ADAPTIVE MANAGEMENT AS A TOOL IN THE RESTORATION OF GRASSLAND, WETLAND, AND RIPARIAN ECOSYSTEMS WITHIN THE NORTHERN GREAT PLAINS

A Thesis
Submitted to the Graduate Faculty
of the
North Dakota State University
of Agriculture and Applied Science

By

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In Partial Fulfillment of the Requirements
for the Degree of
MASTER OF SCIENCE

Major Program:
Natural Resources Management

February 2022

Fargo, North Dakota

North Dakota State University
Graduate School

Title

Adaptive Management as a Tool in the Restoration of Grassland, Wetland, and Riparian Ecosystems within the Northern Great Plains

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MASTER OF SCIENCE

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ABSTRACT

Four individual restoration projects were conducted across grassland, wetland, and riparian ecosystems in the Northern Great Plains, with common themes of adaptive management and enhancing native plant species presence. The first project, a grassland restoration, assessed interseeding treatment combinations to evaluated influences on plant community composition. The second grassland restoration focused on revegetation efforts utilizing multiple seed mixes on a highly modified site and looked to understand influences on species establishment and invasive species control. A wetland restoration project was conducted employing varying levels of treatment intensities with goals of establishing native vegetation in an invasive dominated site. The last project evaluated the potential to use riparian grazing as a means of stream restoration with goals of increasing floodplain accessibility and stream stability. Given the essential ecosystem services each system provides, it is important to conduct restoration studies to understand mechanisms supporting the continued rehabilitation of degraded systems.

ACKNOWLEDGMENTS

I would like expression my most sincere gratitude to the following people and so many others who have supported me in my time at North Dakota State University.

To my advisors: Dr. Miranda Meehan, thank you for the countless hours you have spent teaching, guiding, and supporting me. I have learned so much from you and will always be grateful for the lessons you have taught me. You have helped me grow both as an academic and as an individual. Dr. Edward S. DeKeyser, I incredibly grateful for your patience, wisdom, and inspiration. Your never-ending passion and energy made long days in the field even more enjoyable. Thank you for pushing me, believing in me, and helping me see life and science in new lenses.

To my committee members: Dr. Peter O'Brien, thank you for your time, vison, and encouragement. I am grateful for your guidance and perspectives in navigating the wonderful world of statistics and life beyond graduate school. Dr. Caley Gasch, thank you for your unwavering enthusiasm, knowledge, and outlook on traversing graduate school. The beautiful community you have helped create has been a wonderful addition to my graduate experience.

Dr. Breanna Paradeis Kobiela, I am deeply appreciative of your support, guidance, and advice on research, data analysis, and writing. I am grateful for the time you put into helping me succeed. This thesis would not be where it is without you.

Thank you, Stacy Swenson and the US Forest Service, for spending their time with us on the prairie, proving guidance on grazing practices, and aiding in the implementation of our riparian grazing study.

I would also to express my sincerest appreciation for the immense impact the Hatch Fund, Ekre Grassland Foundation, Cross Ranch Fellowship, and Post Eligibility scholarship has had on my graduate research and education.

To my fellow graduate students, I will always be grateful for the time we have spent together both in and out of the field. The long hours we spent clipping grass will not be forgotten. Lastly, thank you to my family and friends for the endless love and support. You have helped me more than you know and I will be forever grateful.

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LIST OF ABBREVIATIONS

ANOVA ....Analysis of Variance.

AUD..Animal Unit Days.

AUE ....Animal Unit Equivalents.

BHR ....Bank Height Ratio.

GCS...Camp Grafton South.

DPG ....Dakota Prairie Grasslands.

ESD...Ecological Site Descriptions.

ER ....Entrenchment Ratio.

GPS ....Global Positioning System.

GE ....Grazing Exclusion.

HI ....High Intensity Short Duration Grazing.

LPI...Line Point Intercept.

MLRA ....Major Land Resource Area.

MWR..Meander-Width Ratio.

MANOVA...Multivariate Analysis of Variance.

NEPA ....National Environmental Policy Act.

NWG..Native Warm-Season Grass.

NMS...Non-metric Multidimensional Scaling.

PCC ....Plant Community Components.

PPR ....Prairie Pothole Region.

PCA..Principal Components Analysis.

RCESD...Riparian Complex Ecological Site Descriptions.

RCS ....Rosgen Stream Classification System.

RG ....Rotational Grazing.

STM ....State-and-Transition Models.

HSD ....Tukey's Honesty Significant Difference.

VW..Valley Width.

WDR ....Width-to-Depth Ratio.

GENERAL INTRODUCTION

Ecological restoration looks to rectify negative disturbances through the promotion and reestablishment of diverse ecosystems capable of functioning without human intervention along a natural successional pathway (Aaland 2010; Howell et al. 2012; Biró et al. 2019; Gann et al. 2019). Goals of ecological restoration center around bringing sites to their most productive and natural state, a position allowing the site to provide the greatest number of ecosystem services. Ecological restoration is not a recreation of the historic conditions but the movement towards a dynamic system free from human constraints or at least capable of handling human influence through increased resilience and stability (Howell et al. 2012).

Across the world, reductions are being seen in species diversity, available habitat niches, and ecosystem services, prompting the need to restore altered ecosystems (Rayburn and Laca 2013). Restoring human altered ecosystems cut off from their disturbance regimes pose difficult management strategy questions (Dornbusch et al. 2018). A large portion of environmental degradation originates from land use practices for food production for both human and livestock consumption (Hilderbrand et al. 2005). These alterations can have unintended and detrimental consequences towards a decline in valuable ecosystem services.

Grasslands within the Midwest have seen mass alterations (Wick et al. 2016; Liu et al. 2019; Burke et al. 2020; Grant et al. 2020) and are often highly fragmented (Jackson 1999). Upland sites dominated by invasive species, such as *Bromus inermis* (smooth brome) and *Poa pratensis* (Kentucky bluegrass) have the potential to see alterations in litter coverage, soil characteristics, vegetation dynamics (Meehan et al. 2021), and genetic diversity (Jackson 1999). Reductions in streambank stability and supportive capacity for multiple organisms' habitats may occur within riparian systems surrounded by uplands containing a high percentage of invasive species (Hecker et al. 2019; Meehan et al. 2021). Additionally, the management of upland species can have an impact on water movement into wetland systems (Gleason et al. 2011). Across the United States, wetland coverage has been reduced to 53%, with riparian systems seeing losses up to 70% (Howell et al. 2012). Given the interconnection between upland and riparian sites, it is crucial to focus management efforts beyond the riparian system alone and carry out restoration in the upland as well (Gleason et al. 2011). Despite their minimal land coverage within rangelands, riparian areas play a central role in providing necessary ecosystem functions to landscapes (Gregory et al. 1991; Moerke and Lamberti 2004; Holland et al. 2005; Swanson et al. 2015). Further the restoration of wetlands can carry over into the upland system and increase the floristic quality of vegetation (Gleason et al. 2008). Supporting and restoring both upland and lowland systems across a landscape can encourage improved connections and increase system resilience (Downard et al. 2014).

Adaptive management supports land managers in accounting for the evolving nature of restoration sites through an interchangeable approach focused on reducing uncertainty (Kentula 2000; Birge et al. 2016; Yang et al. 2016; Ahlering et al. 2020). Adaptive actions are needed to consider the many variables present within a system which may affect restoration treatments (Kentula 2000; Sunding et al. 2004; Zedler and Kercher 2005; Matthews et al. 2020). This allows researchers and land managers to consider additional treatment combinations capable of bringing forward greater successes (Kentula 2000; Birge et al. 2016; Ahlering et al. 2020). Contrary to a passive approach, an adaptive and/or intensive management plan allows researchers and land managers to determine the best course of action in real time. The results from research employing adaptive management will aid in future land management decisions by providing an analysis of successes and failures of restoration methods (Kentula 2000; U.S. Fish and Wildlife

Service 2020; McCauley et al. 2019). Challenges presented and strategies employed will further the capacity of future researchers to develop their experiments based on a greater understanding of best management practices (Kentula 2000; McCauley et al. 2019). Despite the benefits, limited studies within the natural resource management field have demonstrated the implications of using adaptive management (Ahlering et al. 2020).

To evaluate the impacts of restoration treatments paired with adaptive management, four individual research projects were conducted within the Northern Great Plains and are outline in the following chapters. Restoration projects were carried out across grassland, wetland, and riparian ecosystems all within agricultural settings. Given the essential ecosystem services each system provides, it is important to conduct restoration studies to understand mechanisms that support the continued rehabilitation of degraded systems. Goals of restoration center around returning sites to their most productive and natural state, a position allowing the site to provide the greatest number of ecosystem services.

The first grassland restoration project is based on a site highly invaded by cool season invasive grasses, with the site no longer producing to its greatest potential. This research project studies the ability of herbicide, interseeding, grazing, and burning to reduce the competitive advantages of Kentucky bluegrass and smooth brome. The study evaluates the impacts to vegetation biomass and species richness between different treatment applications to determine increases or reductions in both native warm season grasses and cool season invasive grasses.

The second grassland restoration project is located at a degraded military training site impacted by mass soil removal. The study site is currently undergoing restoration efforts to improve soil health and native species richness, while also decreasing invasive species richness with a specific adaptive management focus on *Euphorbia esula* (leafy spurge). This research looks to explore methods for native species establishment and noxious weed control by using varied treatment combinations of topsoil additions, seeding, and herbicide applications. Identifying successful treatment combinations will direct future land management decisions and support in developing recommendations for ranchers to improve the productivity of desirable forage species within degraded grasslands.

The next ecosystem being studied is a wetland previously managed as cropland. This study looks to analyze the actual cost of various plant community restoration methods within wetland ecosystems and the associated economics of treatments. The information collected provides details on early successional alterations to the wetland plant communities following restoration treatments. Treatments began in 2013, with the study site receiving treatment applications of seed only, seed and mulch, or seed, mulch, and transplanted soil plugs. Assessments of the study site are ongoing and through an adaptive management approach, post restoration treatments will be applied to the study site including prescribed burns, grazing, and herbicide applications. Future analyses will provide more insight on factors driving species composition within the study site and allow for an analysis on components influencing native species success over the full study duration.

Lastly, to understand the relationship between riparian ecosystems and livestock grazing a study was designed to assess how different riparian grazing treatments influence the health of three intermittent prairie streams. Riparian grazing has long been a controversial topic with few studies depicting its potential success towards the improvement of riparian systems in the form of stabilized banks, enhanced riparian vegetation, and increased benefits towards ranchers (Clary and Kinney 2002; Lucas et al. 2004; Agouridis et al. 2005). The study compared the effect of high intensity short duration grazing, rotational grazing, and grazing exclusion on vegetation cover and stream morphology. Results from this study will aid in the development of land management strategies to be considered for riparian restoration methods within ecosystems with grazing as a primary land use.

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CHAPTER 1: USE OF HERBICIDE, PRESCRIBED BURNS, INTERSEEDING, AND GRAZING AS TOOLS IN AN ADAPTIVELY MANAGED GRASSLAND RESTORATION

Abstract

Native grasslands within the Northern Great Plains are facing detrimental homogenization due to invasions by Kentucky bluegrass (*Poa pratensis*) and smooth brome (*Bromus inermis*). To evaluate management strategies capable of decreasing the competitive advantages of Kentucky bluegrass and smooth brome, while also improving site biodiversity and production, a randomized complete block design study was conducted to evaluate the effects of interseeding, spring burning, and glyphosate applications. The experimental design consisted of five treatments, 1) control (no treatment applications), 2) native species interseeding, 3) spring burn prior to native species interseeding, 4) glyphosate application prior to native species interseeding, and 5) spring burn plus glyphosate applications prior to native species interseeding. Following treatment applications in 2010, half the treatments were twice-over summer rotationally grazed annually from 2011 to 2013, until grazing was deemed to have no significant impact on seedling growth, and the full study site was opened to twice-over summer rotational grazing in 2014. An additional prescribed burn was applied across all treatments during the spring of 2020, in an adaptive effort to further control both target invasive species. In 2020 and 2021, biomass samples were collected through clipping eight 0.25 m^2 quadrats per treatment to assess influences on native species biomass, diversity, and richness, as well as Kentucky bluegrass and smooth brome biomass production. Analysis of variance (ANOVA) indicated between 2020 and 2021, there was a decrease in total biomass and native warm-season biomass. However, this may be attributed to the extended drought present during the 2021 sampling

period and not the additional prescribed burn. The spring burn plus glyphosate application prior to native species interseeding significantly decreased Kentucky bluegrass and smooth brome biomass production in comparison to the other treatments. Research efforts will continue to evaluate post management implications for future grassland restoration projects.

Literature Review

Impacts to Native Grasslands

Healthy native grasslands provide numerous benefits to land managers and surrounding systems. Benefits can include erosion control, filtration, food security, and carbon sequestration (Wick et al. 2016; Liu et al. 2019; Blackburn et al. 2020). However, across the globe, grasslands are at risk, facing increasing alterations away from their natural states (Wick et al. 2016; Liu et al. 2019; Burke et al. 2020; Grant et al. 2020). Within the United States, grasslands in the Great Plains have faced the most impacts, primarily through conversions to agricultural lands due to the high productivity of their soils (Masters et al. 1996; Wick et al. 2016; Liu et al. 2019; Burke et al. 2020; Grant et al. 2020). It is estimated 82% to 99% of native grasslands have been altered to other uses (Blackburn et al. 2020). Specifically, the area of tallgrass prairie has been reduced by 96.8% (Wick et al. 2016). Within the Northern Great Plains, estimates of grassland losses are roughly 70%, with land development and invasive nonnative species invasions being cited as the most substantial factors in this reduction (Palit et al. 2021).

Prominent impacts to grasslands began upon Euro-American entry to the region, with continued impacts carrying over into modern day (Grant et al. 2020). Grassland biodiversity has been hindered due to increases in urban/suburban sprawl, energy demand, invasive species introductions, alterations to natural fire regimes, and poorly managed livestock (Masters et al. 1996; Wick et al. 2016; Blackburn et al. 2020; Grant et al. 2020). Humans have brought many species across the globe to places outside their natural habitats with hopes of supporting lucrative agriculturally productive species (van Kleunen et al. 2020).

Alien species supported through breeding efforts are overrepresented, with an 18% greater chance of successful biological invasion, compared to native species (van Kleunen et al. 2020). Species within the *Poaceae* family are more likely to find success outside of their natural ranges in comparison to other global seed plant families with a lower economic value and naturalization potential. Data collected from 2002 to 2006 indicated *Poa pratensis* (Kentucky bluegrass) made up 27% to 36% of vegetation within three of five National Wildlife Refuge Complexes in North Dakota and South Dakota (Grant et al. 2009). *Bromus inermis* (smooth brome), another common invasive species in the Northern Great Plains, was found to be present at a rate of 45% to 49% within two of five National Wildlife Refuge Complexes in North Dakota and South Dakota. Unnatural disruptions to disturbance regimes within grasslands result in declines in their ecosystem services leading to decreases in the quality of wildlife habitats, water and nutrient cycles, and species productivity (Wick et al. 2016; Grant et al. 2020). Reestablishing natural disturbance regimes to promote biodiverse native grasslands is in the interest of society and the ecosystems on which we rely.

Grassland Invasions

Limitations to grazing and natural fire regimes have altered grasslands, creating favorable conditions for invasive species, specifically Kentucky bluegrass and smooth brome (Dornbusch et al. 2020; Grant et al. 2020; Palit et al. 2021). This invasion has decreased grasslands floristic diversity, with conclusions from multiple studies indicating the limitation of grazing and natural fire regimes have in part, resulted in grassland ecosystems being broken down by Kentucky bluegrass and smooth brome (Larson et al. 2001; DeKeyser et al. 2009; Fink and Wilson 2011;

Sinkins and Otfinowski 2012; DeKeyser et al. 2013; Grant et al. 2020). Kentucky bluegrass and smooth brome alter grassland systems by replacing native species and changing natural processes, such as plant-soil feedbacks, thus increasing their chance of survival (Grant et al. 2020; Palit et al. 2021). Kentucky bluegrass and smooth brome find survival success in large part based in their ability to begin growing before most native species (DeKeyser et al. 2013, DeKeyser et al. 2015). Both Kentucky bluegrass and smooth brome can develop dominance at an earlier time frame, in turn stifling competition from native species (DeKeyser et al. 2015; Printz and Hendrickson 2015). The production of both invasive C3 grasses declines in times of low precipitation, whereas, pastures containing C4 grasses have greater drought tolerance (Hall et al. 1982; Jackson 1999). Grazing practices play a large role in the success of invasive species. Long term continuous grazing can decrease native plant cover, while Kentucky bluegrass can tolerate this grazing system (Jackson 1999). Unmanaged grasslands within the Great Plains can be overrun by the previously mentioned invasive species, and therefore require restoration treatments to protect grassland heterogeneity (Gasch et al. 2020).

Kentucky Bluegrass

Kentucky bluegrass has increased in ground coverage since 1978 (Dennhardt et al. 2021) and is a dominant invasive nonnative species within the Northern Great Plains (Cully et al. 2003; Grant et al. 2009; DeKeyser et al. 2013; DeKeyser et al. 2015; O'Brien 2014; Toledo et al. 2014; Dennhardt et al. 2021; Palit et al. 2021). Within North Dakota alone, Kentucky bluegrass makes up over 50% of assessed rangeland sites based on surveys performed for the National Resources Inventory (Toledo et al. 2014; Palit et al. 2021). This dominance is attributed to Kentucky bluegrass' fast rate of reproduction and development, strong presence in soil seedbanks, ability to alter plant-soil feedbacks, and capacity to develop densely rooted rhizomatous mats (Palit et al. 2021). Despite Kentucky bluegrass being an invasive nonnative species, it is still commonly accepted based on its forage value for livestock. Once native pastures become dominated by Kentucky bluegrass there are reductions and alterations away from complex plant community compositions. Rises in yearly rainfall and soil water ushered in by climate change result in prairies seeing a conversion from higher forb coverage, to instead, greater coverage of C3 grasses (Nie et al. 1992; Clark et al. 2002; Collins et al. 2012; Dennhardt et al. 2021).

