that plant biomass production has not yet recovered, and this happened despite relatively normal precipitation (within 15%) during the first two post-fire growing seasons and higher than average in the third growing season. Similar results have been found near my study sites, in an Alberta mixedgrass prairie, where the recovery of plant production took three years or more because of prolonged drought conditions (Clarke et al. 1947; Erichsen-Arychuk et al. 2002). On my study sites, precipitation was at or below average in the first two growing seasons following wildfire, which may be contributing to delayed recovery of plant production. My results also indicate that forb biomass is more resistant (i.e., less likely to decline) after fall wildfire with average post-fire precipitation because it was not affected by burn in any year, while grass biomass is susceptible to declines in production but moderately resilient, having increased to comparable levels to that of the non-burned areas by the third growing season following fire. My results show that grass biomass is lower but comparable by the third post-fire growing season and that forb biomass is resistant, however total biomass is still different by the third growing season after fire and that indicates that burned areas have not recovered during the duration of this experiment.

Sites with a higher pre-burned range health score had greater plant biomass after burning, and this pattern did not appear to differ between burned and non-burned subplots. Rangeland health is an indicator of plant vigour because different levels of grazing disturbance influence plant vigour (Adams et al. 2016). There are not many studies evaluating pre-fire pasture management effects on post-fire recovery, other than studies indicating that higher litter accumulation resulted in a longer period for recovery (Erichsen-Arychuk et al. 2002). However, a recent study found that moderate grazing before burning increased plant community resistance to invasive species (Davies et al. 2016). These results in combination with my own indicate that pasture management that improves rangeland health can have a positive influence on post-fire recovery. Future research should focus on how pre-fire conditions affect post-fire recovery, including the specific contribution of individual range health scoring components.

Three years following wildfire, litter on burned subplots had accumulated to only 35% of the non-burned, which may explain the reduced plant production in burned areas. In an experiment near my study sites in the mixedgrass prairie, litter removal for three consecutive years led to a 57% reduction in plant production due to increased soil temperature and evaporation (Willms et al. 1986), which suggests that litter is critically important for plant productivity in this arid environment where moisture is highly limiting for plant growth (Willms and Jefferson 1993). In my study, litter mass was correlated with plant production for both burned and non-burned subplots, which demonstrates how important the litter layer is for plant production.

My PERMANOVA showed there was no difference in plant species composition between burned and non-burned subplots in 2018 or 2019; however, in 2020, there was a difference. Further, species richness and Shannon's diversity index did not differ in any year during the duration of my experiment. My plant species NMDS ordination was overlaid with explanatory environmental variables: litter cover, bare ground cover, and range health score. Bare ground cover and litter cover were directionally opposite on the 2020 plant species ordination with bare ground cover slightly more related to burned subplots and litter cover more related to non-burned. Similarly, studies have found that fire reduces litter and increases bare ground, thereby increasing temperatures and water loss via evaporation (Erichsen-Arychuk et al. 2002). My results show that plant species composition following these particular fires remained relatively resistant to burning and previous studies in the mixedgrass prairie have also found that the dominant plant community, *Hesperostipa comata*-*Bouteloua gracilis*, is tolerant of fire (Clarke et al. 1947). Indicator species analysis on my plant species data did not yield any results in 2020 and within the NMDS ordination, the four most abundant plant species in my experiment, *H. comata*, *B. gracilis*, *Carex spp.*, and *Elymus spp.*, did not respond to burning. Although there was a difference in plant species composition in 2020, as illustrated in the PERMANOVA and NMDS ordination, there were no indicator species and dominate species did not appear to be affected on the ordination. Increased rainfall levels in 2020 may be a possible explanation for the difference in plant species composition between burned and non-burned.

There was no difference in percent protein in July 2018 or July 2019. The lack of effect in this experiment may be because it was already the end of the growing season. In Chapter 2, my results indicated that there was an increase in percent protein for grasses and forbs at the beginning of the growing season, but these differences disappeared by the end, which is similar to the results found in this experiment (i.e. biomass collected at peak biomass did not differ between burned and non-burned). Other studies have reported no difference in percent protein by the end of the growing season (Dufek et al. 2014). Similarly, studies have reported increased utilization on the burned portion of pastures containing both burned and non-burned areas during the first growing season following wildfire, with no difference in the following year, which could be explained by initial increases in forage quality (Erichsen-Arychuk et al. 2002). There were differences between burned and non-burned subplots for total and grass protein availability, but these differences are driven by greater average plant biomass in non-burned subplots.

3.6 Summary

Plant production can take several years to recover following burning, but recovery is highly variable due to pasture management and environmental conditions. Because of variable lengths of recovery across the Great Plains of North America, the predominant management recommendation following fire is an extended period of rest from grazing (Bureau of Land Management; Government of Alberta 2018). My results indicate that average total productivity on burned subplots remained lower than non-burned by the third growing season following fire, which suggests that plant biomass production has not recovered. Although forb biomass was resistant to fire and grass biomass appears resilient after several years, total biomass remained reduced three years after burning. Further, my results indicated that higher non-burned range health scores were positively related to total biomass in burned and non-burned pastures. Consequently, management that maintains or improves rangeland health may have a positive influence on the recovery of post-fire plant production.

Plant species composition did not differ between burned and non-burned subplots in 2018 or 2019; however, in 2020, there was a difference. In 2020, the environmental variables explaining the difference between burned and non-burned subplots were bare ground cover, litter cover, and range health score. My ordination illustrates that burning did not have an influence on the dominant plant species, *H. comata*, *B. gracilis*, *Carex spp.*, and *Elymus spp*.

Three years following wildfire, litter mass on burned subplots had accumulated to only 35% of non-burned subplots and previous studies have found that litter is critically important for plant production (Willms et al. 1986). Further, I found a relationship between litter mass and total biomass on burned and non-burned subplots in my experiment. However, my experiment did not include direct use by livestock, which could increase soil erosion from grazing and trampling (Naeth et al. 1991). Litter significantly affected plant production in my experiment and was reduced following burning, which could explain reduced plant productivity in burned areas. Litter material is an essential component of rangeland health and recovery, therefore post-fire management should ensure that litter will accumulate for increased soil moisture retention that will aid in future plant productivity.

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Tables

Table 3.1. Results of generalized linear model testing effects of wildfire and year on total, grass, forb, and litter. P values <0.05 are indicated in bold font.

Variable	*df*	*F*	*p*
Total Biomass (g / 0.25 m^2)			
Burn	1	57.5	**<0.001**
Year	2	4.61	**0.012**
Burn*Year	2	3.81	**0.025**
Grass Biomass (g / 0.25 m^2)			
Burn	1	45.9	**<0.001**
Year	2	2.63	0.076
Burn*Year	2	2.69	0.072
Forb Biomass (g / 0.25 m^2)			
Burn	1	6.29	**0.013**
Year	2	0.34	0.71
Burn*Year	2	0.55	0.58
Litter Mass (g / 0.25 m^2)			
Burn	1	368	**<0.001**
Year	2	24.8	**<0.001**
Burn*Year	2	36.5	**<0.001**

Table 3.2. Results of generalized linear model testing total biomass as a function of litter in burned and non-burned subplots using pooled data from 2018 to 2020. Results of generalized linear model testing total biomass as a function of pre-burn range health score and wildfire using pooled data from 2018 to 2020. P values <0.05 are indicated in bold font.

Variable	***df***	***F***	***p***
Total Biomass (g / 0.25 m^2)			
Litter (Burn sites)	1	17.6	**<0.001**
Total Biomass (g / 0.25 m^2)			
Litter (Non-Burn sites)	1	22.6	**<0.001**
Total Biomass (g / 0.25 m^2)			
Pre-Burn Range Score	1	9.77	**<0.01**
Burn	1	0.35	0.55
Pre-Burn Range Score*Burn	1	1.43	0.23

Table 3.3. Results of generalized linear model testing effects of a wildfire on percent crude protein and crude protein yield in 2018 and 2019. P values <0.05 are indicated in bold font.

Variable	***df***	***F***	***p***
Grass Percent Crude Protein			
Burn	1	0.031	0.86
Year	1	67.1	**<0.001**
Burn*Year	1	0.37	0.55
Forb Percent Crude Protein			
Burn	1	0.042	0.84
Year	1	6.64	**0.013**
Burn*Year	1	0.22	0.64
Total Crude Protein Yield (g / 0.25 m^2)			
Burn	1	44.7	**<0.001**
Year	1	14.8	**<0.001**
Burn*Year	1	3.21	0.077
Grass Crude Protein Yield (g / 0.25 m^2)			
Burn	1	16.1	**<0.001**
Year	1	13.2	**<0.001**
Burn*Year	1	0.036	0.85
Forb Crude Protein Yield (g / 0.25 m^2)			
Burn	1	3.71	0.060
Year	1	0.97	0.33
Burn*Year	1	1.81	0.19

Table 3.4. Results of generalized linear model testing effects of a wildfire on species richness, Shannon's Diversity, as well as permutational analysis of variance (PERMANOVA) testing effects of wildfire on species composition in 2018, 2019, and 2020. P values <0.05 are indicated in bold font.