Smooth Brome

Smooth brome also has become dominant across many native prairies, largely because of its value for forage and its ability to rapidly colonize (Hendrickson and Lund 2010; Salesman and Thomsen 2011; Slopek and Lamb 2017). Smooth brome can take over native species niches through its quick establishment of close-knit rhizomes and its ability to alter feedback cycles through changes in litter accumulation and nitrogen conversion processes (Piper et al. 2015; Slopek and Lamb 2017). Smooth brome can reduce vegetation biodiversity by as much as 70%, with impacts commonly noted in highly disturbed areas (Piper et al. 2015). Monocultures formed by smooth brome can diminish available forage (Hendrickson and Lund 2010), ecosystem services, and habitat for fauna dependent on biodiverse grassland ecosystems (Hendrickson and Lund 2010; Salesman and Thomsen 2011); therefore, prompting the need for restoration actions to take place (Hendrickson and Lund 2010; Salesman and Thomsen 2011; Piper et al. 2015).

Grassland Restoration

Given the loss of connectivity between native prairies within the Northern Great Plains, ecological restoration is a pivotal management action needed to ensure biota within these fragmented systems do not see declines in genetic diversity and subsequent reductions in biodiversity (Jackson 1999). Ecological restoration projects often center their goals around increasing site biodiversity and limiting the presence of introduced species. This unfortunately can be an extremely demanding goal given many sites are dominated by invasive species that have altered sites to the point where feedback loops favor invasive species (Sheley et al. 2010; Vinton and Goergen, 2006; Link et al. 2017). Correspondingly, restoration treatments must account for processes specific to the region to reinstate natural ecological functions (Link et al. 2017).

Improvements in native biodiversity is a common goal among grassland restoration projects (Jackson 1999; Corbin et al. 2004; Rook et al. 2011) given the value biodiversity has on improving ecosystem productivity (Tilman et al. 2006), function, and resiliency (Isbell et al. 2011; Tilman et al. 2014). Biodiversity may also have a positive connection to more resilient and economical biomass production and ecosystem services (Tilman et al. 2006). Biodiversity has an influence on the spread of plant seeds, pollination, biological control of weedy and pest species, the cycling of nutrients across living and non-living components, and soil properties (Díaz et al. 2006). Given the value biodiversity poses to society, Díaz et al. (2006) argues there are strong grounds for ensuring the biotic integrity of ecosystems are protected and/or restored. Ecological theory indicates ecosystems with greater numbers of species may be more resilient to environmental stressors such as droughts because even if less tolerant species are heavily impacted by environmental impacts, there are still many other species to fill their niches within their community. Lehman and Tilman (2000) found sites with high biodiversity impacted by fluctuating precipitation levels maintained more stable biomass outputs across the entire community in comparison to those with less biodiversity.

Interseeding

The effective restoration of degraded grasslands is largely influenced by the presence of native seed within the soil seedbank. Seedbanks lacking in native seed will have diminished potential of accomplishing desired outcomes (Foster and Tilman 2003; Clark et al. 2007). Seeding treatments can improve soil surface cover and native vegetation abundance (Brudvig et al. 2011; Stanley et al. 2011). Interseeding native species can be especially valuable because it is less invasive to soil than other seeding methods such as tillage, which may impact the present vegetation and soil more dramatically (Bailey and Martin 2007; Link et al. 2017). Timing can be a drawback to this method, given planted vegetation may not develop for multiple years because of competition with current vegetation. Comparisons by Jackson (1999) on degraded tallgrass prairies between drill seeding, broadcast seeding, and no seeding restoration treatments indicated sites with no seeding treatments were almost entirely devoid of native species and sites drill seeded following herbicide applications had higher native species. Sites drill seeded followed by livestock grazing were shown to produce better than those utilizing broadcast seeding and subsequent livestock grazing.

Herbicide

The restoration of isolated grassland sites dominated by introduced species will benefit from herbicide applications prior to seeding treatments (Samson and Moser 1982; Waller and Schmidt 1983; Jackson 1999; Endress et al. 2012). Bahm et al. (2011) found herbicide to be a useful tool in limiting the cover of both smooth brome and Kentucky bluegrass, especially when paired with other treatments. Link et al. (2017) indicates the value of herbicide treatments paired with prescribed burns, seeding, and grazing as an effective management combination. Aronson and Galatowitsch (2008) attest to the value of herbicide applications as a necessary first step in controlling introduced species. This method of restoration paired with grazing and seeding treatments can support an increased transitional time period towards reaching restoration goals within sites used for cropping (Taylor et al 2013). However, the value of herbicide applications as a restoration treatment can be highly dependent on the current state of the land and level of invasion.

Reinstating Historic Fire and Grazing Regimes

Fire and grazing play a key role in the management of grassland systems. Historically grazing and natural fire regimes were present in the Great Plains (Masters et al. 1996; Gates et al. 2017; Blackburn et al. 2020; Dornbusch et al. 2020; Grant et al. 2020; Palit et al. 2021). Historic regimes began to be altered in the early 1900s when grasslands were converted in masses to provide for growing agricultural demands (Grant et al. 2020). Land managers at this time and continuing to present day, have prevented and/or altered the natural disturbance regimes of grasslands (Wick et al. 2016; Grant et al. 2020). As a result, grasslands have not been provided the natural disturbances they need to promote the survival and growth of native species.

Grazing influences on grasslands can be predominately attributed to the frequency and intensity of the grazing (Launchbaugh 2003). Resting or overgrazing native grasslands can result in the invasion of Kentucky bluegrass and smooth brome (Launchbaugh 2003; Grant et al. 2020). Therefore, when employing grazing for restoration a balance must be found between overgrazing and not grazing enough to support site biodiversity. Ranchers who neglect to account for proper stocking rates and species composition tend to have land overrun by invasive species leading to a degraded and less productive site (Delaney et al. 2016). Grazing practices must account for the ability of smooth brome and Kentucky bluegrass to find success in grazing systems without the use of prescribed burns (DeKeyser et al. 2013). Dornbusch et al. (2020) states full season grazing practices result in greater amounts of smooth brome, however, early season (early May to early June) high intensity grazing paired with patch burns were able to control smooth brome levels, while increasing native species cover. Given the nature for Kentucky bluegrass and smooth brome to initiate growth prior to native species, a late spring grazing plan may limit the invasive C3 grass' success based on their appeal to livestock (Dornbusch et al. 2020; Grant et al. 2020). Targeted grazing of invasive C3 grasses within northern tallgrass prairies by locating grazers in plots dominated by actively growing C3 grasses can diminish the competitive advantages of these grasses and support increases in plant species diversity (Smart et al. 2013). Grazing systems often work best when vegetation is given time to rest following defoliation, as certain species decrease after high grazing pressure (Jackson 1999). Proper grazing practices can improve biodiversity, water infiltration into soil (Toledo et al. 2014), nutrient cycling (Greeenwood and McKenzie 2001), microbial activity, and plant regrowth potential through increased root-to-shoot biomass rations (Anderson 2011).

Applications of prescribed burns should be carried out in conjunction with grazing to continue to support habitat diversity while accounting for invasive species (Jackson 1999; Burke et al. 2020; Dornbusch et al. 2020). Fire can be utilized to change vegetation composition, development, and production, along with altering nutrient cycles, microbial functioning, and hydrology (Knapp et al. 1984). Improvements in vegetation development and production can be attributed to decreases in litter leading to increased available nutrients and light penetration (Sharrow and Wright 1977). Dornbusch et al. (2020) recommends carrying out grazing following burn treatments to limit the ability of smooth brome to compete against resident native species.

Control methods for Kentucky bluegrass often incorporate applications of prescribed burns to reduce the species' spread, however, timing of burns is a key element in the treatment's effectiveness (Kral et al. 2018; Dennhardt et al. 2021). Prescribed burns taking place outside of the active growing season support declines in Kentucky bluegrass spread for up to three years following the application (Kral et al. 2018), with further research indicating the value of burning consistently in increments of two years (Li et al. 2013). Applications of prescribed burns should be carried out in accordance with grazing, to continue to support habitat diversity while still accounting for invasive species (Burke et al. 2020; Dornbusch et al. 2020). By pairing prescribed patch burns with grazing, ranchers are able to support their desire to maintain a working site and also account for invasive species (Delaney et al. 2016). Burns are successful in increasing forage value and grazing potential (Gasch et al. 2020). Within the Great Plains livestock find native species such as *Andropogon gerardii* Vitman (Big bluestem) and *Sorghastrum nutans* (L.) Nash (Indiangrass) more palatable compared to invasive species (Burke et al. 2020).

Jackson (1999) assessed the potential for native grasslands to be restored by employing treatments of rotational grazing or no grazing, broadcast seeding or drilled seeding, prescribed burns, and glyphosate applications. Three years following treatment applications, Jackson (1999) found there were significant differences between broadcast seeded plots and drill seeded sites, with the drill seeded sites containing a greater percentage of native grasses. More recently, Leahy et al. (2020) conducted a 20-year tallgrass prairie restoration project in Missouri to limit the presence of a cool season invasive, *Schedonorus arundinaceus* (tall fescue). Treatments included burns, herbicide applications targeting invasive nonnatives, and seeding of native grassland species. Applied treatments led to increases in native plant species richness, cover, and abundance. Through careful and attentive management, livestock grazing, prescribed burns, herbicide applications, and seeding can be used to support the restoration of native grasslands (Delaney et al. 2016; Leahy et al. 2020).

Ahlering et al. (2020) conducted a study using adaptive management to analyze the effectiveness of different restoration treatments on tallgrass prairies. Following nine years of treatments and analysis, they found prescribed burning to be most impactful on improving low quality sites with increases in native species productivity over invasive species. This was compared between combinations of grazing, burning, and rest. Gasch et al. (2020) considered the potential to use adaptive management with similar treatments to Ahlering et al. (2020), involving a combination of burning and grazing to limit the presence of Kentucky bluegrass. Gasch et al.'s (2020) research displayed the value of using prescribed burns and grazing in conjunction or separately to control Kentucky bluegrass to raise native species abundance.

Methods

Given the pervasiveness of invasive nonnative species such as Kentucky bluegrass and smooth brome within many native grasslands, there is a need to provide land managers and ranchers strategies for site restoration. Strategies provided need to account for the reliance ranchers have on maintaining a working site throughout the restoration process, given the financial dependence connected to the managed site. The objective of this restoration project is to assess the long-term impact of various grassland restoration methods and the influence post management treatments have towards restoring native tallgrass prairie and controlling invasive species. By incorporating an adaptive post management approach into the project framework, treatments can be applied based on assessed site needs. This experiment will provide additional research on native grassland restoration, while assessing the potential to increase rancher incentives towards restoration actions through increased biomass production.

Response variables evaluated include total biomass, native warm-season grass (NWG) biomass, grass species richness, smooth brome biomass, Kentucky bluegrass biomass, and smooth brome and Kentucky bluegrass combined biomass. Evaluations of biomass data allow researchers to assess whether project goals were met by comparing current site conditions to prior production history as well as location specific biomass production standards found in Ecological Site Descriptions.

Study Site

The study site is a grassland located within the Albert K. Ekre Grassland Preserve in Richland County, North Dakota (46°32'31.31"N, 97°8'34.92"W). The study site falls within Major Land Resource Area (MLRA) 56, Red River Valley of the North (NRCS 2006). Over the last 30 years (1991-2020) the study region averaged an annual average temperature of 5.67°C, with the average maximum temperate being 11.83°C, and the average minimum temperature being -0.4°C based on climatic readings from McLeod 3E station (NOAA 2021). Average annual rainfall was 60.93 cm, with peak precipitation occurring during the summer in June. The study site received above average rainfall in 2020 from June to August, however, in 2021 the study site received close to average or below average rainfall from May until August when precipitation increased (Figure 1.1).

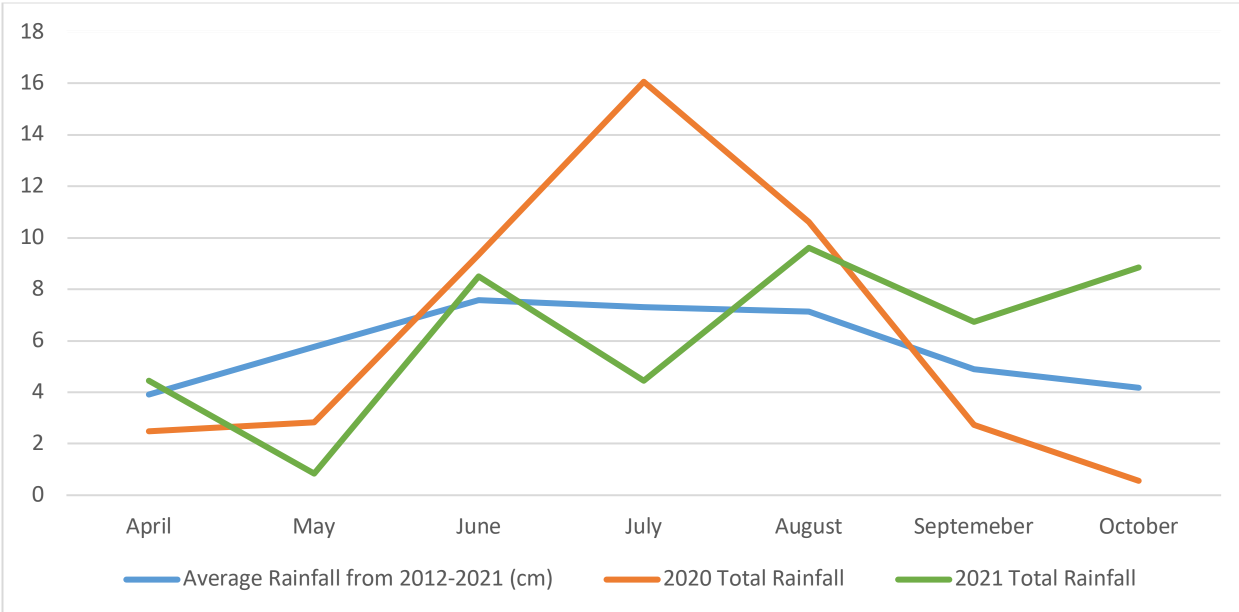

Figure 1.1: Monthly average rainfall (cm) across the full study duration (2012-2021) and monthly rainfall totals for 2020 and 2021 at the Ekre NDAWN Station.

Natural prairie vegetation within the region includes *Andripogon gerardii* (big bluestem), *Panicum virgatum* L. (switchgrass), *Sorghastrum nutans* (Indiangrass), and *Schizachyrium scoparium* (little bluestem). Soil orders are dominated by Mollisols and Vertisols with most of the region used for private cropland, which accounts for 79% of land use. This site's soils are composed of an Aylmer-Banty complex with slopes ranging from 0-6% (USDA-NRCS 2021). The Alymer soil is a sub irrigated sand classified as mixed, frigid Aquic Udipsamments with a moderate capacity for drainage. The Alymer soil originates from sand transformed by wind, spread across plains and delta plains (USDA-NRCS 2021). The Bantry soil is classified as mixed, frigid Typic Psammaquents and formed from windblown glaciofluvial deposits.

Restoration Project Description

The cumulative size of the study site is 12.1 ha, initially separated into two equally sized 6 ha plots, one grazed and one grazing excluded. The study site was previously used for cultivation purposes originating prior to the 1970's which included seeding practices for grasses

(Link et al. 2017). Following cultivation and seeding into tame grass, the land was grazed by cattle, with rotational grazing continuing up until 2010 prior to this study beginning. The site is presently considered to be a degraded pasture that was formerly tallgrass prairie. This takes into account the limited presence of warm-season tallgrass vegetation. The site is predominantly dominated by the invasive nonnative cool season grasses Kentucky bluegrass and smooth brome. The degraded status of the study site was partially assigned by a forage production analysis which determine forage production was falling short by 560kg/ha, proving to be the least productive grazing site within the Albert K. Ekre Grassland Preserve (Huffington 2011). Overall project goals are to assess the effectiveness of each treatment and their combinations to determine their impact on reestablishing native grasslands with native species while also accounting for the increasing presence of Kentucky bluegrass and smooth brome.

Initial evaluations of the study site began in 2012 with assessments predominantly focused on comparisons between current and pretreatment site conditions. This evaluation method persisted until 2020 when a prescribed burn was applied to the full study site. At this point, study site evaluations shifted to a shorter time scale to assess treatment impacts prior to the prescribed burn and following the prescribed burn. In 2021 biomass samples were collected following the 2020 prescribed burn and subsequent rotational grazing, with data analysis now focused predominantly on the 2020 and 2021 data. This data analysis aimed to assess the interaction between a prescribed burn and rotational grazing. Study assessments have continued to evaluate biomass production throughout the full study duration (2012 – 2021), however, beginning in 2019 assessments were refined to focus more specifically on post management impacts towards improving native vegetation biomass production through the reinstatement of historic regimes.

This experiment used a split plot, randomized complete block design from 2010 until 2020, with the whole plot variables being grazing and year. This involved half the study site being rotationally grazed, with the second half excluded from grazing using fencing. Plots were designed with a distinct separation between grazing and grazing exclusion, with applications of 1) control (no treatment applications), 2) native species interseeding, 3) spring burn prior to native species interseeding, 4) glyphosate application prior to native species interseeding, and 5) spring burn plus glyphosate applications prior to native species interseeding randomly assigned to each block. In 2021, the split plot factor was dropped from the study design given the entire study site has been twice over rotational grazing since 2014 and there has continued to be no significant differences between the two whole plots. At this point the study utilizes a randomized complete block design with six blocks, each of which includes a randomly assigned 1) control (no treatment applications), 2) native species interseeding, 3) spring burn prior to native species interseeding, 4) glyphosate application prior to native species interseeding, and 5) spring burn plus glyphosate applications prior to native species interseeding treatment (Figure 1.2).