Variable	***df***	***F***	***p***	***df***	***F***	***p***	***df***	***F***	***p***
		2018			**2019**			**2020**	
Species Richness									
Burn	1	0.17	0.68	1	0.32	0.58	1	0.12	0.73
Shannon's Diversity									
Burn	1	0.27	0.61	1	0.65	0.43	1	0.80	0.38
PERMANOVA									
Burn	1	1.17	0.28	1	1.21	0.23	1	1.60	**0.048**

Figures

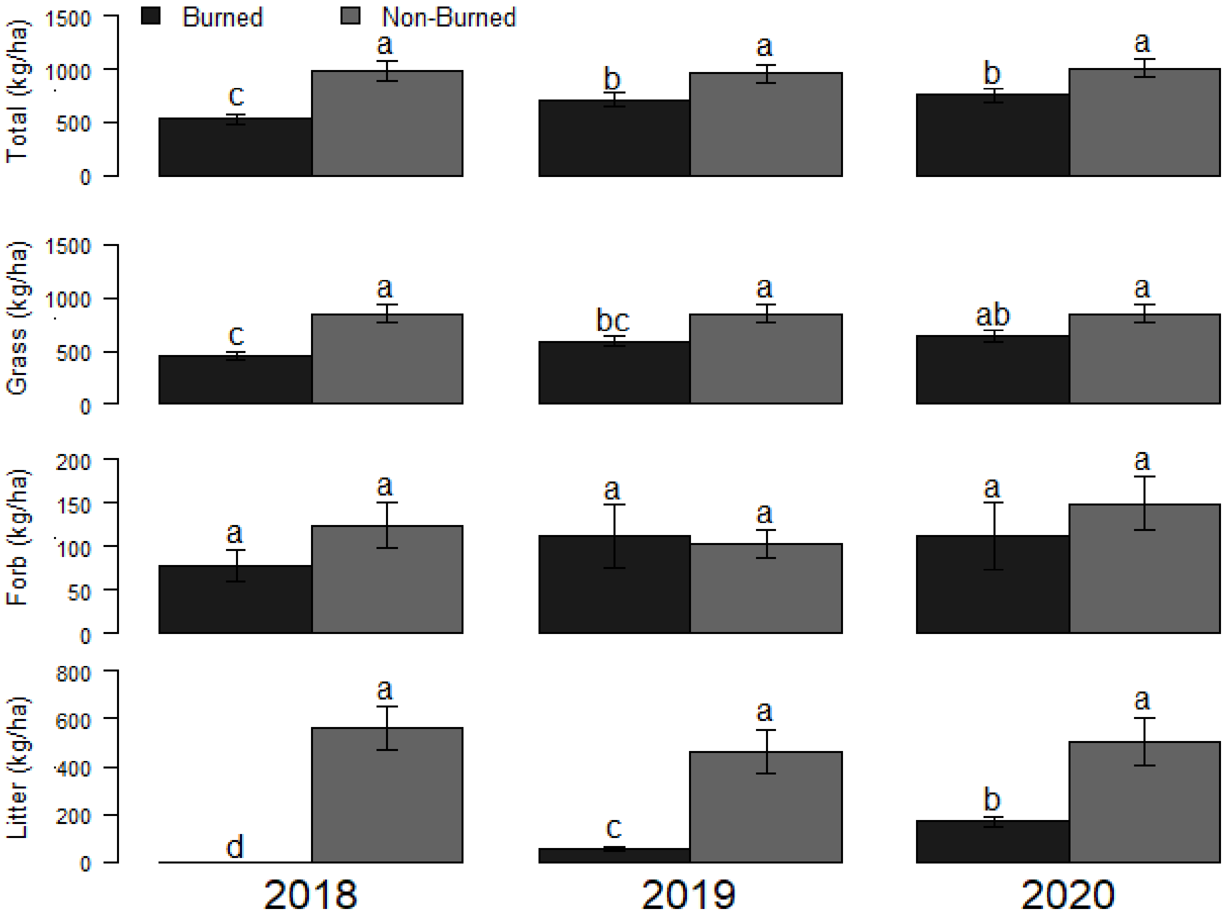

Figure 3.1. Total, grass, forb, and litter in kg/ha (SE) removed in 2018, 2019, and 2020. Standing biomass (total, grass, forbs) clipped to 2 cm in burned and non-burned subplots in July. Litter material raked by hand in burned and non-burned subplots in July. Bars sharing the same letters are not significantly different (Tukey HSD test, $p < 0.05$).

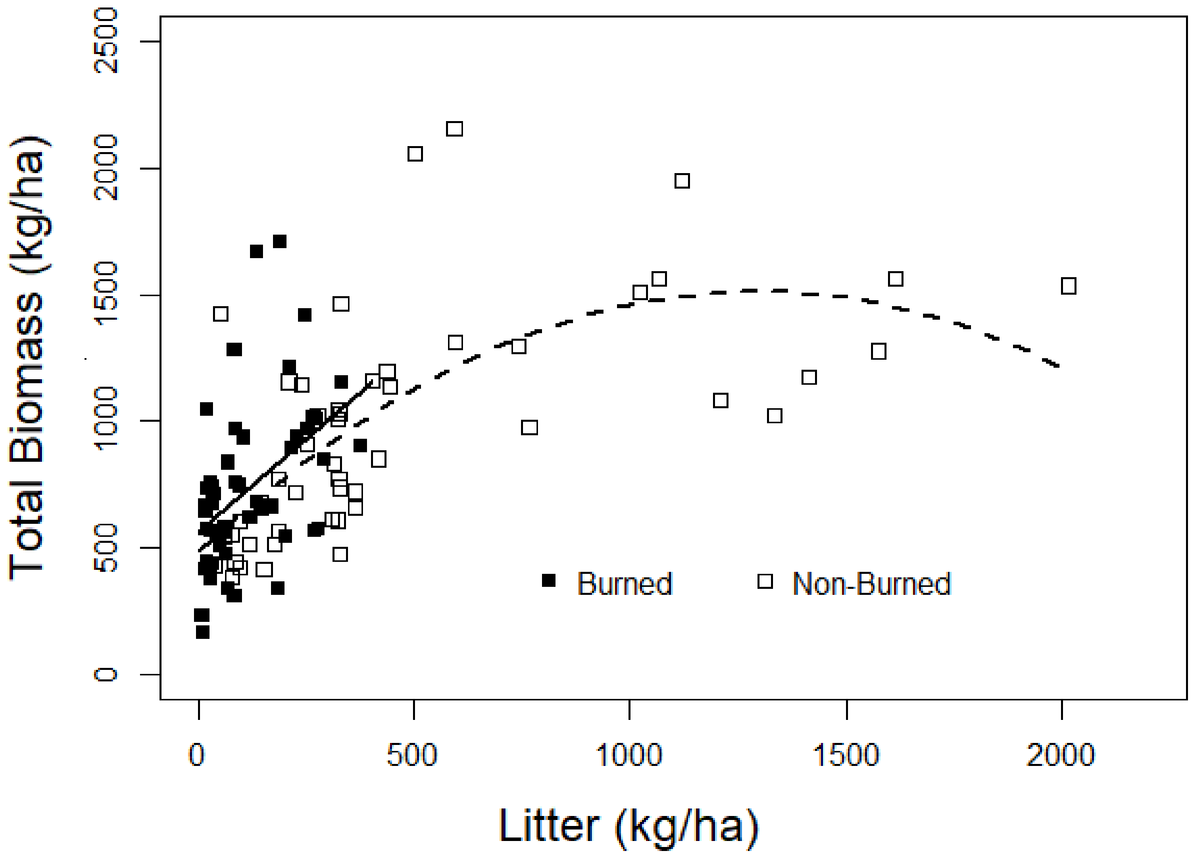

Figure 3.2. Total biomass as a function of litter mass following wildfire for 25 sites from 2018 to 2020 (Burn: Adjusted R-squared = 0.18, p-value = <0.01; Non-Burn: Adjusted R-squared = 0.443, p-value = <0.001).

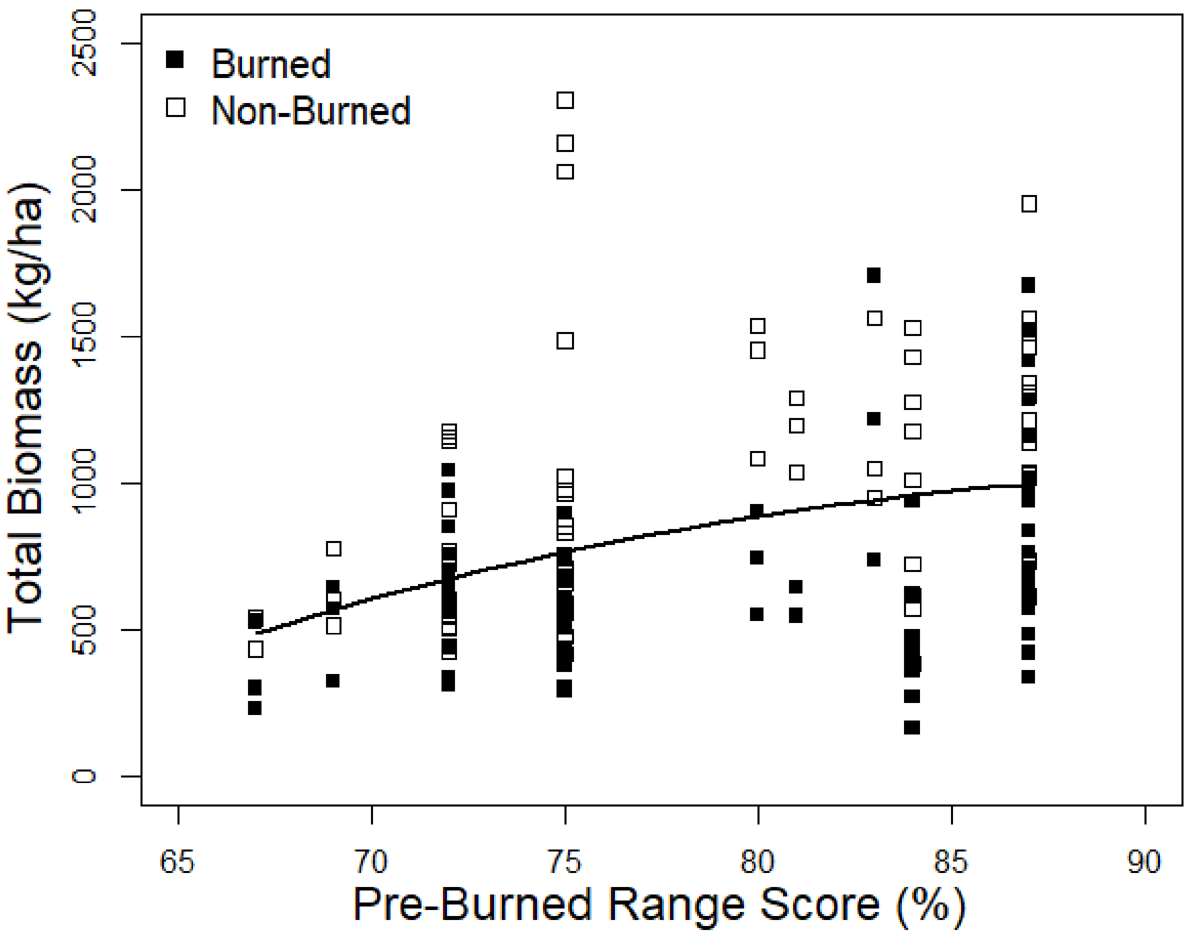

Figure 3.3. Total biomass as a function of non-burned range health score following wildfire for 25 sites from 2018 to 2020 (Adjusted R-squared = 0.12, p-value = <0.001).

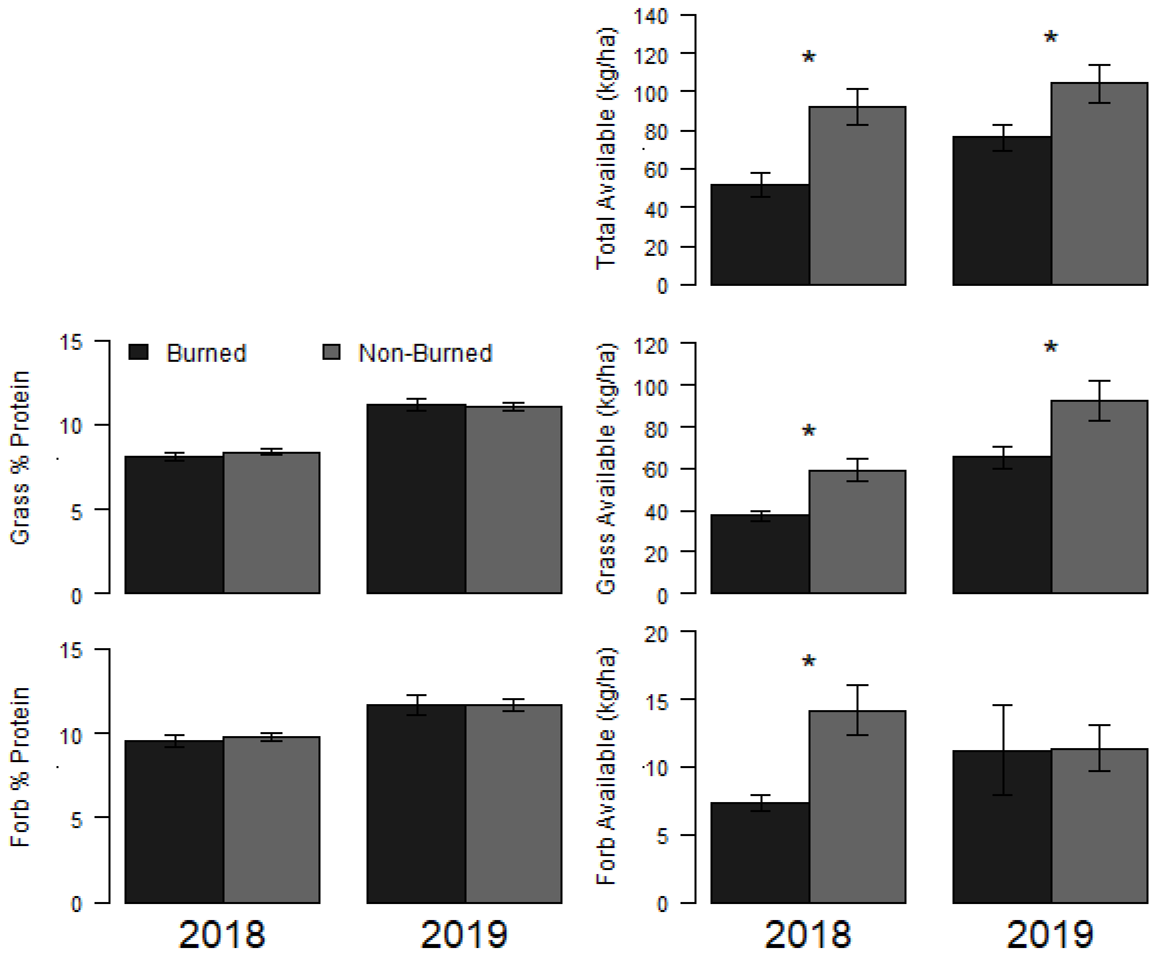

Figure 3.4. Grass and forb percent crude protein (SE), as well as total, grass, and forb crude protein yield (kg/ha SE) on burned and non-burned subplots from 2018 and 2019. A * above bars indicates a significant difference within year (Tukey HSD test, $p < 0.05$).