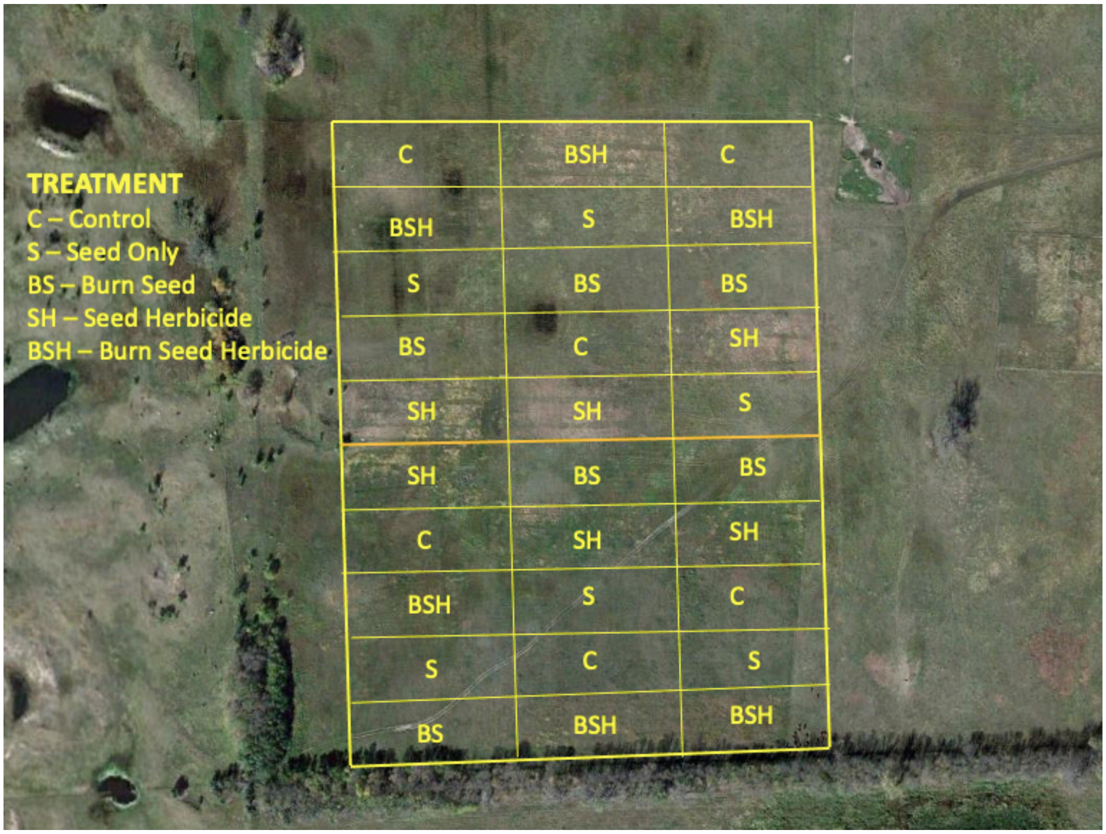

Figure 1.2: The study utilizes a randomized complete block design with six blocks. Treatments include control (C), seed (S), burn and seed (BS), seed and herbicide (SH), and burn, seed, and herbicide (BSH).

A Truax FLEX II drill model FLXII-818 was used to carry out the interseeding treatments. To decrease potential negative impacts to seed germination and seedling health, all burn and herbicide applications took place three weeks before interseeding. Initial burn treatments were applied as strip burns leading to predominantly head fires. A post management prescribed burn occurred in April of 2020, with burn applications carried out over the complete study site area. The burned was followed up with twice over rotational grazing as an additional post management treatment. Initial seeding treatments took place in July of 2010, excluding control plots. Seeding densities and the application procedures chosen were designed to account for the restoration of tallgrass prairie plant communities naturally present at the study site. Applications of seeding were spaced at intervals of 20 cm and had a seeding depth of 0.25 to

1.25 cm. The seed mix was attained from the Natural Resource Conservation Service- Ecological Site Description for sub irrigated and sands for Major Land Resource Areas (USDA-NRCS 2020). All species within the seed mix were native to the region and considered desirable forage for cattle (Table 1.1.). Millborn Seeds provided the seed mix for the study. To protect native seed establishment, rotational grazing was removed from the pasture prior to the seeding application and the site remained ungrazed until the following spring in 2011. Grazing applications on half the study site began again in mid-May of 2011 and routine twice over rotational grazing treatments continued through 2013, until the site was opened up fully to routine twice over rotational grazing in 2014. Herbicide applications used RoundUp® Concentrate Plus (The Scotts Company LLC, Worldwide Rights Reserved) mixed at a 60:1 ratio with water. A boom sprayer set to a rate of 23 L/ha was used to apply treatments.

Table 1.1: Species composition and seeding densities for interseeding seed mix.

Species	**Scientific Name[a]**	**Kg/Ha**
Big Bluestem – Bison	*Andropogen gerardii*	2.69
Prairie Sandreed – Goshen	*Calmolvilfa longifolia*	1.12
Canada Wildrye – Mandan	*Elymus canadensis*	0.56
Little Bluestem	*Schizachyrium scoparium*	0.56
Switchgrass – Dakota	*Panicum virgatum*	0.56
Western Wheatgrass – Rodan	*Pascopyrum smithii*	0.56
Sand Bluestem	*Andropogon hallii*	0.34
Green Needlegrass – Londorn	*Nassella viridula*	0.28
Indiangrass – Tomahawk	*Sorghastrum nutans*	0.28
Purple Prairie Clover	*Dalea purpurea*	0.28
White Prairie Clover	*Dalea candida*	0.28
Blue Grama – Bad River	*Bouteloua gracilis*	0.17
Prairie Cordgrass – Red River Germplasm	*Spartina pectinata*	0.17
Porcupine Grass – South Dakota Native Collection	*Hesperostipa spartea*	0.11
Prairie Junegrass	*Koeleria macrantha*	0.11

[a] Scientific names are from The PLANTS Database: USDA, NRCS. 2021. The PLANTS Database (http://plants.usda.gov, 21 December 2021).

Vegetation surveys have been conducted in 2012, 2013, 2014, 2015, 2019, 2020, and 2021 using 0.25m^2 quadrats. Within each block, eight quadrats per treatment were clipped.

Biomass within quadrats were clipped and individual grass species were separated into bags, with forbs, sedges, and shrubs being bagged as groups. Biomass clippings were then dried for a minimum of 72 hours at a temperature of 37.78°C prior to weight measurements being taken. Prior to analysis, all collected data points were modified to kg/ha and averaged within each replication.

A two-way factorial ANOVA and Tukey's Test were used to analyze sampled vegetation values and compare treatment means (SAS Enterprise Guide 7.1 (Copyright © 2017 by SAS Institute Inc. Cary, NC, USA)). A log transformation (log base 10) was performed to meet distributional assumptions for total biomass and Kentucky bluegrass total biomass. Despite smooth brome failing to meet distributional assumptions (Initial Skew Value = 1.338), a log application was not performed because this application increased skewedness. When smooth brome and Kentucky bluegrass were combined for their total biomass, a log transformation (log base 10) was performed to meet distributional assumptions. Log transformations were not performed on NWG and species richness because these variables met distributional assumptions. Dependent variables analyzed included total biomass, NWG biomass, grass species richness, smooth brome biomass, Kentucky bluegrass biomass, and smooth brome and Kentucky bluegrass combined biomass. Independent variable analyzed included block (one, two, three, four, five, and six), treatment (control, seed, burn + seed, seed + herbicide, and burn + seed + herbicide), year (2020 and 2021), and their interactions.

Results

The ANOVA indicated, based on data from 2020 and 2021, average total biomass responded to treatment ($F_{14,45} = 6.48$, $P = 0.0003$) (Figure 1.3) and year ($F_{14,45} = 58.83$, $P = < 0.0001$) (Figure 1.4). Seed + herbicide yielded the greatest average total biomass (mean =

4,075.60 kg/ha ± standard error) (Figure 1.3) compared to all other treatments. Seed only, burn + seed, and burn + seed + herbicide treatments showed no marked difference when compared to the control. Average total biomass was higher in 2020 (mean = 3,745.50 kg/ha ± standard error) than 2021 (mean = 2,354.20 kg/ha ± standard error) (Figure 1.4).

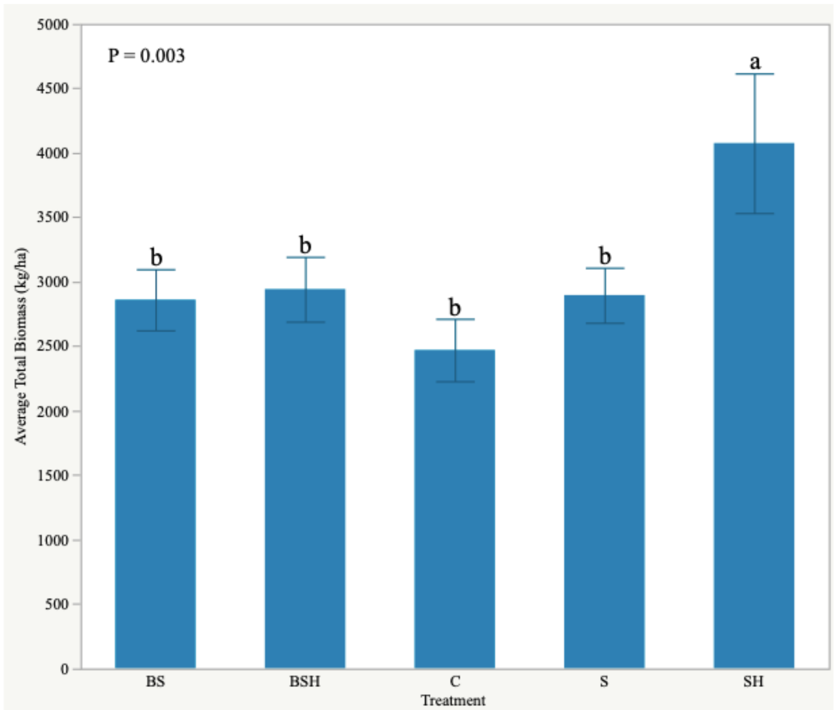

Figure 1.3: Average total biomass (kg/ha) (± standard error) by treatment for 2020 and 2021. The seed + herbicide treatment produced significantly more biomass than the seed only, burn + seed, burn + seed + herbicide, and control treatments. A statistical difference at α = 0.05 is indicated by different letters.

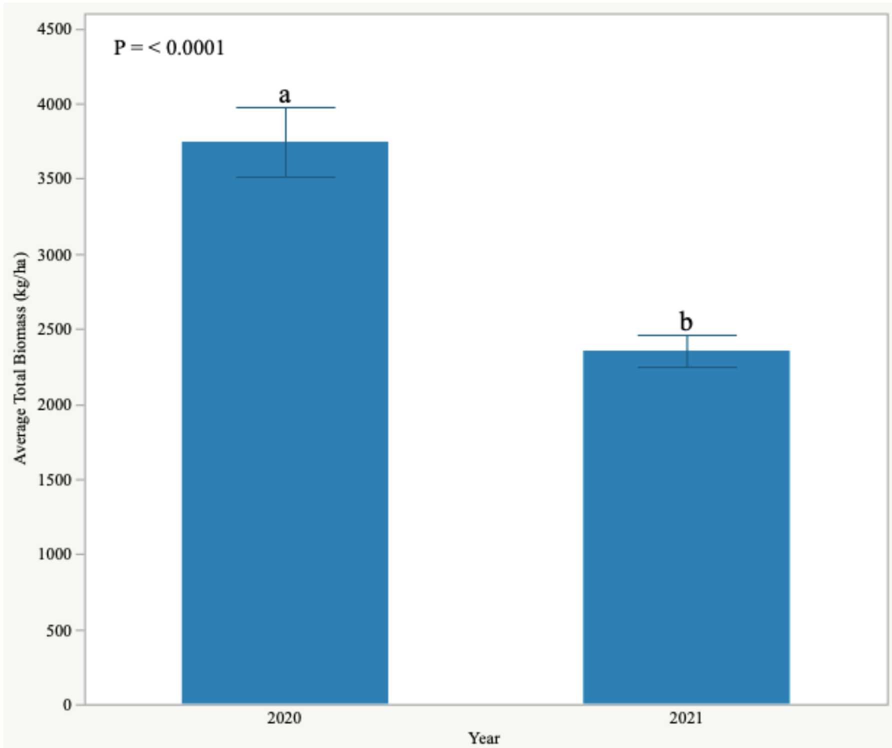

Figure 1.4: Average total biomass (kg/ha) (± standard error) by year. Significantly more average total biomass was produced in 2020 compared to 2021. A statistical difference at α = 0.05 is indicated by different letters.

Total NWG average biomass was greater in treated plots (seed only, burn + seed, seed + herbicide, and burn + seed + herbicide) compared to control plots ($F_{14,45}$ = 29.11, P = < 0.0001) (Figure 1.5). Seed + herbicide (mean = 2,606.60 kg/ha ± standard error) had significantly greater NWG average biomass compared to burn + seed (mean = 1,835.20 kg/ha ± standard error) and seed only (mean = 1,765.30 kg/ha ± standard error) (Figure 1.5). Total NWG average biomass responded differently across years ($F_{14,45}$ = 56.01, P = < 0.0001) with 2020 NWG average biomass (mean = 2,265.535 kg/ha ± standard error) being greater than 2021 NWG average biomass (mean = 1,212.725 kg/ha ± standard error) (Figure 1.6). There was also a significant interaction term (Treatment x Year) ($F_{14,45}$ =7.57, P = < 0.0001).

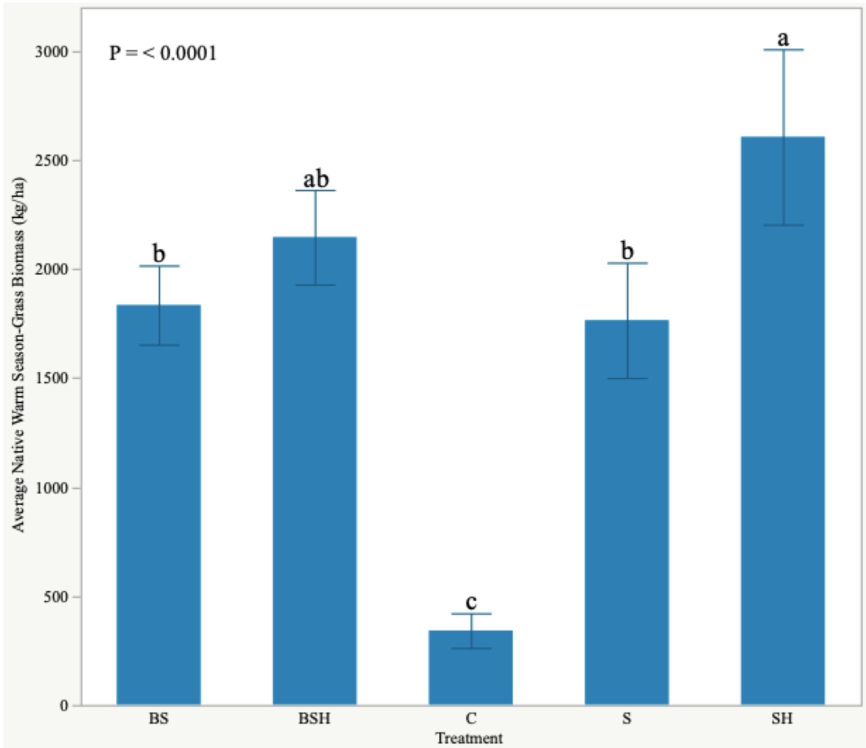

Figure 1.5: Average native warm-season biomass (kg/ha) (± standard error) by treatment across years. The seed + herbicide treatment had significantly greater average NWG biomass production compared to the seed only, burn + seed, and control. All treatments had greater average NWG biomass than the control. A statistical difference at α = 0.05 is indicated by different letters.

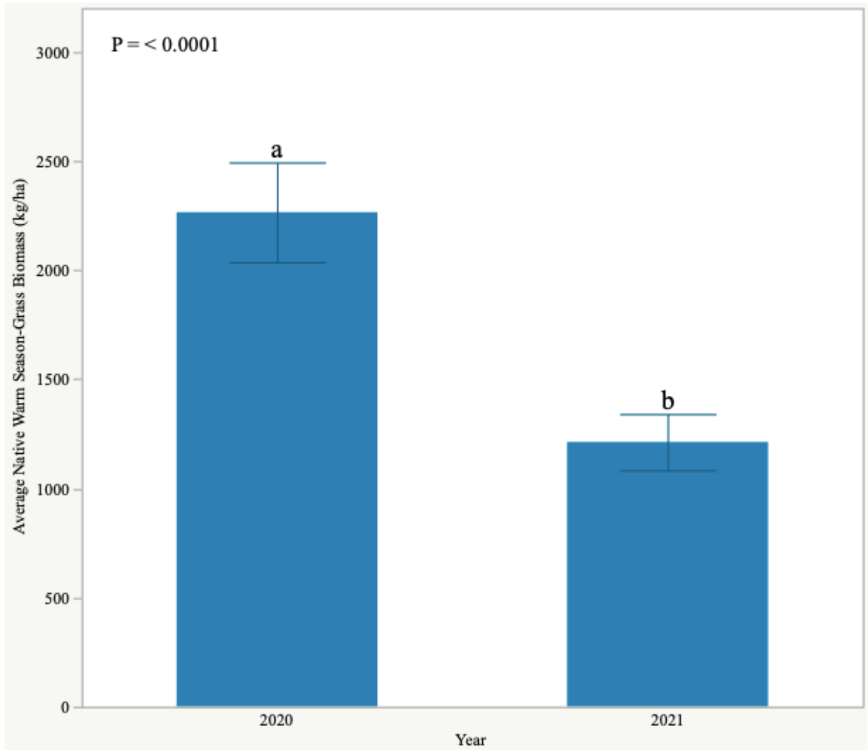

Figure 1.6: Average native warm-season biomass (kg/ha) (± standard error) by year. Significantly more average NWG biomass was produced in 2020 compared to 2021. A statistical difference at α = 0.05 is indicated by different letters.

Average grass species richness responded to treatment ($F_{14,15}$ =14.25, P = < 0.0001, Figure 1.7) and block ($F_{14,45}$ = 4.39, P = 0.0024) (Figure 1.8). Across the restoration, treatment applications yielded higher mean grass species richness when compared to the control (Figure 1.7). Mean grass species richness was not significantly different across restoration treatments through. Blocks three (mean = 5.275 grass species ± standard error) and six (mean = 5.025 grass species ± standard error) were significantly different from block 1 (mean = 3.9875 grass species ± standard error) (Figure 1.8).