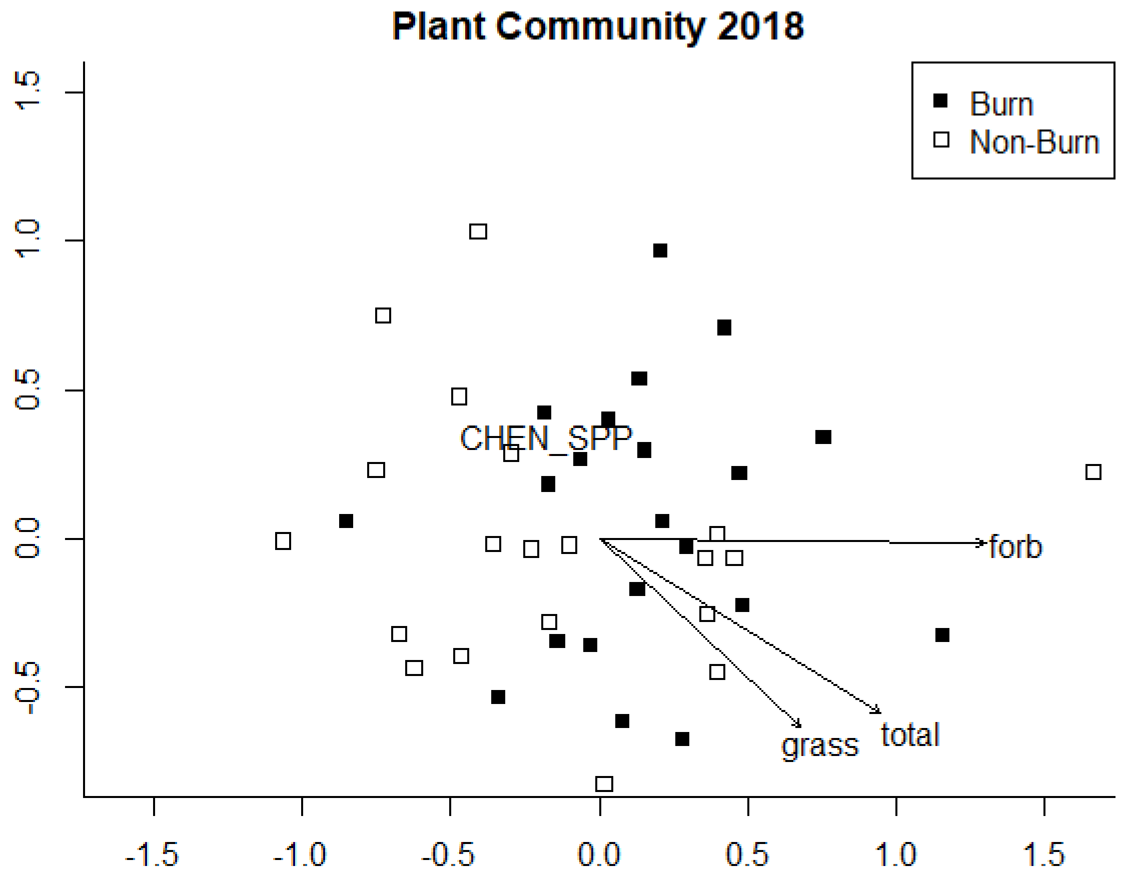

Figure 3.5. NMDS Ordination of 2018 burned and non-burned plant communities. The overlaid explanatory environmental variables are grass biomass ($R^2 = 0.11$, p-value < 0.05), forb biomass ($R^2 = 0.23$, p-value < 0.05), and total biomass ($R^2 = 0.17$, p-value < 0.05). Indicator species analysis found that *Chenopodium spp*. (CHEN_SPP) was associated with burned subplots.

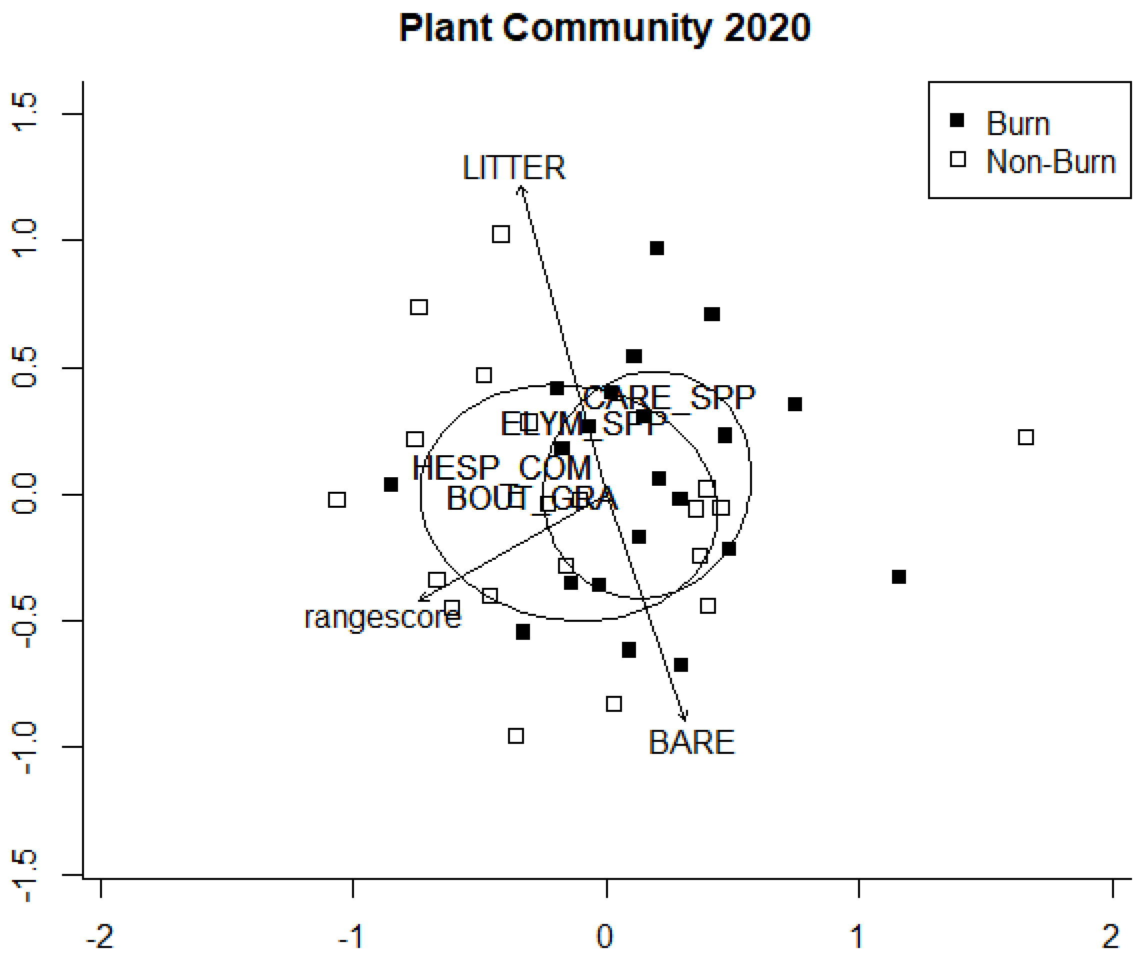

Figure 3.6. NMDS Ordination of 2020 burned and non-burned vegetation communities. Ellipses depict the plant species community for burned and non-burned subplots with the significant environmental variables overlaid: bare ground cover ($R^2 = 0.18$, p-value < 0.05), litter cover ($R^2 = 0.33$, p-value < 0.01), and range health score ($R^2 = 0.15$, p-value < 0.05). The four most common species across all sites were added to the ordination: *Bouteloua gracilis* (BOUT_GRA), *Carex spp.* (CARE_SPP), *Elymus spp.* (ELYM_SPP), and *Hesperostipa comata* (HESP_COM).

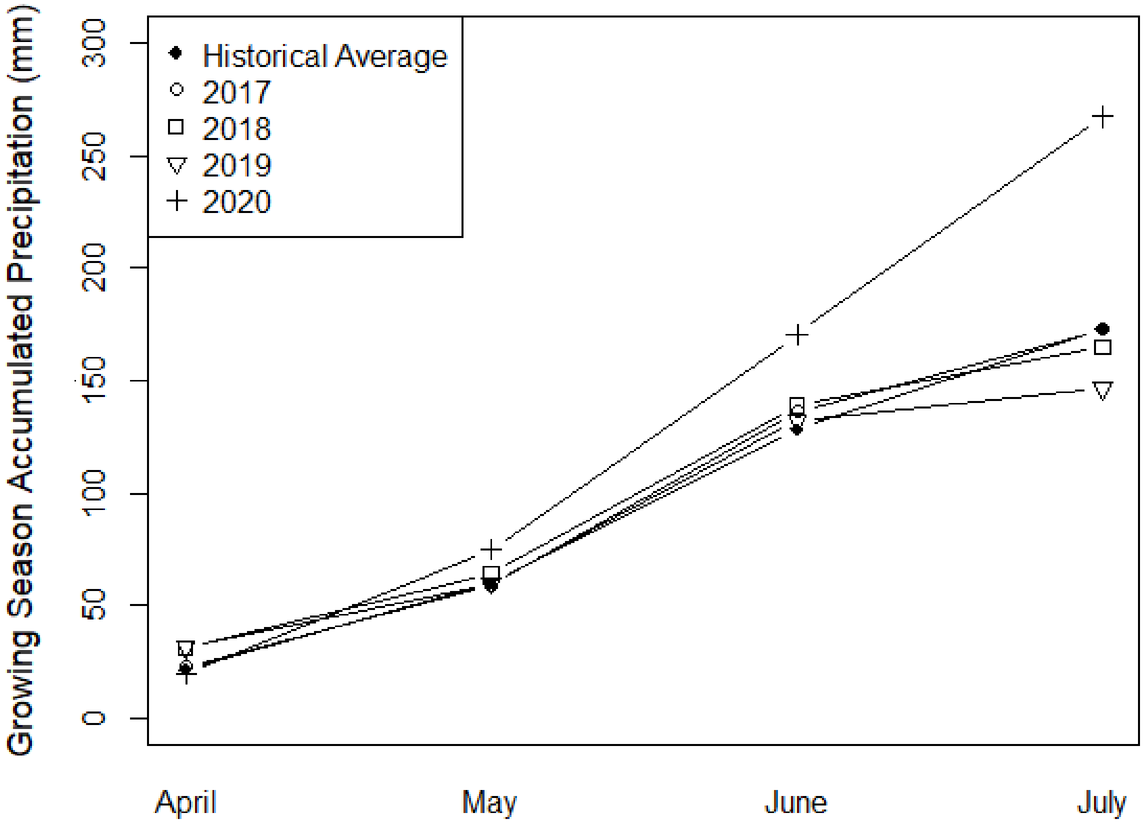

Figure 3.7. Mean accumulated growing season precipitation (April to July) from 2017 to 2020 with comparison to long-term average precipitation. Precipitation data is averaged from the three nearest Alberta Agriculture and Forestry weather stations (Acadia Valley, Schuler, and Social Plains) to the October 2017 wildfires (Government of Alberta 2020). Long-term average precipitation data is from 1961-2020.

Chapter 4: Post-fire Rangeland Management Recommendations

4.1 Synthesis

At the start of this project, I had two main objectives that I wanted to assess throughout the project duration. First, I tested if timing of defoliation in the first growing season following fall wildfire in native mixedgrass prairie would have an impact on vegetative recovery in the subsequent year. Second, I monitored wildfire effects on various environmental variables for assessment of grassland vegetation recovery in order to determine what length of time was required for post-fire recovery. The results gathered from these quantitative experiments answered my initial objectives and added to the body of scientific knowledge on the effects of grassland fire, with particular relevance to the mixedgrass prairie of the northern Great Plains.

In Chapter 2, my objective was to determine if timing of defoliation in the first growing season following wildfire would affect recovery in the subsequent year. This arose because of recent studies suggesting that post-fire rest from grazing is not necessary (Vermiere et al 2014; Gates et al. 2017), despite most jurisdictions in North America recommending rest on public lands following fire (Bureau of Land Management 2002; Government of Alberta 2018). My results indicate that by the second growing season following wildfires plant production did not differ between burned and non-burned treatments; however, the July defoliation reduced plant biomass compared to non-clipped control and June defoliations. This difference in effect from defoliation timing could be the result of my fixed clipping height (2 cm) because plants defoliated in June had less biomass removed and were probably not as stressed by the clipping treatment as those defoliated in July (Bogen et al. 2003). Importantly, litter mass was greatly reduced by the burn and further reduced by all defoliation treatments, which may have long-term implications for the recovery of plant production given the critical role of litter in maintaining soil moisture (Willms et al. 1986). In this region, it is recommended to graze native pastures no earlier than mid-July (Bailey et al. 2010), so it is important share negative impacts to forage production and litter accumulation with rangeland managers if they plan to graze burned pastures in the first growing season after fire. This chapter also found that vegetative percent crude protein was high on burned subplots at the beginning of the first growing season following wildfires, but did not differ by the end.

In Chapter 3, my objective was to monitor the effects of the wildfires and assess the length of time required for vegetative recovery. Length of post-fire rangeland recovery is highly variable due to many different environmental variables and pasture management, therefore I assessed factors that contribute to post-fire recovery in order to aid land managers in future management decisions. The results showed that by the third growing season following wildfire, total plant production on burned subplots was still lower than non-burned. Litter was also greatly reduced by the third growing season following wildfires, which could be affecting plant productivity. Similar to Chapter 2, there was no difference in percent crude protein between burned and non-burned by the end of the first growing season. Species composition was not different between burned and non-burned in the first or second growing seasons following wildfires, but there was a difference in the third. This difference could be the result of a differential effect of greater than average precipitation that year on the burned and non-burned subplots. Importantly, sites with higher range health scores had more plant biomass. I also found a positive relationship between total biomass and litter mass on burned and non-burned subplots. Finally, this chapter demonstrated that even with greater than average precipitation burned subplots still were not producing the same amount of biomass as non-burned.

Combined, Chapters 2 and 3 address important questions about post-fire grazing management and how grasslands recover from fire. Both chapters indicate that litter is greatly reduced on burned subplots. Chapter 2 illustrates that litter is further reduced by defoliation, which has also been found in previous studies (Bates et al. 2009). Chapter 3 shows that burned subplot litter mass is still greatly reduced (65% lower) by the third growing season following wildfires. Further, total biomass was positively correlated with litter mass in both burned and non-burned subplots. A reduced litter layer could have a long-term influence on plant productivity, especially in drought years (Willms et al. 1986; Erichsen-Arychuk et al. 2002). Additionally, the results in both chapters found that vegetative percent crude protein did not differ between burned and non-burned by the end of the first growing season; although chapter 2 results did show an increase in percent crude protein at the beginning of the first growing season.

There are some apparent discrepancies between Chapters 2 and 3. Chapter 2 concurs with some recent papers (Vermiere et al. 2014; Gates et al. 2017) that grazing in the first post-fire growing season may not affect plant biomass, and Chapter 3, similar to other studies (Erichsen-Arychuk et al. 2002), indicates that recovery, even without grazing, can take multiple years. There may be a number of possible explanations for the different conclusion between chapters. First, Chapter 2 is a subset of sites in Chapter 3 and may represent a pattern from this random subset. Second, sample size may be too small in Chapter 2 with not as much variation captured as in Chapter 3, both at the site level and across the wildfire study region. This raises questions on grassland fire research studies that do not cover a wide variety of microclimates and pasture management strategies. This project continues for another year and future analysis will focus on if there is a relationship between wildfire and subsequent grazing management on the recovery of plant production.

4.2 Management Considerations

Due to variable rates of post-fire recovery, current recommendations for rest from grazing following burning appear appropriate to ensure litter accumulates and plant productivity recovers. I recommend post-fire rest despite my contradictory results from Chapter 2 and 3. Chapter 2 results indicated that post-fire defoliation resulted in slower litter accumulation, which may affect future plant productivity. Post-fire management must ensure that litter builds up to give the plant community the ability to limit evaporation and in turn improve plant productivity (Willms et al. 1986).

Another key finding from my research is that pre-fire range health scores influence post-fire plant production. Range health scores can potentially indicate plant vigour, but the underlying mechanism for this relationship requires further examination. This is important to share with land managers because if pastures are managed at appropriate stocking rates, then post-fire recovery appears to be faster.

Finally, any benefits from increased forage quality had disappeared by the end of the first growing season following wildfires. However, if land managers decide or are required to graze burned pastures in the first year following fires, then they should be careful about the timing. July grazing, which is the grazing period recommended on native rangelands in this region (Bailey et al. 2010), may result in negative impacts to plant productivity in the subsequent growing season due to a depleted litter layer. Land managers should also be aware that livestock will preferentially graze burned areas on partially burned pastures in the first growing season following fire (Erichsen-Arychuk et al. 2002).

4.3 Future Research

This research has added to the significant body of scientific information of rangeland fires; however, it has also raised some important questions. My research has found that July defoliation in the first post-fire growing season may reduce subsequent growing season plant production. This research also found that non-burned rangeland health score influenced post-fire plant production, which indicates pre-fire pasture management may lead to shorter post-fire recovery time. Finally, discrepancies between my Chapter 2 and Chapter 3 results indicate that future experiments should ensure they cover a variety of pre and post-fire management strategies and environmental variables. Future research on grassland fires should focus on improving our understanding of the effects of post-fire defoliation timing, of the influence of pre-fire plant vigour and rangeland health on post-fire recovery, as well as if field experiments capture sufficient variation in pre and post-fire management strategies and environmental variables to draw robust conclusions.