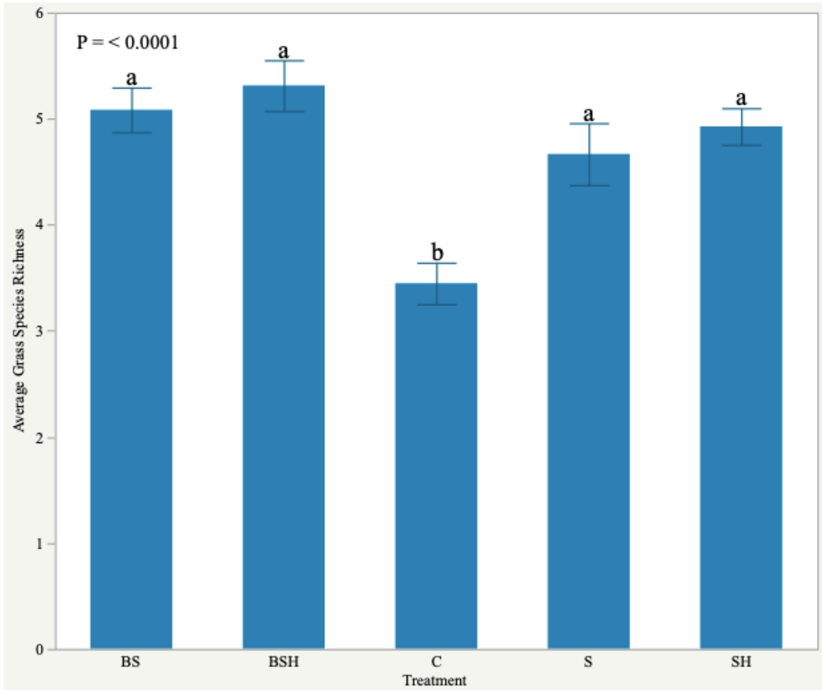

Figure 1.7: Average grass species richness (± standard error) by treatment. All treatments had significantly greater average grass species richness compared to the control. A statistical difference at α = 0.05 is indicated by different letters.

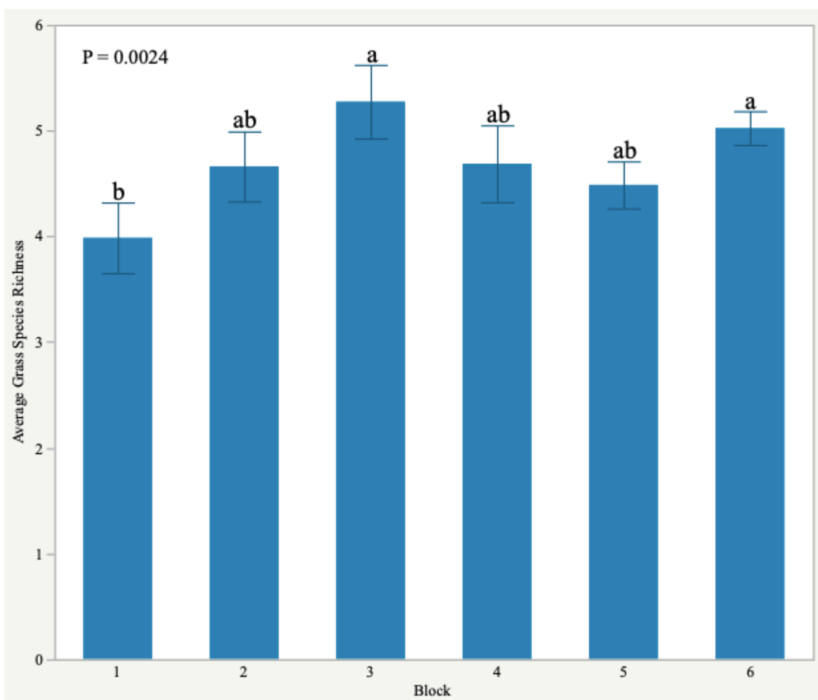

Figure 1.8: Average grass species richness (± standard error) by block. Blocks three and six produced more average grass species richness than block one. A statistical difference at α = 0.05 is indicated by different letters.

Smooth brome average total biomass was higher in control plots (mean = 514.70 kg/ha ± standard error, $F_{14,45} = 10.98$, P = < 0.0001) compared to all other treatments (Figure 1.9). Smooth brome average total biomass differed across blocks ($F_{14,45} = 3.68$, P = 0.007), with blocks one (mean = 344.30 kg/ha ± standard error) and two (mean = 358.70 kg/ha ± standard error) being significantly different from block four (mean = 125.40 kg/ha ± standard error) (Figure 1.10). The interaction between smooth brome biomass across years was not significant ($F_{14,45} = 0.38$, P = 0.8187).

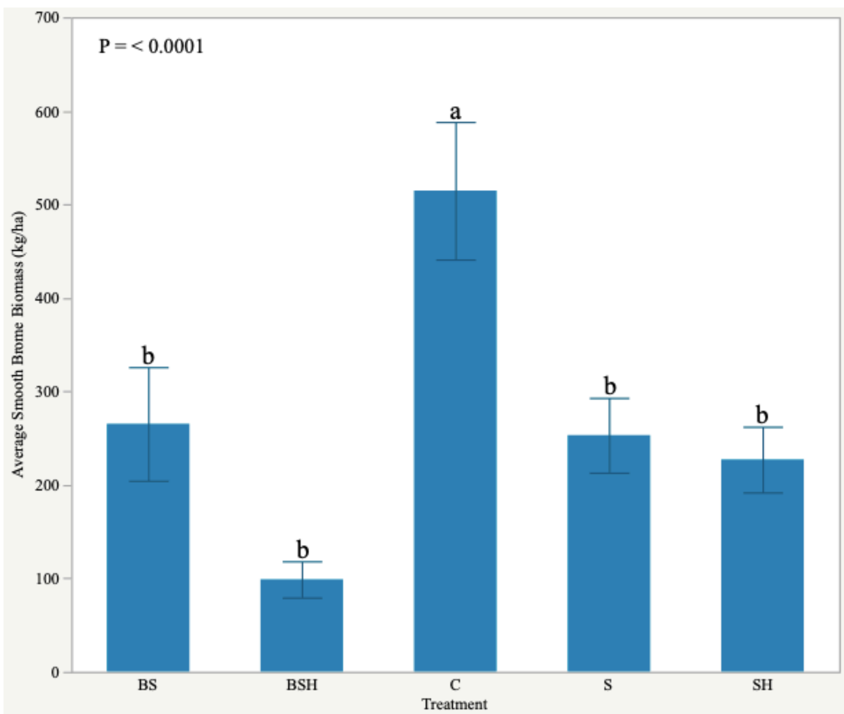

Figure 1.9: Average smooth brome biomass (kg/ha) (± standard error) by treatment. The control plots produced significantly greater average smooth brome biomass compared to all treatments. A statistical difference at α = 0.05 is indicated by different letters.

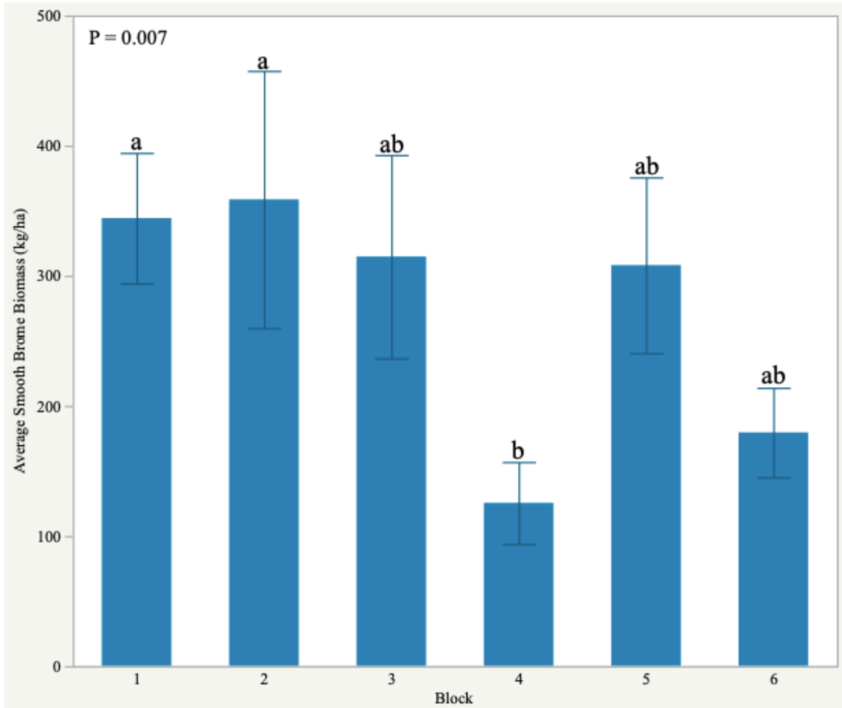

Figure 1.10: Average smooth brome biomass (kg/ha) (± standard error) by block. Blocks one and two produced more average smooth brome biomass than block four. A statistical difference at $\alpha = 0.05$ is indicated by different letters.

Kentucky bluegrass average biomass responded to treatment ($F_{14,45} = 8.9$, $P = < 0.0001$) (Figures 1.11) and block ($F_{14,45} = 4.09$, $P = 0.0038$) (Figure 1.12). Total Kentucky bluegrass biomass was higher in control plots (mean = 708.60 kg/ha ± standard error) in comparison to seed only (mean = 359.20 kg/ha ± standard error), burn + seed (mean = 393.10 kg/ha ± standard error), and burn + seed + herbicide (mean = 218.40 kg/ha ± standard error) treatments (Figure 1.11). Burn + seed + herbicide resulted in less Kentucky bluegrass average biomass compared to the control (mean = 708.60 kg/ha ± standard error) and seed + herbicide (mean = 423.50 kg/ha ± standard error) treatments (Figure 1.11). Across blocks, block one (mean = 641.58 kg/ha ± standard error) responded differently from blocks three (mean = 343.80 kg/ha ± standard error) and five (mean = 253.56 kg/ha ± standard error) (Figure 1.12).

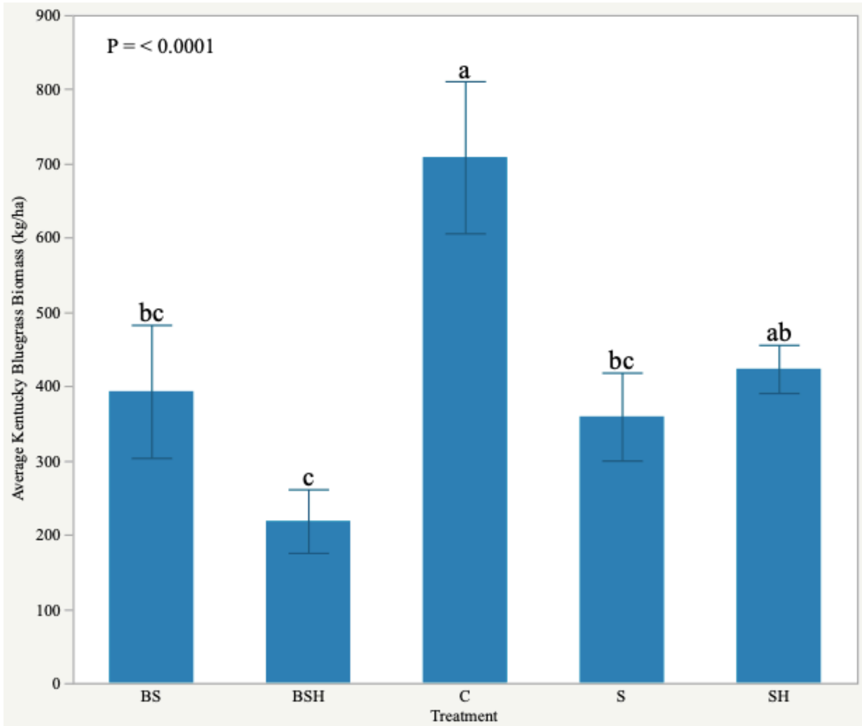

Figure 1.11: Average Kentucky bluegrass biomass (kg/ha) (± standard error) by treatment. Control plots produced significantly greater average Kentucky bluegrass biomass compared to the seed only, burn + seed, and burn + seed + herbicide treatments. The burn + seed + herbicide treatment produced less average Kentucky bluegrass biomass than the seed + herbicide treatment and the control. A statistical difference at $\alpha = 0.05$ is indicated by different letters.

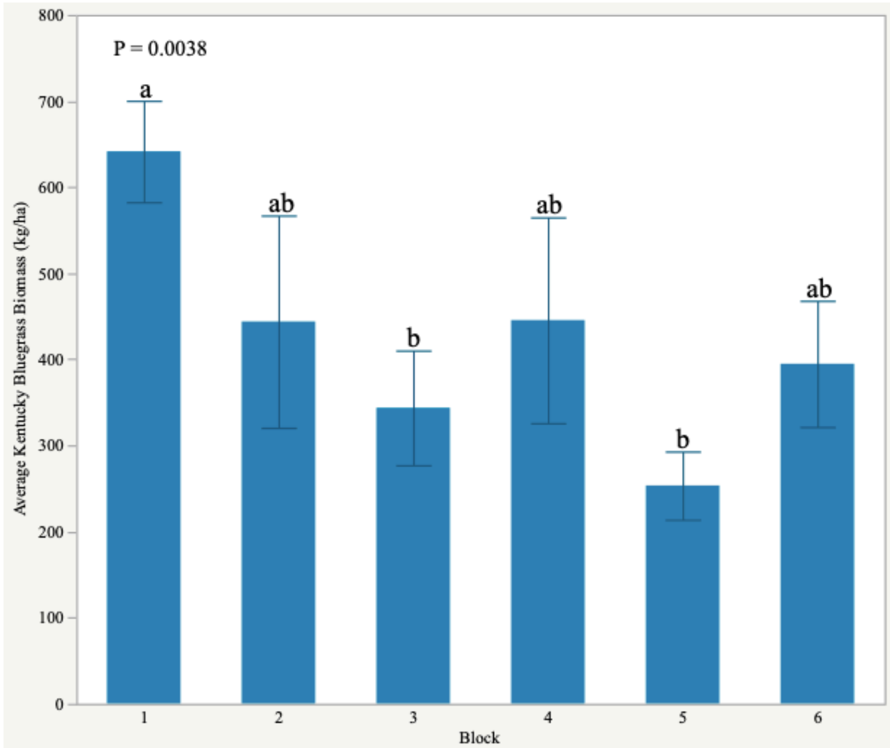

Figure 1.12: Average Kentucky bluegrass biomass (kg/ha) (± standard error) by block. Block one produced significantly more Kentucky bluegrass biomass than blocks three and five. A statistical difference at $\alpha = 0.05$ is indicated by different letters.

When smooth brome and Kentucky bluegrass biomass were combined, their total average biomass was significantly different across treatments ($F_{14,45} = 9.97$, $P = <0.0001$) (Figure 1.13) and blocks ($F_{14,45} = 2.67$, $P = 0.0338$) (Figure 1.14). When comparing the cool season grasses' biomass, burn + seed (mean = 658 kg/ha ± standard error), burn + seed + herbicide (mean = 317 kg/ha ± standard error), and seed only (mean = 612 kg/ha ± standard error) treatments resulted in the lower average combined smooth brome and Kentucky bluegrass biomass than the control (mean = 1,220 kg/ha ± standard error), with the burn + seed + herbicide treatment also having significantly lower biomass than the seed + herbicide (mean = 651 kg/ha ± standard error) and seed only (mean = 612 kg/ha ± standard error) treatments (Figure 1.13). Across blocks, block one (mean = 986 kg/ha ± standard error) contained higher combined smooth brome and Kentucky bluegrass average biomass than block five (mean = 562 kg/ha ± standard error) (Figure 1.14).

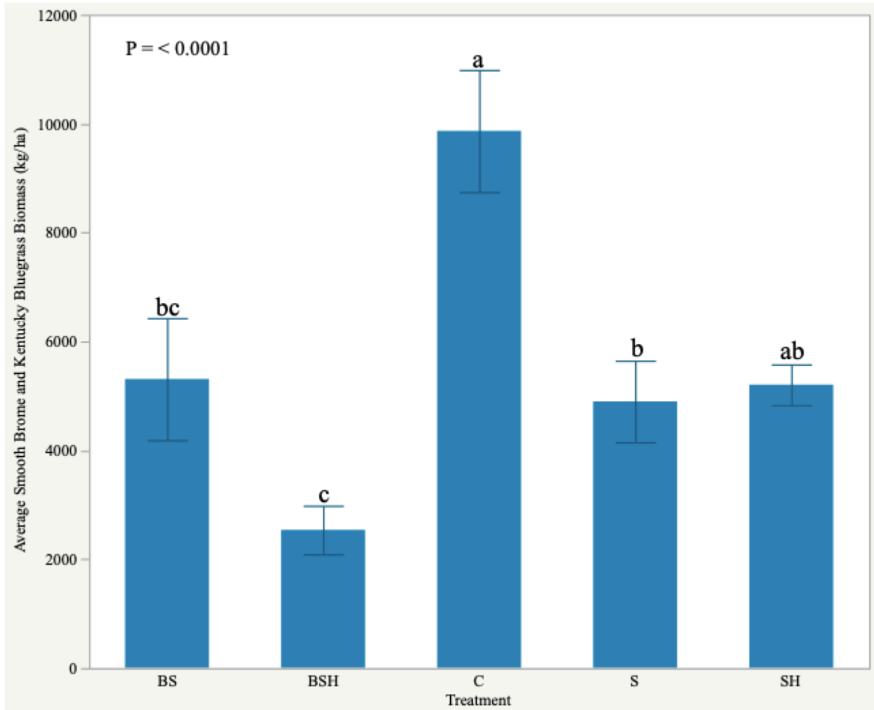

Figure 1.13: Average combined smooth brome and Kentucky bluegrass biomass (kg/ha) (± standard error) by treatment. The control had significantly greater smooth brome and Kentucky bluegrass biomass production than the seed only, burn + seed, and burn + seed + herbicide treatments. The burn + seed + herbicide treatment had lower production compared to the seed only, seed + herbicide, and the control. A statistical difference at α = 0.05 is indicated by different letters.