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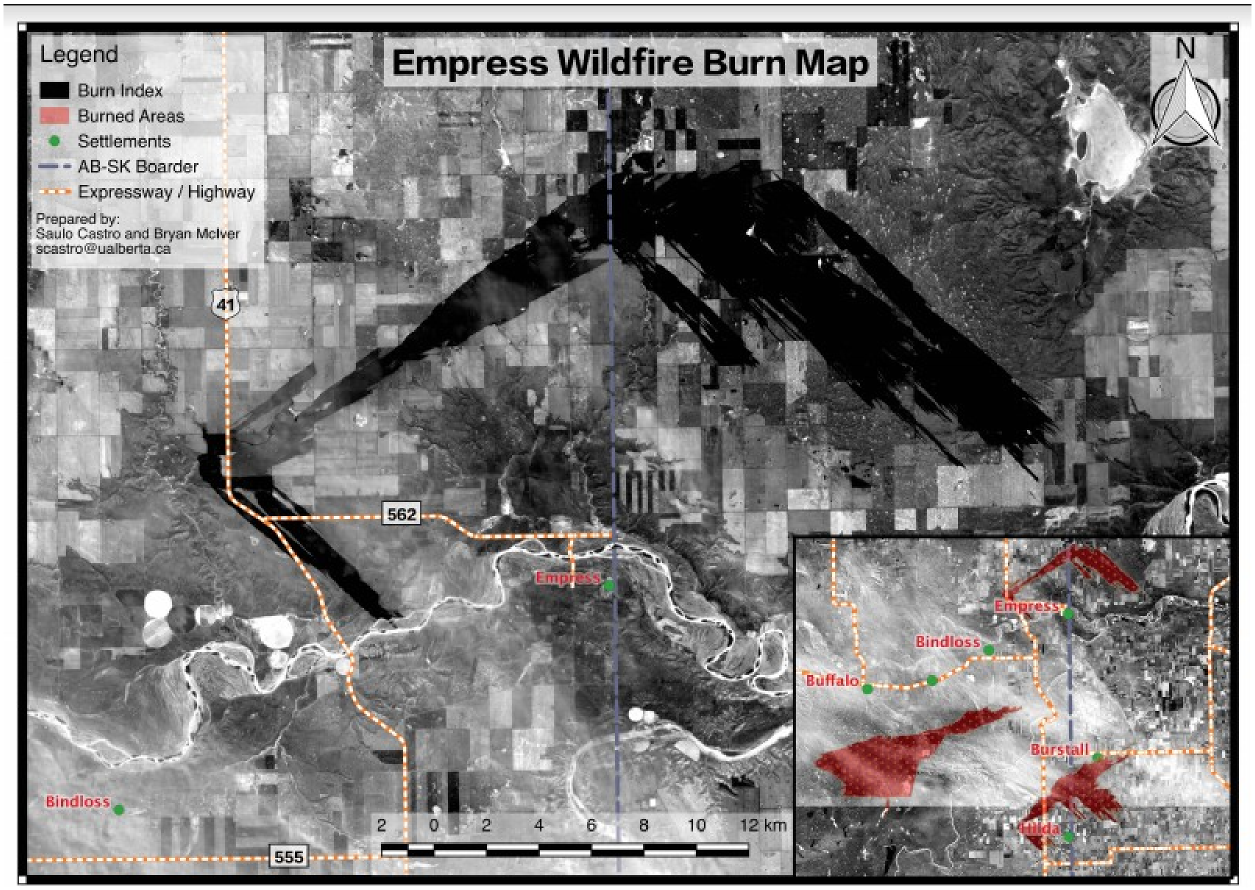
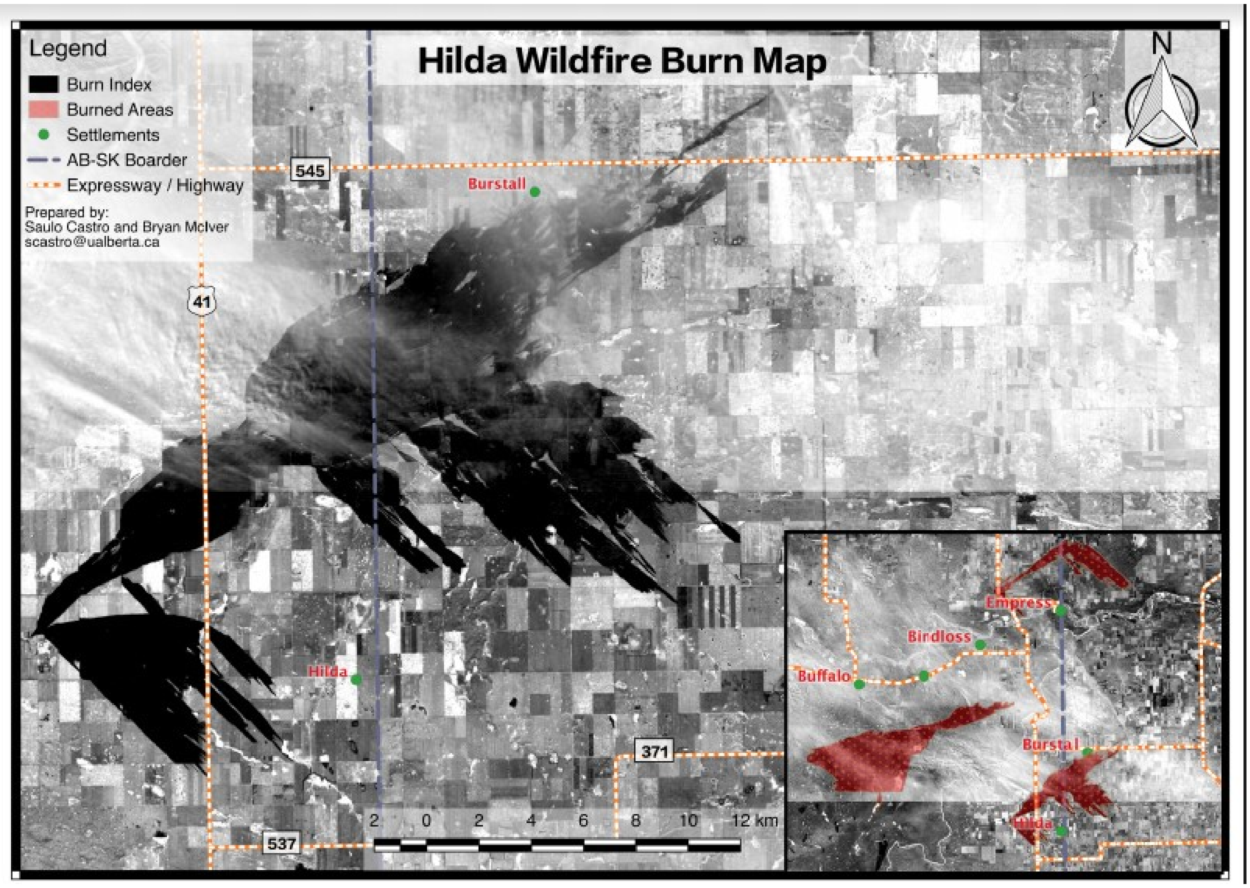
Appendix A. Wildfire Burn Maps



Appendix A - Figure 1. Empress Wildfire Burn Map. Prepared by Saulo Castro and Bryan McIver.



Appendix A - Figure 2. Hilda Wildfire Burn Map. Prepared by Saulo Castro and Bryan McIver.

Appendix B. Plant Species List

Species	Native	Listed Weed	Morphology
Achillea millefolium	Native	No	Forb
Agoseris glauca	Native	No	Forb
Agropyron cristatum	Non-Native	No	Grass
Agrostis scabra	Native	No	Forb
Amaranthus retroflexus	Non-Native	No	Forb
Androsace septentrionalis	Native	No	Forb
Anemone multifidi	Native	No	Forb
Antennaria spp.	Native	No	Forb
Anthoxanthum hirtum	Native	No	Forb
Artemisia cana	Native	No	Shrub
Artemisia frigida	Native	No	Forb
Artemisia ludoviciana	Native	No	Forb
Artemisia tridentata	Native	No	Shrub
Astragalus agrestis	Native	No	Forb
Astragalus missouriensis	Native	No	Forb
Astragalus pectinatus	Native	No	Forb
Astragalus spp.	Native	No	Forb
Atriplex gardneri	Native	No	Forb
Atriplex spp.	Native	No	Forb
Boechera holboellii	Native	No	Forb
Boechera retrofracta	Native	No	Forb
Bouteloua gracilis	Native	No	Grass
Bromus inermis	Non-Native	No	Grass
Calamagrostis montanensis	Native	No	Grass
Campanula rotundifolia	Native	No	Forb

Carex filifolia	Native	No	Sedge
Carex spp.	Native	No	Sedge
Chamaerhodos erecta	Native	No	Forb
Chenopodium album	Non-Native	No	Forb
Chenopodium fremontii	Native	No	Forb
Chenopodium leptophyllum	Native	No	Forb
Chenopodium pratericola	Native	No	Forb
Chenopodium spp.	Uncertain	No	Forb
Cinquefoil spp.	Uncertain	No	Forb
Cirsium flodmanii	Native	No	Forb
Comandra umbellata	Native	No	Forb
Conyza canadensis	Native	No	Forb
Crepis tectorum	Non-Native	Noxious	Forb
Cryptantha spp.	Native	No	Grass
Dalea purpurea	Native	No	Forb
Danthonia intermedia	Native	No	Grass
Deschampsia cespitosa	Native	No	Grass
Descurainia spp.	Non-Native	No	Forb
Descurainia sophia	Non-Native	No	Forb
Distichlis spicata	Native	No	Grass
Elymus lanceolatus var. lanceolatus	Native	No	Grass
Elymus spp.	Native	No	Grass
Elymus repens	Non-Native	Nuisance	Grass
Elymus trachycaulus ssp. subsecundus	Native	No	Grass
Erigeron caespitosus	Native	No	Forb
Erigeron spp.	Native	No	Forb
Erigeron pumilus	Native	No	Forb
Erysimum inconspicuum	Native	No	Forb

Escobaria vivipara	Native	No	Forb
Fallopia convolvulus	Non-Native	No	Forb
Festuca hallii	Native	No	Grass
Galium boreale	Native	No	Forb
Gaura coccinea	Native	No	Forb
Geum triflorum	Native	No	Forb
Grindelia squarrosa	Native	No	Forb
Gutierrezia sarothrae	Native	No	Forb
Hesperostipa comata	Native	No	Grass
Hesperostipa curtiseta	Native	No	Grass
Heterotheca villosa	Native	No	Forb
Hordeum jubatum	Native	No	Grass
Kochia scoparia	Non-Native	Noxious	Forb
Koeleria macrantha	Native	No	Grass
Krascheninnikovia lanata	Native	No	Forb
Lactuca serriola	Non-Native	Noxious	Forb
Lappula squarrosa	Non-Native	No	Forb
Liatris punctate	Native	No	Forb
Lygodesmia juncea	Native	No	Forb
Medicago sativa	Non-Native	No	Forb
Melilotus officinalis	Non-Native	No	Forb
Moss	Ground Cover	No	Ground Cover
Muhlenbergia cuspidata	Native	No	Grass
Nassella viridula	Native	No	Grass
Opuntia polyacantha	Native	No	Shrub
Oxytropis sericea	Native	No	Forb
Pascopyrum smithii	Native	No	Grass
Pediomelum argophyllum	Native	No	Forb

Penstemon gracilis	Native	No	Forb
Penstemon procerus	Native	No	Forb
Penstemon spp.	Native	No	Forb
Petalostemon purpureum	Uncertain	No	Forb
Phleum pratense	Non-Native	No	Grass
Phlox hoodii	Native	No	Forb
Plantago major	Non-Native	No	Forb
Plantago patagonica	Native	No	Forb
Poa cusickii	Native	No	Grass
Poa pratensis	Non-Native	No	Grass
Poa secunda	Native	No	Grass
Potentilla arguta	Native	No	Forb
Potentilla concinna	Native	No	Forb
Potentilla hippiana	Native	No	Forb
Potentilla pensylvanica	Native	No	Forb
Potentilla spp.	Native	No	Forb
Pulsatilla patens	Native	No	Forb
Rosa arkansana	Native	No	Shrub
Rosa acicularis	Native	No	Shrub
Rosa woodsii	Native	No	Shrub
Rumex crispus	Non-Native	No	Forb
Rumex occidentalis	Native	No	Forb
Salsola tragus	Non-Native	Noxious	Forb
Selaginella densa	Native	No	Ground Cover
Solidago canadensis	Native	No	Forb
Solidago missouriensis	Native	No	Forb
Solidago spp.	Native	No	Forb
Sonchus arvensis ssp. uliginosus	Non-Native	Noxious	Forb

Sonchus spp.	Non-Native	Noxious	Forb
Sphaeralcea coccinea	Native	No	Forb
Sporobolus cryptandrus	Native	No	Grass
Symphyotrichum ericoides	Native	No	Forb
Symphyotrichum falcatum	Native	No	Forb
Symphyotrichum laeve	Native	No	Forb
Taraxacum officinale	Non-Native	Nuisance	Forb
Thermopsis rhombifolia	Native	No	Forb
Thinopyrum ponticum	Non-Native	No	Grass
Thlaspi arvense	Non-Native	No	Forb
Tragopogon dubius	Non-Native	Nuisance	Forb
Vicia americana	Native	No	Forb
Xanthisma spinulosum	Native	No	Forb
Zygadenus venenosus	Native	No	Forb

Appendix C. NIR Protein Analysis

Traditional assessment of forage quality uses lab methods that are costly and time consuming. The use of near infrared (NIR) spectroscopy is comparatively quick and inexpensive compared to the traditional lab methods. While NIR methods are well established for forage monocultures, they are not well established for mixed plant species samples such as those examined here. Here I present the methods used to evaluate the use of NIR to measure protein content in mixed species samples for the mixedgrass prairie.

A total of 2089 mixed species vegetation samples were collected from native grassland locations in Alberta and Saskatchewan, Canada. About half, 1156, of the samples were collected in Saskatchewan, primarily from Kernen Prairie, a native rough fescue prairie near Saskatoon, SK, 300 samples were from multiple locations across Alberta, and 692 samples were from the study sites from the wildfire project in Alberta and Saskatchewan. Individual samples were dried, ground, and subsamples were analyzed for nitrogen content using a Thermo Scientific Flash 1112 or a LECO Protein Analyzer, and a FOSS NIR to obtain spectra.

The sample spectra were exported from FOSS NIR software and converted to plain text files using SpectraGryph software (Menges 2018). Spectra files were merged with nitrogen data (LECO or Thermo Scientific Flash 1112) for analysis. A Partial Least Squares Regression calibration model was developed using all U of A and U of S samples using the function *plsr* from the *pls* library (Mevik et al. 2018; R Development Core Team 2018). Spectra were visually inspected for any outliers (Appendix C - Figure 1).

A validation test was done to ensure that University of Alberta (U of A) data did not differ from a calibration model that was previously developed by the University of Saskatchewan (U of S).

A subset of 581 samples from the U of S and 69 samples from U of A with known protein concentrations were used for model validation. Thirty percent of U of A and U of S samples were randomly selected, removed, and labelled "Test" for validation of model prediction accuracy. The remaining samples were retained and labelled "Train". Visual data inspection show no deviation of U of A samples from U of S samples (Appendix C - Figure 2).

To develop a calibration model for the full dataset, 576 samples were randomly selected and the remaining samples were used for independent confirmation. Again, the calibration model was developed using the function *plsr* from the *pls* library (Mevik et al. 2018) with 10 components and Leave-One-Out cross validation. The Root Mean Squared Error of Prediction values (RMSEP) indicates that use of a 6-component model is preferred (Appendix C - Figure 3). The RMSEP is 1.285 for 6-components, which measures average difference between model predicted and estimated (LECO or FLASH) percent protein in model calibration samples. Further analysis was conducted using the 6-component model.

Protein concentration in the remaining samples was estimated using the 6-component PLSR model. Calibration data and confirmatory samples are illustrated in Appendix C - Figure 4. There was a strong correlation between predicted and measured protein. Although prediction was not perfect, samples were estimated within 1.3%. Study samples came from a diversity of heterogeneous plants communities and prediction accuracy remained consistent, therefore use of NIR seems a reasonable trade-off between cost and effort.