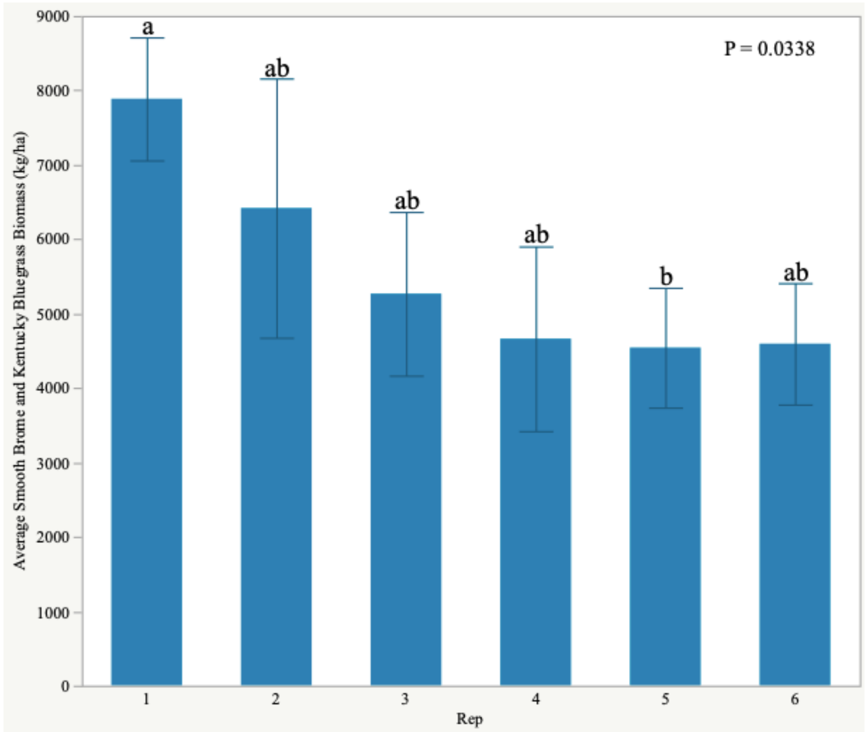

Figure 1.14: Average combined smooth brome and Kentucky bluegrass biomass (kg/ha) (± standard error) by block. Block one produced significantly greater smooth brome and Kentucky bluegrass biomass than block five. A statistical difference at α = 0.05 is indicated by different letters.

Discussion

Given the pervasiveness of invasive nonnative species such as Kentucky bluegrass and smooth brome within native grasslands (Grant et al. 2009; Toledo et al. 2014), control measures reducing the competitive advantages of these invasive species are needed (Link et al. 2017). This study looked to determine the impact interseeding treatments paired with combinations of spring burns and glyphosate applications, with the addition of post management treatments, have on controlling Kentucky bluegrass and smooth brome to increase native species biomass, diversity, and richness within a highly invaded grassland. It was hypothesized that increased incorporation of competition reduction measures would enhance native species establishment and production.

This hypothesis was further evaluated by applying post management treatments of a prescribed burn followed by rotational grazing.

The results of this study indicated interseeding paired with herbicide applications produced the greatest total biomass and average NWG biomass, however, it also resulted in greater combined Kentucky bluegrass and smooth brome biomass when compared to all treatments other than the control. The combination of interseeding with a spring burn and glyphosate application resulted in the lowest biomass of both target invasive species while also producing a similar amount of NWG biomass as the interseeding with herbicide treatment. This finding is supported by research that indicates combining a variety of different invasive species control measures can result in more success than a single method alone (Collins et al. 1998; Sheley et al. 2010; Taylor et al. 2013). Reductions in Kentucky bluegrass abundance have been noted following one year of burning (Kral et al. 2018), with Bahm et al. (2011) indicating the value of herbicide for control of Kentucky bluegrass and smooth brome with the possibility to implement other post management control measures such as grazing or burns for restoration purposes. Variation in treatment impacts were noted between blocks, with block one producing greater Kentucky bluegrass and smooth brome biomass than block five and scoring lowest in grass species richness. The connection between higher invasive species biomass and lower grass species richness within this block can be attributed to the invasive species' ability to stifle native species through alterations in natural processes (Grant et al. 2020; Palit et al. 2021). This difference could be the result of natural variability in vegetation makeup and research plots (Stallman 2020).

Previous years of research from the Ekre grassland restoration have shown interseeding paired with herbicide applications result in the greatest seedling establishment and production of

NWG and total biomass (Huffington 2011; Link et al. 2017; Stallman 2020). This result remained consistent across the study duration, despite the large fluctuations in precipitation rates since the restoration project began. Contrary to previous results, the 2021 results display the spring burn plus glyphosate applications prior to native species interseeding treatment to be the most effective competition reduction measure in its ability to control Kentucky bluegrass and smooth brome. This result could be influenced by the post management treatments of a prescribed burn in 2020 followed up with twice over rotational grazing.

The pairing of prescribed burns with herbicide applications and interseeding was successful in reducing the prevalence of Kentucky bluegrass and smooth brome, despite many studies documenting the difficulty of finding restoration methods that are successful in controlling both invasive species at once (Murphy and Grant 2005; Hendrickson and Lund 2010; DeKeyser et al. 2013; DeKeyser et al. 2015; Link et al. 2017). Further, the control sites had the highest Kentucky bluegrass and smooth brome biomass and scored the lowest for NWG biomass and average grass species richness, validating the treatments applied. Similarly, Bakker et al. (2003) and Leahy et al. (2020) found introduced species control measures utilizing interseeding treatments had the most positive impact when employed in conjunction with other treatment methods. Endress et al. (2012) and Taylor et al. (2013) further validated the findings of this restoration by demonstrating the successful use of herbicide for the reduction of invasive species.

The extended drought in the Northern Great Plains influenced total biomass production and NWG biomass production within the study site, resulting in a decline between 2020 and 2021. These results echo points considered by Guo et al. (2012), Knapp (2015), and Dennhardt et al. (2021), detailing the impact raises in precipitation, as was seen in 2020, can have on improved grassland vegetation production, with periods of drought producing declines in vegetation cover

(Heitschmidt et al. 2005). Increased precipitation within the Northern Great Plains can result in alterations to plant community compositions (Dennhardt et al. 2021), with some studies connecting raises in Kentucky bluegrass to higher precipitation (Weaver 1954; Nie et al. 1992; Patton et al. 2007). Surprisingly, grassland biodiversity was not significantly different between 2020 and 2021. Grasslands effected by prolonged periods of low rainfall tend to see declines in biodiversity (Harrison et al. 2015), with the potential to experience plant community shifts towards greater invasive species dominance (Moran et al. 2014). Within the Northern Great Plains where cyclical periods of drought and high precipitation are historically present (van der Valk 2005), native grassland species adapted to this fluctuation may prevail over invasive plants (Weaver 1954; Dennhardt et al. 2021), giving a possible explanation for the consistent species richness level between 2020 and 2021.

Recovery of native grasslands can result in improved outputs during times of low precipitation because many NWGs are better equipped to withstand decreased available water (Jackson 1999). Sites higher in vegetation biodiversity are better equipped to utilize resources within a variety of different niches, supporting their ability to maintain a greater consistency in production during times of stress (Hooper 1998; Hooper et al. 2005; Tilman et al. 2006). Kentucky bluegrass can maintain palatability during the majority of the year, however, months with limited moisture content and above average temperatures cause a decline in the forage quality of Kentucky bluegrass in comparison to native species (Jackson 1999; Gasch et al. 2020). Correspondingly, NWG can better utilize belowground resources such as moisture based on their rooting depth and mass, therefore increasing their survival success (Biondi 2007; Daigh 2014; Link 2014).

Management Implications

The findings of this study highlight the value of employing multiple treatment methods in conjunction with an interseeding application when restoring grassland ecosystems. Eleven years following initial restoration treatments, the use of interseeding continues to perform better in grass species richness and NWG biomass across all treatments compared to control plots. Treatments of interseeding paired with herbicide applications either alone or in conjunction with a prescribed burn are most effective in increasing NWG biomass production. A spring burn plus glyphosate application prior to native species interseeding reduced combined Kentucky bluegrass and smooth brome biomass significantly more than all treatments other than the spring burn prior to native species interseeding treatment. The inclusion of an additional adaptively applied post management prescribed burn followed by rotational grazing may have aided in the reduction of both invasive species with findings from multiple studies indicating the value of applying prescribed burns and grazing for invasive species control (Ahlering et al. 2020; Dornbusch et al. 2020; Gasch et al. 2020). However, it should be noted that these results come following an extended drought which influenced vegetation production across the study site. Clark et al. (2002) and Heitschmidt et al. (2005) found following extended droughts forage biomass declines, with Heitschmidt et al. (2005) indicating biomass was reduced by 20 to 40% with the most declines in perennial C3 grasses following a spring drought.

Ecosystem components and services respond differently to management strategies and prevalent weather conditions making it important to reduce response uncertainties through applying an adaptive management approach (Delaney et al. 2016; Grant et al. 2020; Palit et al. 2021). The imposing and continual threat of climate change exacerbates the need to conserve and restore grassland ecosystems to promote improved biodiversity (Yang et al. 2019; Grant et al.

2020). Ecological communities high in biodiversity have improved yields (Tilman et al. 1996; Biondini 2007), carbon capture levels (Yang et al. 2019), tolerance to disturbance (Biondini 2007), and resistance to invasion species (Biondini 2007; Tilman et al. 2014). The results of this grassland restoration project demonstrate restoration efforts have a positive impact on vegetation biomass production and species richness, both of which are valuable assets to producers and society as a whole. Further research is needed to evaluate the impact post management prescribed burns and grazing regimes have on biomass production, species richness, and the continued control of Kentucky bluegrass and smooth brome.

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CHAPTER 2: ANALYSIS OF SITE RECOVERY USING TOPSOIL ADDITIONS, SEEDING TREATMENTS, AND HERBICIDE WITHIN A HIGHLY MODIFIED GRASSLAND IN NORTH DAKOTA

Abstract

Grasslands in the Midwest have experienced mass reductions and conversions reducing their historic areas. Land use practices causing soil degradation hinder site productivity, species biodiversity, and can lead to invasive plant dominance. This study looks to evaluate the impact four different treatments (topsoil, Standard, Special, and Guard) may have on promoting native vegetation establishment, cover, and biodiversity while controlling introduced species within a highly modified rangeland site. Across four blocks, three seed mixes formulated based on ecological site descriptions (Standard, Special, and Guard) and a fourth treatment of topsoil prior to seeding were applied. Additionally, half of each treatment per block was sprayed with a herbicide in 2020 to assess its potential to control leafy spurge (*Euphorbia esula*). Despite promising results from 2020 illustrating the value site selected seed mixes paired with topsoil treatments have toward improving native species richness and cover while also reducing introduced species cover, the invasion of the noxious weed absinth wormwood (*Artemisia absinthium)*, likely though the topsoil additions, masked treatment impacts in 2021. The sprayed treatments that demonstrated greater native species cover and introduced species control in 2020, now show no significant differences. Total plant cover, however, was greater in all treatments compared to the Guard seed mix. Current site conditions highlight the dynamic nature of restoration projects and validate the need to employ an adaptive approach to account for unexpected site responses. Highly degraded rangeland sites will require long-term monitoring

and continual treatments to ensure invasive species don't prompt a regression in treatment impacts and successional progress.

Literature Review

Rangeland Degradation

Transitional grasslands in the Midwest have been reduced to 32% of their historic area (USDA NASS 2000; Prosser et al. 2003). Impacts to transitional grasslands can occur at military training sites where mass soil extractions take place during tank ditch trainings (Grantham et al. 2001; Prosser et al. 2003). Tank ditch trainings using heavy equipment and vehicles may injure vegetation, weaken soil structure and stability, limit rooting depths, diminish overall site productivity, and change soil, water, and nutrient dynamics (Voorhees et al. 1986; Alakukku and Elonen 1995; Prosser et al. 2003). Declines in site vegetation can cause increases in soil loss further reducing site productivity and increasing ecosystem degradation (Grantham et al. 2001). The negative impact mass soil removal has on soil health and above ground biomass increases the potential for invasive species to move into the area (Voorhees et al. 1986; Alakukku and Elonen 1995; Prosser et al. 2003).

With over 5 million hectares under their authority, the U.S. Army's land use management practices have considerable impact on the ecological integrity of numerous ecosystems across the United States (U.S. Dept. of Army 1989; Prosser et al. 2003). Within these military sites, heavy equipment is commonly used to support land development, aid in training operations, and transport troops throughout the base (Prosser et al. 2003). A focal point of these bases is to aid in the development of troop knowledge through training operations, however, another component of the training sites is the continued support of biodiverse and productive ecosystems capable of providing a variety of uses. Heavy machinery, although important to site operations, produces detrimental impacts resulting in soil erosion and diminished plant cover (Barker et al. 1998).

Human induced erosion of grasslands has led to declines in soil fertility, available organic matter, and productive soil formation (Frye et al. 1984). Soils can withstand erosion up to a certain threshold. Once the tolerance is passed, soil health deteriorates. Overused grassland sites may experience diminished carbon and nitrogen levels within soil horizons, resulting from increases in soil loss and site degradation (Baer et al. 2002). The loss of soil and organic matter are key components affecting reduced soil productivity (Bruce et al. 1995). Sites with limited and low-quality topsoil will be impacted more by erosion factors, in turn increasing the negative side effects resulting from erosion (Frye et al. 1984). Topsoil losses negatively influence processes associated with soil health such as organic, natural, and chemical components (Bruce et al. 1995; Matthees et al. 2018). Frye et al. (1984) states as the productivity of eroded soils declines, as does potential agricultural yields from the site. Soil structure can be compacted as a result of excess traffic over the surface, hindering the ability of plant roots to travel freely throughout the soil (Voorhees et al. 1986; Doll et al. 1983; Alkukku and Elonen 1995). The reduction in available space for air, water, and root movement resulting from soil compaction leads to declines in vegetation development (Doll et al. 1983; Voorhees et al. 1986; Matthees et al. 2018).

The processes involved in the creation and removal of roads are common causes of soil degradation, leading to shifts in site water movement and topography (Matthees et al. 2018). The creation and removal of oil access wells cause expedited speeds of soil runoff and erosion. Impacts occur from continued heavy impact to the soil and the altered soil composition following soil removal without proper accounts for soil horizons. Consequently, topsoil and subsoils are not distinguished from each other, leading to some soil mixtures accruing additional sodium previously accumulated in the subsoil (Qadir et al. 2014; Matthees et al. 2018). Alterations in native vegetation composition can also occur through an increase in invasive species presence and growth success, aided by diminished quality habitat for native species (Simmers and Galatowitsch 2010; Matthees et al. 2018). Sites altered through road creation and removal have diminished soil organic matter, with continued impacts on nutrients present, native plant diversity, and lowered site capacity to respond to disturbance. Restoration practices are needed to maintain the health and fertility of degraded soils to ensure continued productivity for human and natural functions (Doll et al. 1983; Bruce et al. 1995).

Grassland Restoration

Degraded grasslands may recover at a faster rate when provided with restoration treatments (Baer et al. 2002). Depending on the treatment provided, the site may experience an accelerated speed of succession towards pre-disturbance conditions. Soil surface characteristics have a strong impact on soil health, therefore, considerations of above ground biomass improvements should be made when restoring sites for improved soil productivity (Frye et al. 1984; Bruce et al. 1995). Efforts should be made to increase plant-available-water holding capacity (Frye et al. 1984). Encouraging the productivity of native perennial species with large root systems helps to supports the restoration of native grasslands though improving site production and soil organic matter (Baer et al. 2002; Bach et al. 2010). Incorporating seeding of native species specifically selected for the site's region, may steer ecosystem functions forward to a higher functioning state similar to what was present before degradation occurred. Carbon is an important component within soil given its ability to alter soil water potentials, support fertility, and limit soil loss (Bruce et al. 1995; Wick et al. 2016). Carbon within the soil may also experience faster speeds of succession with increases in carbon pools (Bruce et al. 1995). Increases in vegetation biomass through the incorporation of perennial species have demonstrated positive effects on soil organic matter within degraded sites.

Seeding

Seedbanks are often disturbed and altered following mass soil removal (Iverson and Wali 1982). Disturbances to seedbanks alter species composition and therefore, impact available vegetation for grazing. Rangeland seedbanks are close to the surface of the soil and can be highly sensitive to ground disturbances (Thilmony and Lym 2017). Invasive species establishment after site degradation can increase, therefore, reintroducing desirable native species quickly is advantageous to improving a site's native species biodiversity (Van Epps and McKell 1983; Masters et al. 1996; Török et al. 2012). The quick establishment of deep rooted, forage providing native species supports an improvement in soil stability and structure while also improving grazing forage (Masters et al. 1996; McGranahan et al. 2018). Sites with higher vegetation diversity provide a greater range of habitat niches and yield increases in available forage, if species are adequately selected based on palatability (McGranahan et al. 2018).

Within rangelands across the United States, revegetation efforts are commonly centered around improving available forage at a minimal expense, with little consideration of the site's ecological restoration potentials (Stallman 2021). Consequently, reseeding efforts often contain nonnative invasive vegetation able to be obtained at a low cost while still providing desirable forage (Simmers and Galatowitsch 2010; Redmond et al. 2013; Wood et al. 2015). This method leads to ecosystems with minimal biodiversity not representative of historical native grasslands and the values they provide (Kirmer et al. 2012). Norland et al. (2015) states, based on surveys from 123 grassland reconstructions in North Dakota and Minnesota, the incorporation of seed mixes with 20 native species led to 81% of reconstructions supporting habitats with mostly native species and produced significantly greater native biodiversity than site reconstructions using 10 species in seed mixes. Seeding diverse native species increases the chance of native species dominance within reconstruction sites, a stronger resilience to exotic species, improved resource allocation, increased production, and greater species diversity and spread over an area (Naeem et al. 1995; Tilman et al. 1996; Porkorny et al. 2005; Norland et al. 2015).

Topsoil Additions

Additions of topsoil on reclaimed sites can be used to improve plant biodiversity if adequate considerations of topsoil depths and nutrient cycling are accounted for (Bowen et al. 2005; Ramlow et al. 2018). Topsoil consists of plant material, dormant seeds, valuable nutrients, and microscopic organisms, all valuable components of productive soil (Rokich et al. 2000; Holmes 2001). If properly applied, topsoil additions increase plant community establishment and the economic value of the site through improvements in soil texture, composition, and nutrients (Doll et al. 1983; Bowen et al. 2005; Fehmi and Kong 2012; Mikha et al. 2013). Topsoil can aid in progressing successional periods of degraded sites by providing desired vegetation and nutrients (Sunding et al. 2004). This is especially important in degraded sites lacking adequate seedbanks and soil resources. Seed mixes alone have been found to be less successful than combinations of seeding and topsoil additions at reclaimed sites (Doll et al. 1983). Topsoil and seeding treatments aid vegetation establishment and germination by shielding plants from environmental stressors present at the soil surface (Nelson et al. 1970; Evans and Young 1972; Campbell and Swain 1973; Iverson and Wali 1982; Stallman 2021). Runoff can be a hindering factor in a topsoil treatment’s success towards improving plant species biodiversity (Bowen et al.

2005). A guided focus on the reintroduction of quality topsoil and native plant species to grasslands provides support for soil structures and harbors plant biodiversity (Schuman 2002).