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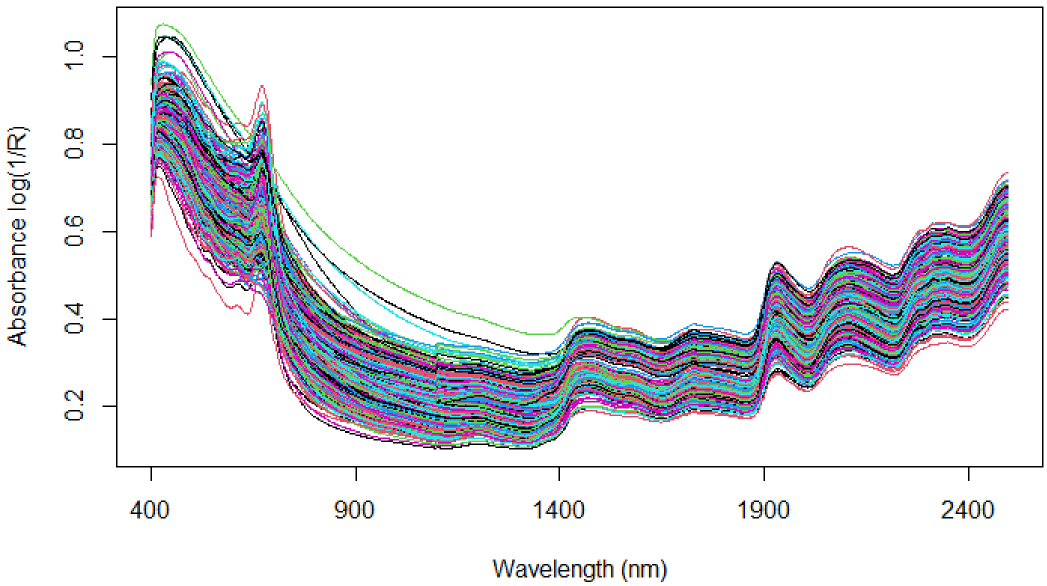
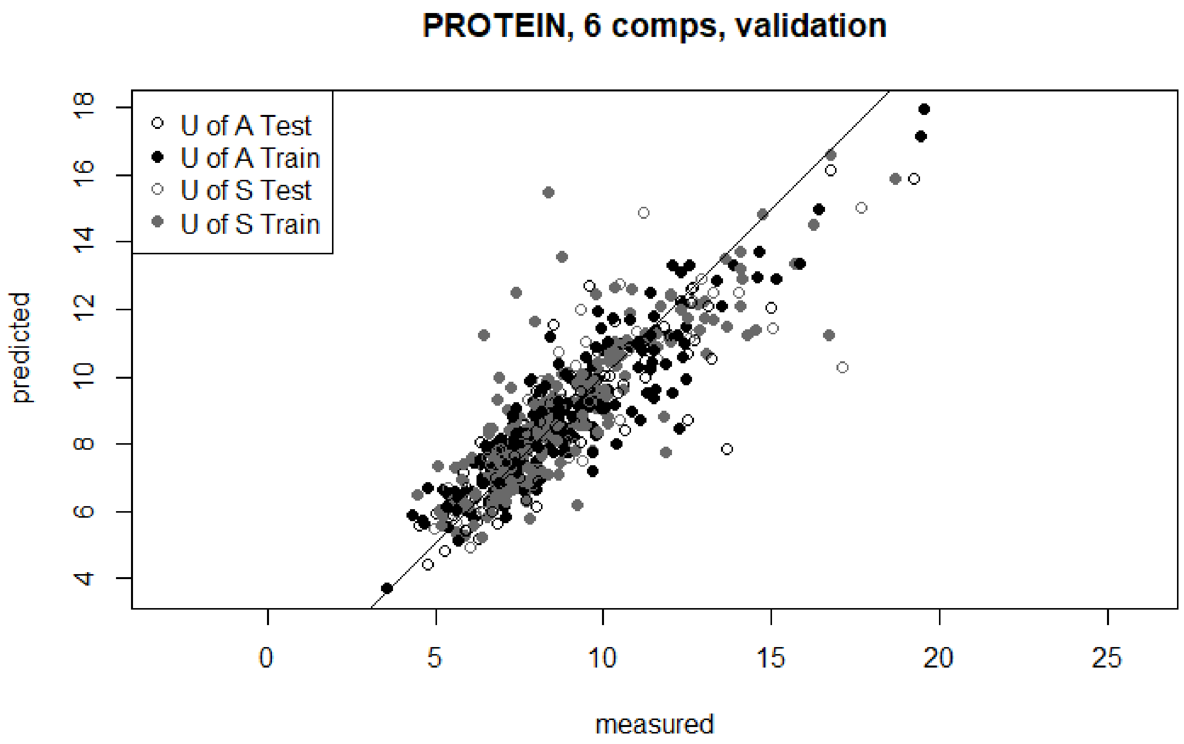
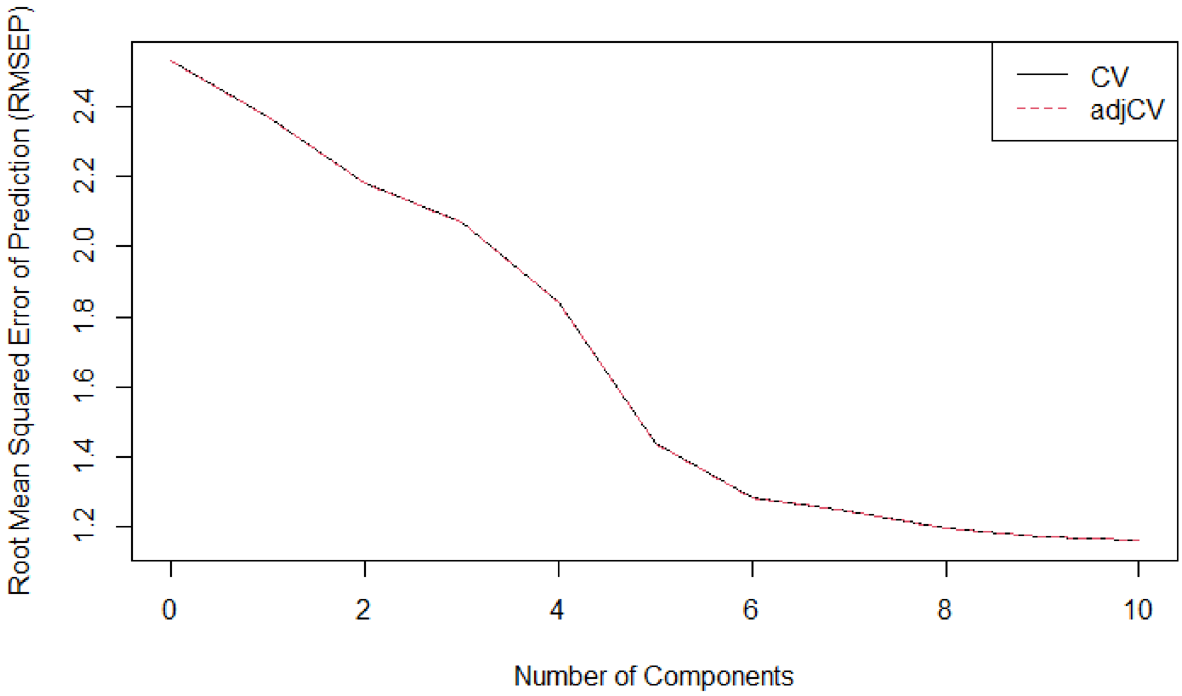


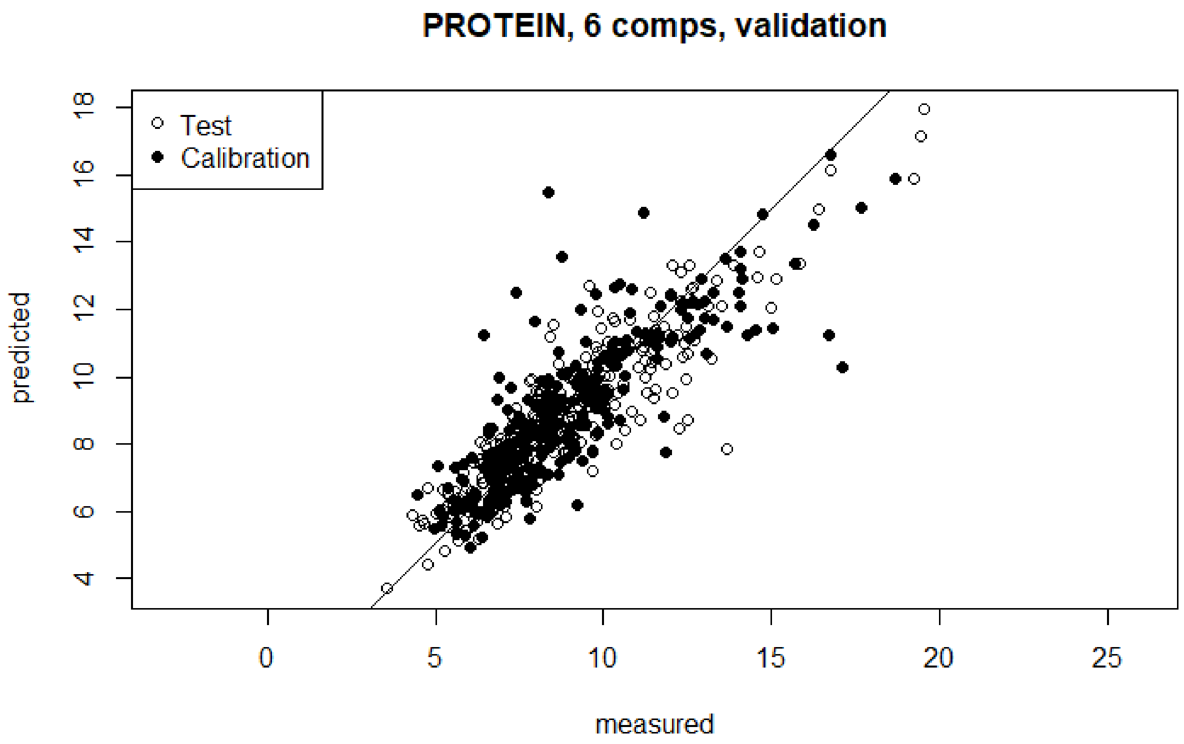
Appendix C - Figure 1. NIR absorbance spectra for 651 mixed native grassland samples.



Appendix C - Figure 2. Validation model for University of Alberta (U of A) and University of Saskatchewan (U of S) data. “Test” denotes samples with known protein concentrations that were removed from model to assess prediction accuracy, while “Train” denotes samples used for plsr calibration model development.



Appendix C - Figure 3. Calibration model performance (RMSEP) as a function of model components.



Appendix C - Figure 4. Measured protein (LECO or FLASH) and PLSR model-estimated protein for the calibration and test sample datasets. Test sample data points were not used in the development of the calibration model. The line indicates a 1:1 relationship between estimated and test-sample data.