Vegetative Threats

Leafy Spurge

Weedy species are a detrimental force impacting the success of restoration projects within reclaimed sites based the competition they give native species and their potential to spread beyond initial establishment (Masters et al. 1996; Espeland and Perkins 2017). Native species in both degraded and undisturbed sites can be impacted negatively by leafy spurge (Belcher and Wilson 1989; Masters et al. 1996; Lym et al. 2002). The invasive noxious weed is commonly found in rangelands and leads to a reduction in the sites carrying capacity (Lym and Messersmith 1985; Lym et al. 2002) because of the unpalatable and potentially toxic latex produced within the entire plant (Lym and Messersmith 1985; Lym et al. 2002). Given the value of grazing within rangelands, leafy spurge invasions can be extremely detrimental and costly to ranchers (Lym et al. 2002; Espeland and Perkins 2017). Sites overrun by leafy spurge and used for production can see yield losses up to 75%, with monetary losses totaling above $144 million within Wyoming, Montana, North Dakota, and South Dakota (Lym et al. 2002).

Management of leafy spurge requires an integrated approach given the surplus of defenses leafy spurge possesses providing the plant strong competitive advantages (Lym and Messersmith 1985; Prosser 1995; Masters et al. 1996; Lym et al. 2002). Leafy spurge has an expansive root system extending as deep as 5 meters underground for multiple years (Bakke 1936; Prosser 1995; Lym et al. 2002). The most extensively used management tool to control leafy spurge are herbicides (Prosser 1995; Masters et al. 1996; Lym et al. 2002). However, uses are typically confined to smaller study sites, with larger sites requiring integrations of different management strategies to coincide with herbicide treatments (Masters et al. 1996; Lym et al. 2002; Thilmony and Lym 2017). Treatments should initially tackle smaller sections of plots to limit leafy spurge's expansion. Additionally, Best et al. (1980) recommends limiting actions that increase bare ground presence to ensure soils have adequate vegetation cover, as 45 times more leafy spurge seeds were shown to germinate on sites with open soil compared to those with a high plant cover.

Land managers should not expect significant results after a single treatment and should maintain treatments yearly to continue control of leafy spurge (Lym and Messersmith 1985; Lym et al. 2002). Treatments should continue yearly until leafy spurge is removed from at least 90% of the restoration area (Lym et al. 2002). To support the recovery of sites dominated by leafy spurge, native seeding treatments can be applied to help native species establish themselves faster (Masters et al. 1996; Masters et al. 2001; Lym et al. 2002). Increased competition by native species limits the ability of unwanted species to establish themselves. Although an expensive addition to a restoration project, seeding treatments may be funded by assorted state and federal departments adding to the financial feasibility (Lym et al. 2002). In large leafy spurge dominated sites, an integrated approach using multiple treatments beyond herbicide alone will provide increased benefits while maintaining feasible management costs (Lym et al. 2002; Lesica and Hanna 2009).

Crested Wheatgrass

Agropyron cristatum (crested wheatgrasss) is one of the most widely seeded exotic grasses across western North America (Lesica and DeLuca 1996) and can stifle out native vegetation within grassland ecosystems causing declines in native species richness (Christian and Wilson 1999; Vaness and Wilson 2008; Nafus et al. 2016), biodiversity (Christian and Wilson 1999;

Henderson and Naeth 2005; Vaness and Wilson 2008; Hulet et al. 2010; DiAllesandro et al. 2013), available soil nutrients (Christian and Wilson 1999; Vaness and Wilson 2008), and carbon storage (Christian and Wilson 1999; Henderson and Naeth 2005). Originally introduced from Eurasia to the United States in the early 1900s, crested wheatgrass was hoped to act as a quality forage and agricultural asset (Rogler and Lorenz 1983; Vaness and Wilson 2008; Nafus et al. 2016). Despite the value of crested wheatgrass regarding its forage palatability, drought and cold tolerance, and naturalization potential (Rogler and Lorenz 1983; Lesica and DeLuca 1996; Vaness and Wilson 2008), there are numerous concerns regarding its spread within grassland ecosystems (Lesica and DeLuca 1996; Christian and Wilson 1999; Henderson and Naeth 2005; Vaness and Wilson 2008; Nafus et al. 2016). Crested wheatgrass is not only able to outcompete some weedy species but can also displace native vegetation and create monocultural stands (Lesica and DeLuca 1996; Christian and Wilson 1999; Henderson and Naeth 2005; Nafus et al. 2016) resulting in lower root mass, soil organic matter (Dormaar et al. 1995), and ecosystem services. The impact crested wheatgrass has on wildlife habitat, specifically sage-grouse habitat, has been a central concern for many land managers (McAdoo et al. 2017). Correspondingly, researchers caution against the continual plantings of crested wheatgrass (Lesica and DeLuca 1996; Henderson and Naeth 2005). Lesica and Cooper (2019) recommend seeding grasses including *Agropyron smithi* (western wheatgrass), *Stipa comata* (needlegrass), *Poa secunda* (sandberg bluegrass), and *Bouteloua gracilis* (blue grama) to help restore sites dominated by crested wheatgrass within the Great Plains. DiAllesandro et al. (2013) encourages improving a sites species richness by managing crested wheatgrass.

Yellow Sweetclover

Melilotus officinalis (yellow sweetclover), a nitrogen fixing legume, was introduced to the United States from Eurasia and provides forage for livestock and wildlife (Lesica and DeLuca 2009). The legume is highly competitive when competing for resources such as light, water, and space (Lesica and DeLuca 2000; Dickson et al. 2010; Spellman et al. 2011), and often decreases the abundance of not only native vegetation (Dickson et al. 2010) but even other introduced species such as crested wheatgrass (Lesica and DeLuca 2000). Within the Northern Great Plains where vegetation is adapted to systems lower in nitrogen levels (Lesica and DeLuca 2000; Dornbush et al. 2018), yellow sweetclover can alter the balance by increasing soil nitrogen levels (Conn and Seefeldt 2009; Dickson et al. 2010; Van Riper et al. 2010; Dornbush et al. 2018). Through increasing available nitrogen, yellow sweetclover shifts rangeland ecosystems towards favoring species more adept to utilizing high levels of nitrogen, often invasive species (Lesica and DeLuca 2000; Conn and Seefeldt 2009; Dickson et al. 2010; Van Riper et al. 2010; Dornbush et al. 2018). Dornbush et al. (2018) found species such as yellow sweetclover may encourage the spread of Kentucky bluegrass based on the increased levels of nitrogen. Species adapted to environments with limited resources often have less rapid growth rates making their systems more susceptible to invasion by early successional plants based on their ability to utilize resource more rapidly (Van Riper et al. 2010). Within degraded sites where restoration goals center around successional progression, yellow sweetclover may be detrimental to this goal (Wolf et al. 2003). Despite the alterations yellow sweetclover can cause to rangelands, the financial value associated with yellow sweetclover creates difficulty in its management and reduction (Van Riper et al. 2010).

Absinth Wormwood

Artemisia absinthium (absinth wormwood), introduced from Europe, is listed as a noxious weed in the state of North Dakota, as well as in numerous other states (Lym 2018). Absinth wormwood has medicinal value which was a component influencing its spread (Maw et al. 1985; Goud et al. 2015). Absinth wormwood not only negatively impacts forage availability and production, but consumption of absinth wormwood by livestock can contaminate the milk they produce (Maw et al. 1985; Lym et al. 1995; Goud et al. 2015; Lym 2018). Absinth wormwood has expanded quickly across the grasslands of North Dakota, especially within overgrazed sites, and because of its limited palatability it is of concern to producers (Maw et al. 1985; Lym et al. 1995; Lym 2018; Reed et al. 2018). Within degraded rangelands, absinth wormwood has an increased capacity to spread based on limited competition from native grasses (Maw et al. 1985; Reed et al. 2018).

Methods

Utilization of rangelands by the U.S. Army in the form of heavy equipment trainings are necessary but can negatively impact the landscape. Limited research has evaluated techniques for restoring rangelands following tank ditch trainings and the subsequent removal of topsoil. The multiuse value rangelands provide highlights the need to restore degraded rangelands to resilient positions where they can produce valuable forage for livestock and provide their plethora of ecosystem services. The objective of this restoration project is to increase native vegetation establishment and ground coverage within a highly modified and invaded rangeland site. Project findings will provide insight on potential restoration pathways following heavy equipment usage that may be capable of establishing vegetation on sites with limited to no topsoil availability.

The restoration project initially began in 2019 when topsoil, seeding, and herbicide treatments were applied. The following year, in 2020, an additional herbicide application was applied within study plot halves, with relative cover data having been collected from 2019 to 2021. By evaluating response variables of species richness, native species richness, introduced species richness, total cover, absinth wormwood relative cover, leafy spurge relative cover, forb relative cover, native relative cover, and introduced relative cover researchers can assess if improvements have been made towards reaching restoration goals. Project success will be defined by improvements in native vegetation coverage, reductions in bareground, and control of invasive species.

The study site is located in Eddy County, North Dakota within the Camp Grafton South (CGS) Tank Ditch Area (47°43'26.4"N, 98°39'40.4"W). CGS is a U.S. Army training center, managed by the North Dakota Army National Guard (U.S. Dept. of Army 1989; Prosser et al. 2003). CGS attributes roughly 4,000 ha of transitional grasslands and provides valuable land for grazing, recreation, and military training operations (Global Security 2011). High intensity military exercises, specifically in the form of tank ditch trainings, have led to declines in soil health and subsequent reductions in native vegetation composition, cover, and diversity (Grantham et al. 2001; Prosser et al. 2003). The area of study is 1.44 ha of rangeland, previously excavated during heavy equipment operation trainings performed by the North Dakota National Guard (Figure 2.1). The study site has also been subjected to heavy grazing for extended periods of time continuing up until study site treatments began.

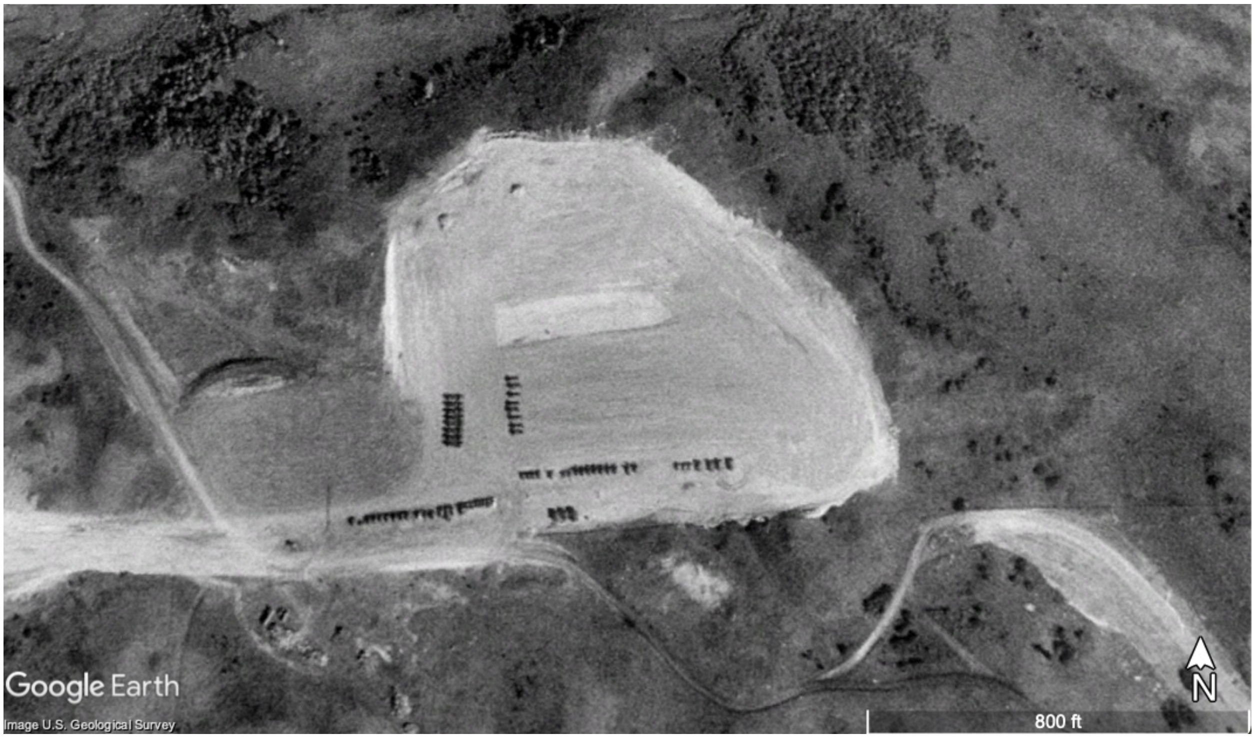

Figure 2.1: Google Earth image of the Camp Grafton study site in 1997 illustrating the impact of military trainings on the landscape (Google Earth 2021).

The site is in MLRA 55B, the Central Black Glaciated Plains (USDA-NRCS 2006). This region is characterized by gently rolling glacial till plains with glacial lacustrine deposits. CGS characteristics are based on its location within the Transitional Grasslands prairie region with close proximity to the End Moraine Complex. The site was historically influenced by Wisconsin glacial presence producing lacustrine deposits (Barker and Whitman 1988; Bryce et al. 1996). Contemporary impacts, specifically the use of heavy equipment operation trainings, have played a strong role in altering the available topsoil, with grazing and farming considered unsuitable for the site. The study site also slopes downward from the northern section of the site towards the southern section of the site.

Average climatic conditions over a 30-year period from 1991-2020 from the McHenry 3W Station, the closest station to the study site (24km), list the average annual maximum temperature at 10.5°C, the minimum temperature at -1.11°C, and the average temperature at 4.67°C (NOAA 2021). Peak precipitation occurs during the summer, averaging of 24.74 cm, with June accruing the most rainfall at an average of 9.25 cm. Annual average precipitation is 53.7 cm.

Vegetation is made up of mixed-grass and tallgrass prairie species, with a strong presence of leafy spurge within portions of the site. Historic vegetation within this region included *Pascopyrum smithii* (western wheatgrass), *Stipa viridula* (green needlegrass), and *Bouteloua gracilis* (blue grama), with *Schizachyrium scoparium* (little bluestem) and *Bouteloua curtipendula* (sideoats grama) attributed to more erodible soils (USDA-NRCS 2006). Sites with greater soil saturation in this region are characterized by *Spartina pectinata* (prairie cordgrass), *Calamagrostis stricta* (northern reedgrass), *Andropogon gerardii* (big bluestem) and *Carex atherodes* (wheat sedge). Typically found shrubs and half shrubs within this region include *Symphoricarpos occidentalis* (western snowberry), *Amorpha canescens* (leadplant), and *Rosa arkansana* (prairie rose). Major sources of soil degradation are attributed to wind and water erosion with key focuses on potential impact points to soil health and productivity. The soils within this site are predominantly Udorthents distinguished as a loamy, mixed, superactive, calcareous, and frigid (USDA-NRCS 2021a). The site has slopes ranging from 0 to 15% and has deep, well-drained soils within the medium runoff class.

The National Environmental Policy Act (NEPA) mandates all federal agencies or partners of federal agencies comply with assessing environmental impacts of a study site and following mandated guidelines set into place based on the project impacts (16 U.S.C. §§1531-1544). This entails the recovery of potentially lost species diversity, land productivity, and accounting for impacts to threatened and endangered species as indicated by the Threatened and

Endangered Species Act. Additionally, sites should be, based on Executive Order 13112, managed to limit the spread of invasive species and work towards improving native species presence and habitats upon invasive species presence (Exec. Order No. 13112, 1999).

Within the CGS, considerations on the impact heavy machinery and mass soil removal and replacement were made to assess repercussions towards soil health, site biodiversity, and land productivity. Based on the required restoration actions put into place following tank ditch trainings, adequate environmental conditions of the site must be reached. Therefore, the study site is undergoing restoration efforts to improve soil health, native species richness, and decrease invasive species presence with a specific focus on leafy spurge. Restoration of the study site will allow for future use by the National Guard and ranchers, while supporting necessary ecosystem services.

The study design consists of a 1.44 ha plot separated into four blocks with all blocks including four 30m x 30m randomly assigned treatment replications (Figure 2.2). Treatments include 1) 5 cm (2 in) topsoil added to the soil surface and carried out in conjunction with a standard seed mix (Topsoil), 2) a specialty seed mix (Specialty) (no added soil), 3) a standard seed mix (Standard) (no added soil), and 4) a National Guard seed mix (Guard) (no added soil). Topsoil treatments were obtained from a surplus at CGS within 3.22 km from the study site (Figure 2.3). Prior to treatments beginning, the study site was sectioned off to exclude livestock from grazing. Treatment replications within each block were also sectioned into halves, with a single half receiving herbicide treatments and the other half being treated normally. This was done to understand the potential for herbicide to be used alongside treatments to control leafy spurge.

Figure 2.2: Camp Grafton restoration layout including topsoil, special, standard, and guard seed mixes and herbicide application design.

Figure 2.3: Picture taken in 2019 of Camp Grafton study site visualizing topsoil additions, depicted in darker colored squares (Photo by Stallman 2021).

Specialty seed mixes including eleven native grasses, five native forbs, and one native sedge were obtained from the Natural Resource Conservation Service – Ecological Site Descriptions for MLRA 58C, with considerations of both sites similarly degraded soils (Table 2.1). Within the specialty mix 13 species were considered desirable for cattle forage and four were considered undesirable for cattle forage. Applications of specialty seeding took place at a rate of 17.53 of pure live seed (PLS) kg/ha. Standard seed mixes were obtained from the Natural Resource Conservation Service – Ecological Site Descriptions for MLRA 55B with the inclusion of seven native grasses and five forb species. The standard seed mix consisted of 8 desirable cattle forage species and 4 undesirable cattle forage species (Table 2.2). Application of the standard seed mix took place at a rate of 18.61 of PLS kg/ha. The Guard seed mix included seven grass species with a mix of native and nonnative grasses as well as one non-native clover. All species are considered desirable forage for cattle (Table 2.3.). The Guard seed mix was chosen based on rapid establishment and the commonality of its use to provide ample forage throughout CGS. Millborn Seeds in Brookings, South Dakota provided all seed mixes.

Table 2.1: Species composition and seeding rate for the specialty mix.

Common Name	Scientific Name[a]	Seeding Rate (kg/ha)	Origin[a]	Forage Value for Cattle [b]
Little Bluestem - MN native Badlands	*Schizachyrium scoparium*	5.949	Native	Desirable
Thickspike Wheatgrass - certified Critana	*Elymus lanceolatus*	2.852	Native	Desirable
Sideoats Grama - Pierre	*Bouteloua curtipendula*	2.050	Native	Desirable
Bluebunch Wheatgrass - certified Goldar	*Pseudoroegneria spicata*	1.348	Native	Desirable
Canada Wildrye - Mandan	*Elymus canadensis*	1.158	Native	Desirable
Indian Ricegrass - certified Rimrock	*Achnatherum hymenoides*	0.917	Native	Desirable
Western Yarrow	*Achillea millefolium*	0.657	Native	Undesirable
Bluebunch Wheatgrass - Anatone	*Pseudoroegneria spicata*	0.652	Native	Desirable
Sand Dropseed - ND native sourced	*Sporobolus cryptandrus*	0.531	Native	Desirable
Prairie Junegrass	*Koeleria macrantha*	0.366	Native	Desirable
Blue Grama - Bad River	*Bouteloua gracilis*	0.326	Native	Desirable
Prairie Dropseed - MN native sourced	*Sporobolus heterolepis*	0.246	Native	Desirable
Blanketflower	*Gaillardia aristata*	0.210	Native	Undesirable
Brown Fox Sedge - IA native sourced	*Carex vulpinoidea*	0.100	Native	Desirable
Prairie Onion - IA native sourced	*Allium stellatum*	0.090	Native	Undesirable
Dotted Blazing Star - ND native sourced	*Liatris punctata*	0.070	Native	Desirable
Prairie Sage - IA sourced	*Artemisia frigida*	0.010	Native	Undesirable

[a]Scientific names and species origins are from The PLANTS Database: USDA, NRCS. 2021. The PLANTS Database (http://plants.usda.gov, October 2021). National Plant Data Team, Greensboro, NC 27401-4901 USDA. Plant origins are based on native or introduced status within the contiguous United States.
[b] Forage value indicator is based off the sixth edition Range Judging Handbook for North Dakota (Sedivec and Elemes 2019).

Table 2.2: Species composition and seeding rate for the standard mix.

Common Name	Scientific Name[a]	Seeding Rate (kg/ha)	Origin[a]	Forage Value for Cattle[b]
Blue Gramma - Bad River	*Bouteloua gracilis*	0.251	Native	Desirable
Western Wheatgrass - certified Rosane	*Pascopyrum smithii*	3.498	Native	Desirable
Slender Wheatgrass - certified Revenue	*Elymus trachycaulus*	3.368	Native	Desirable
Sideoats Grama - Pierre	*Bouteloua curtipendula*	2.386	Native	Desirable
Switchgrass - Sunburst	*Panicum virgatum*	1.133	Native	Desirable
Prairie Junegrass	*Koeleria macrantha*	0.431	Native	Desirable
White Prairie Clover - MN native sourced	*Dalea candida*	0.281	Native	Desirable
Blue Gramma - Bad River	*Bouteloua gracilis*	0.251	Native	Desirable
Prairie Coneflower - IA native sourced	*Ratibida columnifera*	0.170	Native	Undesirable
Canada Milkvetch - ND native sourced	*Astragalus canadensis*	0.110	Native	Undesirable
Western Yarrow	*Achillea millefolium*	0.050	Native	Undesirable
Prairie/Fringed Sage - IA native sourced	*Artemisia frigida*	0.010	Native	Undesirable

[a]Scientific names and species origins are from The PLANTS Database: USDA, NRCS. 2021. The PLANTS Database (http://plants.usda.gov, October 2021). National Plant Data Team, Greensboro, NC 27401-4901 USDA. Plant origins are based on native or introduced status within the contiguous United States.
[b] Forage value indicator is based off the sixth edition Range Judging Handbook for North Dakota (Sedivec and Elemes 2019).

Table 2.3: Species composition and seeding rate for the National Guard mix.

Common Name	Scientific Name[a]	Seeding Rate (kg/ha)	Origin[a]	Forage Value for Cattle [b]
Jerry Oats	*Avena sativa*	12.178	Introduced	Desirable
Western Wheatgrass - certified Rosana	*Pascopyrum smithii*	2.977	Native	Desirable
Big Bluestem - Bison	*Andropogon gerardii*	2.345	Native	Desirable
Sideoats Grama - Pierre	*Bouteloua curtipendula*	2.285	Native	Desirable
Green Needlegrass - Lodorm	*Nassella viridula*	2.215	Native	Desirable
Yellow Blossom Sweet Clover	*Melilotus officinalis*	2.205	Introduced	Desirable
Crested Wheatgrass - certified Hycrest	*Agropyron cristatum*	1.694	Introduced	Desirable
Switchgrass - Sunburst	*Panicum virgatum*	1.443	Native	Desirable

[a] Scientific names and species origins are from The PLANTS Database: USDA, NRCS. 2021. The PLANTS Database (http://plants.usda.gov, October 2021). National Plant Data Team, Greensboro, NC 27401-4901 USDA. Plant origins are based on native or introduced status within the contiguous United States.
[b] Forage value indicator is based off the sixth edition Range Judging Handbook for North Dakota (Sedivec and Elemes 2019).

Herbicide pre-treatments of glyphosate were applied at a rate of 2.33 L/ha and paired with methylated seed oil (MSO) (Southern Agricultural Insecticides, Inc. Worldwide Rights Reserved) applied at 1.75 L/ha. Treatments of glyphosate and methylated seed oil (MSO) (Southern Agricultural Insecticides, Inc. Worldwide Rights Reserved) were applied the first year of experimentation prior to seeding on the entire site. After seeding (i.e. the following year), treatments of Facet® applied at 4.68 L/ha (BASF Corporation, Worldwide Rights Reserved) paired with Overdrive® were applied at 0.42 kg/ha (BASF Corporation, Worldwide Rights Reserved), and MSO applied at 1.75 L/ha was used to treat the study site for leafy spurge. Herbicide treatments were carried out using a boom sprayer mounted on an ATV.

In August 2019, seedling counts of forbs and grasses took place using a 0.25 m^2 quadrat. Each count occurred four times per block, amounting to 64 total samples. Further vegetation surveys were conducted in August 2020 and 2021 using a surface coverage survey. In each block, 1 m^2 quadrats were sampled six times, with three samples in the sprayed half and three in the unsprayed half. In 2020 and 2021 a total of 96 quadrats were taken to assess ground cover percentages, with bare ground and litter coverage classes assigned in each sample.

A multi-way analysis of variance using SAS procedure, version 9.4 of the SAS system for Windows (Copyright © 2013 by SAS Institute Inc. Cary, NC, USA) was conducted. An ANOVA was used to analyze results from 2021 with the interaction term of being A + B + A x B. Data was not transformed for the ANOVA analysis based on the skewness equally less than zero, plus or minus 1. The dependent variables analyzed were species richness, native species richness, introduced species richness, total cover, absinth wormwood relative cover, leafy spurge relative cover, forb relative cover, native relative cover, and introduced relative cover. Independent variables included block (one, two, three, and four), treatment (topsoil, specialty, standard, guard), herbicide application (sprayed and unsprayed), and their interactions. Least squares (LS) means estimated comparisons were used to assess the interaction between treatments, with each group including two members (sprayed/unsprayed and block).

Results

The ANOVA indicated ($F_{19,12} = 3.71$, $P = 0.0121$) total cover was influenced by treatment ($F_{19,12} = 8.34$, $P = 0.0029$) and treatment by block ($F_{19,12} = 3.35$, $P = 0.0274$). Guard treatments were significantly lower in total cover in comparison to the topsoil, special, and standard treatments (Figure 2.4). Significant differences were present in block 1 between guard and topsoil treatments ($P = 0.0356$) (Figure 2.5).

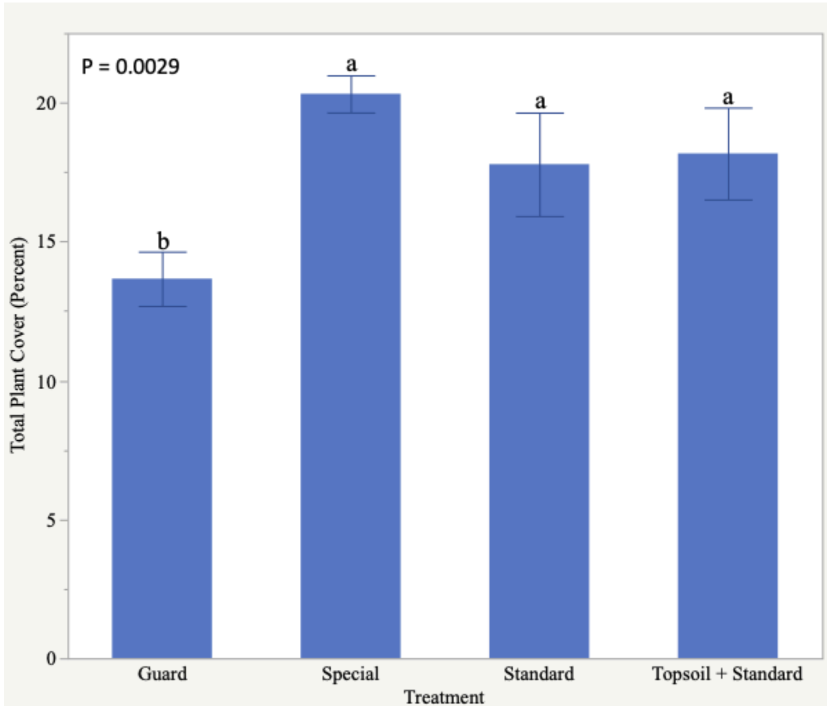

Figure 2.4: Average total plant cover percentage (± standard error) by treatment in 2021. A statistical difference is indicated with different letters at α = 0.05.

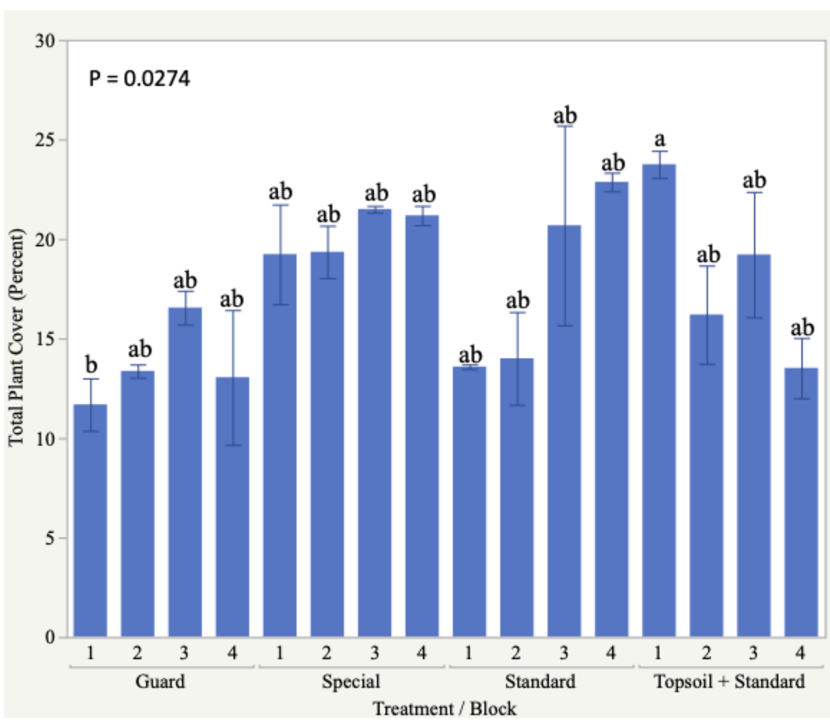

Figure 2.5: Average total plant cover percentage (± standard error) by treatment by block in 2021. A statistical difference is indicated with different letters at α = 0.05.

The ANOVA indicated no significant differences in species richness ($F_{19,12} = 0.78$, P = 0.693) (Figure 2.6a), native species richness ($F_{19,12} = 0.93$, P = 0.5722) (Figure 2.6b), introduced species richness ($F_{19,12} = 0.47$, P = 0.9294) (Figure 2.6c), leafy spurge relative cover ($F_{19,12} =$ 1.37, P = 0.2906) (Figure 2.7a), forb relative cover ($F_{10,21} = 1.79$, P = 0. 0.1249, Figure 2.7b), native relative cover ($F_{19,12} = 1.7$, P = 0.1753) (Figure 2.7c), and introduced relative cover ($F_{19,12}$ = 1.7, P = 0.1753) (Figure 2.7d).

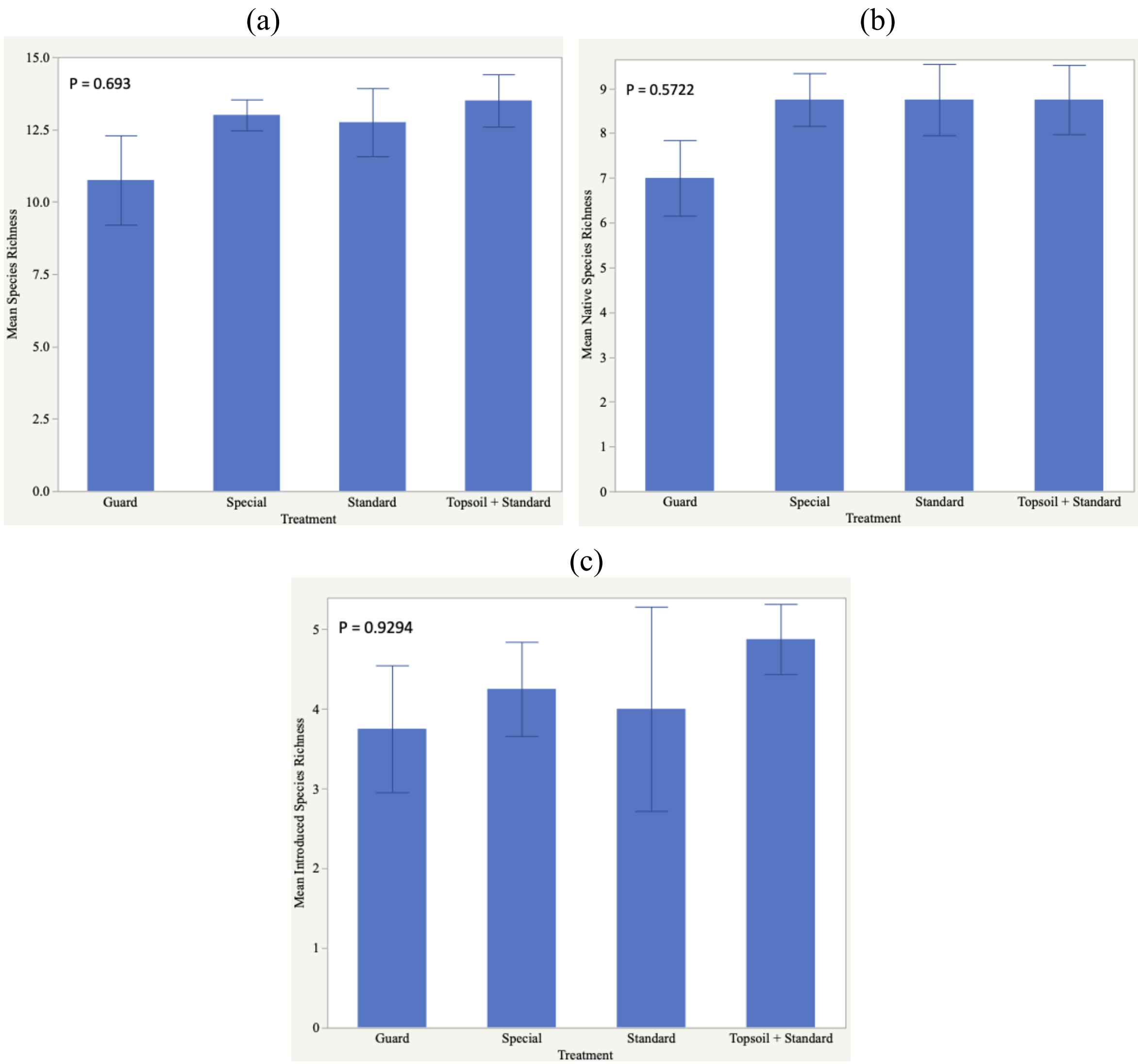

Figure 2.6: Average species richness (a), native species richness (b) and introduced species richness (c) (± standard error) by restoration treatment. Average species richness, native species richness, and introduced species richness were not significantly different across topsoil + standard, special, standard, and guard treatments in 2021.

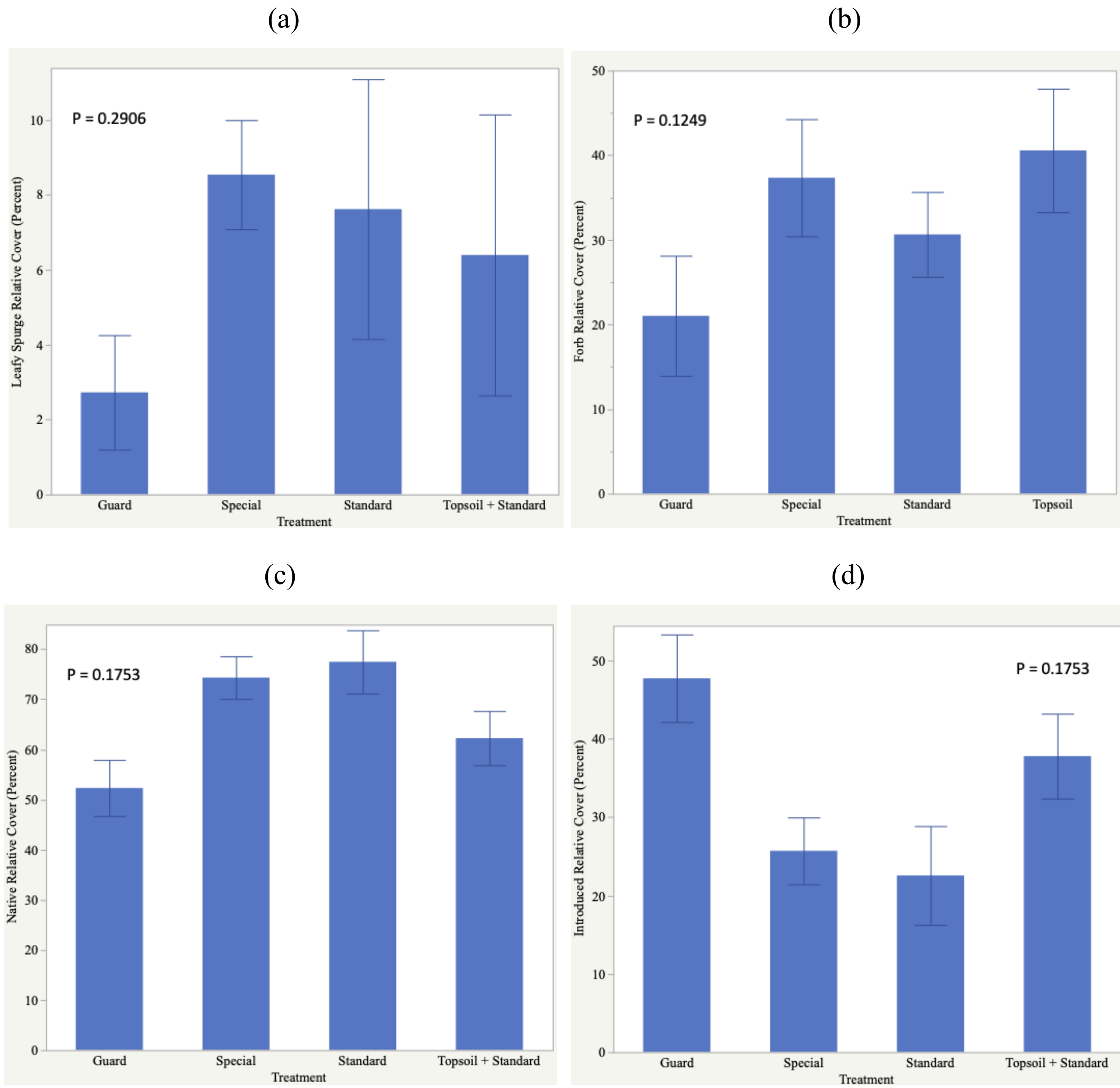

Figure 2.7: Average leafy spurge relative cover (a), forb relative cover (b), native relative cover (c), and introduced relative cover (d) (± standard error) by restoration treatment. Average leafy spurge relative cover, forb relative cover, native relative cover, and introduced relative cover were not significantly different across topsoil + standard, special, standard, and guard treatments in 2021.

The ANOVA indicated ($F_{19,12}$ = 4.69, P = 0.0044) absinth wormwood relative cover was influenced by treatment ($F_{19,12}$= 8.21, P = 0.003) (Figure 2.8, Figure 2.9), herbicide application by treatment ($F_{19,12}$ = 7.03, P = 0.0055, Figure 2.10), and treatment by block ($F_{19,12}$ = 4.51, P =

0.009) (Figure 2.11). Absinth wormwood relative cover was higher in topsoil treatments than special and standard treatments (Figure 2.8; Figure 2.9).

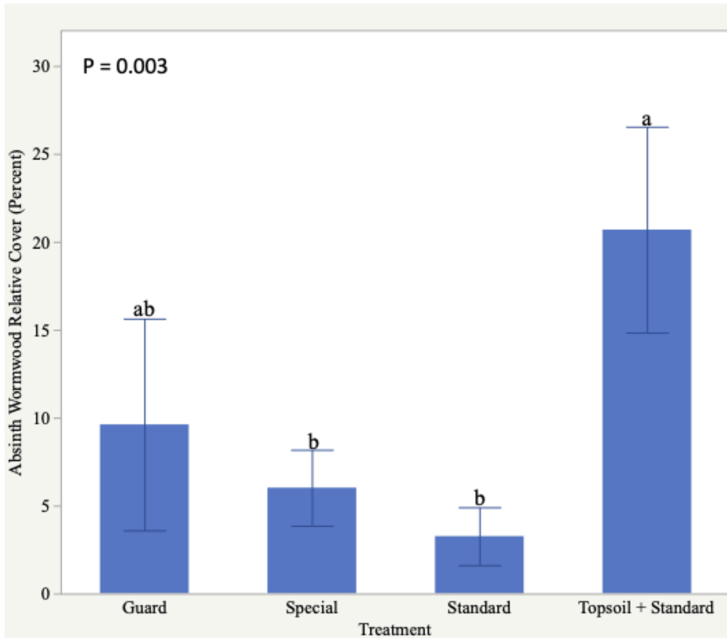

Figure 2.8: Average absinth wormwood cover percentage (± standard error) by treatment in 2021. A statistical difference is indicated with different letters at $\alpha = 0.05$.

Figure 2.9: Picture taken July 2021 of Camp Grafton illustrating establishment of species within the study site. This picture specifically highlights the presence of absinth wormwood, with a strong visual presence in the topsoil plots (Photo by Dr. Edward DeKeyser, 2021).

Each herbicide treatment group had four members (unsprayed: topsoil, special, standard, and guard; sprayed: topsoil, special, standard, and guard). Across herbicide application type by treatment, unsprayed treatments of topsoil had significantly greater absinth wormwood cover compared to all other treatments except sprayed guard treatments (Figure 2.10).

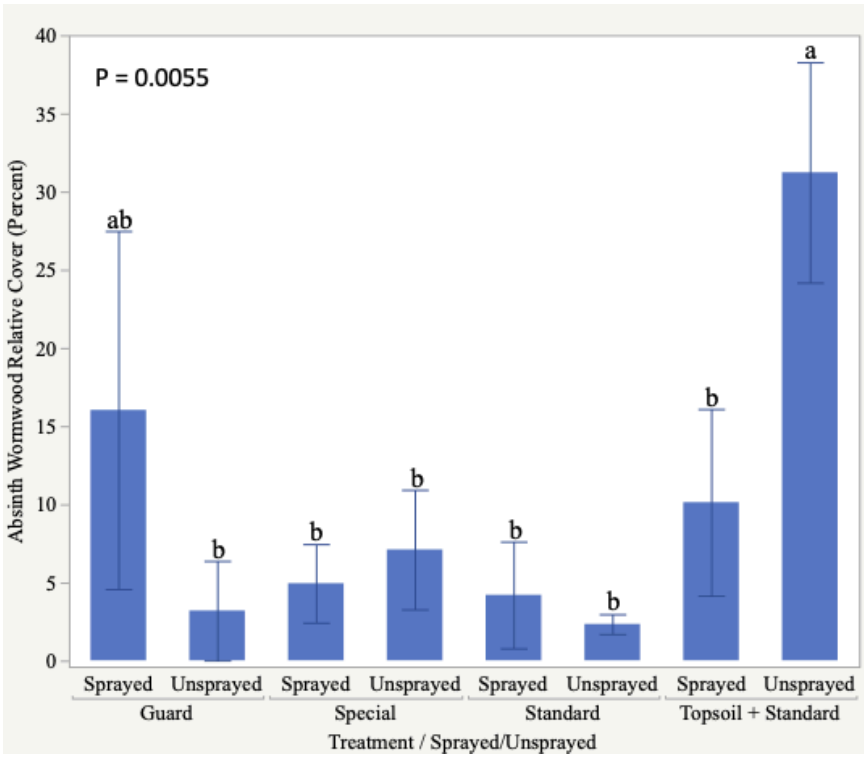

Figure 2.10: Average absinth wormwood cover percentage (± standard error) across treatment by herbicide application in 2021. A statistical difference is indicated with different letters at α = 0.05.

There was a significant interaction term (treatment x block) resulting in the topsoil treatment in block 1 responding differently than blocks 2 ($P = 0.0444$) and 3 ($P = 0.0287$) of the special treatments, blocks 1 ($P = 0.0345$), 2 ($P = 0.0317$) and 3 ($P = 0.0238$) of the standard treatments, and blocks 1 ($P = 0.0209$) and 4 ($P = 0.0316$) of the guard treatments (Figure 2.11).

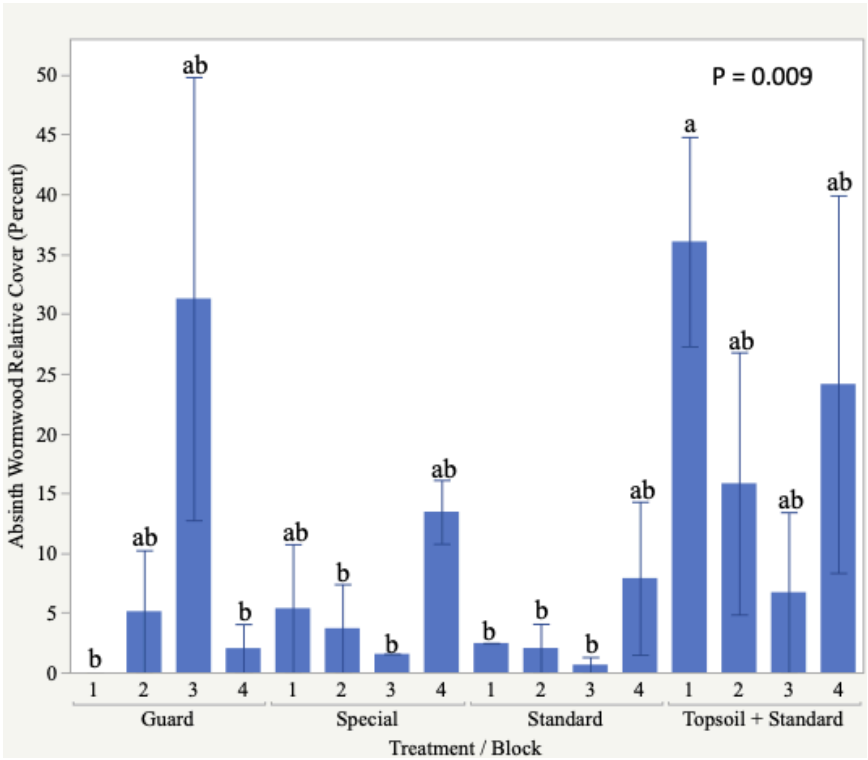

Figure 2.11: Average absinth wormwood cover percentage (± standard error) across treatment by block in 2021. A statistical difference is indicated with different letters at α = 0.05.

Discussion

This grassland restoration evaluated the ability of four different treatments to improve native species diversity and cover, and reduce introduced species diversity and cover within a degraded and eroded rangeland site. Further goals of the study were to assess the influence herbicide treatments would have on the noxious weed leafy spurge, which dominated the CGS site. It was predicted treatments utilizing a standard seed mix paired with topsoil additions would result in the greatest establishment of native species and have the greatest influence on achieving restoration objectives. Despite promising results from 2020 surveys displaying improvements in native species richness within the topsoil treatments (Stallman 2021), current results display a dominance of absinth wormwood within the topsoil treatments. There are no longer differences in native or introduced species richness, native or introduce relative cover, leafy spurge relative cover, or forb relative cover across any treatments as were seen in 2020.

Absinth wormwood's dominant coverage across topsoil treatments indicates the potential for absinth wormwood seeds to have been present in the topsoil's seedbank. It is possible absinth wormwood was transferred to the study site via soil or wind, and from there was able to establish relatively quickly (Rokich et al. 2000; Golos and Dixon 2014; Khan et a. 2018). The topsoil's higher nutrient levels may have facilitated the invasive species spread (DiAllesandro et al. 2013). DiAllesandro et al. (2013) highlights the need for caution when applying topsoil within degraded systems as they may increase invasive species spread given the topsoil's potential to have higher nitrogen or phosphorous levels. Along with the topsoil treatment, the guard treatment had more absinth wormwood than the standard and specialty seed mix treatments. The greater coverage of absinth wormwood in the guard treatment could be attributed to the yellow sweetclover that was planted within the guard treatment. Yellow sweetclover's ability to increase available nitrogen within soil favors weedy and ruderal species (Lesica and DeLuca 2000; Conn and Seefeldt 2009; Dickson et al. 2010; Van Riper et al. 2010; Dornbush et al. 2018), and therefore, it is possible absinth wormwood had increased support towards its spread within the guard treatments.

Analysis on leafy spurge cover indicated no differences were present between treatment, block, or herbicide applications. Despite differences between sprayed and unsprayed plots in 2020, those findings were not exhibited in 2021. The lack of differences in leafy spurge cover could be attributed to the dominance of absinth wormwood across the site. Often control of one invasive species, as was present in 2020, can lead to another invasive species filling the available niche (Murphy and Grant 2005; Hendrickson and Lund 2010).

Total plant cover was lowest in the guard treatment compared to all other treatments. Seeding treatments to revegetate degraded sites utilizing a diverse grouping of species chosen to represent a given site can support improved revegetation and plant coverage (Van Epps and McKell 1983; Holmes 2001). Given the standard and specialty seed mixes were selected specifically for CGS it makes sense they resulted in the greatest total coverage. Results demonstrating a significant difference in total plant cover influenced by seed treatment type and block are not unexpected given the terrain associate with the site. The blocks were arranged along a slope gradient with surface rills and erosional impacts, therefore, it is not surprising block and treatment interactions were present. Environmental conditions have a major influence on the impact treatments may have (Lepš et al. 2007), thus small fluctuations in conditions are likely present between replications and blocks creating differences in the effects treatments may have across the study site.

Unfortunately, evaluations on the effectiveness of seed mixes derived from ecological site descriptions were hindered by the unexpected invasion of absinth wormwood. The rapid invasion of absinth wormwood may have masked the impact seed mixes, topsoil additions, and herbicide applications had on reaching restoration goals. The spread of absinth wormwood within the restoration site highlights the need to ensure topsoil treatments are certified weed free. In the two years since restoration began at CGS the study site has seen a shift from highly eroded soils invaded by leafy spurge to preliminary improvements in native species establishing and introduced species control in 2020 to the study sites current state displaying a regression in observable treatment impacts likely due to the absinth wormwood invasion. The study is an example of the dynamic nature of restoration projects and further validates the need to employ adaptive management to restoration projects as unexpected alterations in the restoration site arise

(Kentula 2000; Sunding et al. 2004; Zedler and Kercher 2005; Matthews et al. 2020). Given the current condition of the Camp Grafton restoration, limited conclusions can be made on the effectiveness of each treatment given no significant differences exist between treatments beyond absinth wormwood relative cover and total cover.

Management Implications

Absinth wormwood control measures are recommended given the present invasion is mostly confined to the study area. Herbicide applications are the most common control treatment recommended (Lym et al. 1995; Lym 2018; Reed et al. 2018). Treatments utilizing aminopyralid (e.g. Milestone ®), 2, 4-D or glyphosate have proved to be effective towards suppressing the weed, however considerations on the present grass coverage are needed to ensure excess harm does not occur to native vegetation.

Future restoration projects within CGS should utilize seeding mixes containing only native species and avoid the intentional seeding of yellow sweetclover or crested wheatgrass. Both species are considered opportunistic invaders (Northern Great Plains Floristic Quality Assessment Panel 2001) and despite their forage value, the detrimental impacts they have on native biodiversity, vegetation dynamics (Dickson et al. 2010), soil nutrient levels (Lesica 2019), and ecosystem resilience (Tilman 2006) make them undesirable candidates for seeding mixes. Seed mixes should instead utilize native vegetation mixes curated with accounts for site specific conditions (Holmes 2001; Kimiti er al. 2017). Especially within sites with limited soil fertility, site selected vegetation can aid in reaching revegetation goals. Based on our data and the species most often encountered during surveys this suggested seed mix will continue to provide the desired palatable forage to livestock while supporting a more diverse ecosystem with improved capacity for successional progress. We believe, based on our data, we have come up with an economical seed mix of native species graminoids and forbs that will provide soil stability, high quality forage, and pollinating habitat in an acceptable time frame (Table. 2.4).

Table 2.4: Suggested seed mix and seeding rates for the National Guard to use in future grassland restorations.

Common Name	Scientific Name[a]	Seeding Rate (kg/ha)	Origin[a]	Forage Value for Cattle[b]
Western Wheatgrass - certified Rosana	*Pascopyrum smithii*	2.977	Native	Desirable
Big Bluestem - Bison	*Andropogon gerardii*	2.345	Native	Desirable
Little Bluestem - MN native Badlands	*Schizachyrium scoparium*	5.949	Native	Desirable
Sand Dropseed - ND native sourced	*Sporobolus cryptandrus*	0.531	Native	Desirable
Sideoats Grama - Pierre	*Bouteloua curtipendula*	2.285	Native	Desirable
Green Needlegrass - Lodorm	*Nassella viridula*	2.215	Native	Desirable
Slender Wheatgrass - Certified Revenue	*Elymus trachycaulus*	3.368	Native	Desirable
Switchgrass - Sunburst	*Panicum virgatum*	1.443	Native	Desirable
Blue Gramma - Bad River	*Bouteloua gracilis*	0.251	Native	Desirable
Prairie Coneflower - IA native sourced	*Ratibida columnifera*	0.170	Native	Undesirable
White Prairie Clover - MN native sourced	*Dalea candida*	0.281	Native	Desirable
Purple Prairie Clover	*Dalea purpurea*	4.266	Native	Desirable
Western Yarrow	*Achillea millefolium*	0.657	Native	Undesirable

[a] Scientific names and species origins are from The PLANTS Database: USDA, NRCS. 2021. The PLANTS Database (http://plants.usda.gov, October 2021). National Plant Data Team, Greensboro, NC 27401-4901 USDA. Plant origins are based on native or introduced status within the contiguous United States.
[b] Forage value indicator is based off the sixth edition Range Judging Handbook for North Dakota (Sedivec and Elemes 2019).

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CHAPTER 3: ASSESSMENT OF WETLAND RESTORATION TECHNIQUES WITHIN SOUTHEASTERN NORTH DAKOTA

Abstract

Wetlands provide essential ecosystem services within and beyond their system's bounds. However, factors from multiple fronts reduce the health of wetlands across the United States. Conversions from healthy, natural wetlands deplete the ecosystem services they provide. Due to losses in wetland acreage and productivity, there is a need for research to focus on cost effective restoration measures benefiting both landowners and wetland health. This long-term study looks to account for the actual cost of various plant community restoration methods and the associated benefits regarding monetary investments in additional treatments. The information collected provides details on early successional alterations to the wetland plant communities following restoration treatments. Treatments began in 2013, on a 11.04 ha study site separated into nine plots of roughly the same size (1.23 ha) and organized into three blocks with three treatments. Treatments included applications of 1) Seed only, 2) Seed + mulch, 3) Seed, mulch, + transplanted soil plugs. An ANOVA completed on vegetation survey data in 2017, found as the intensity of the restoration actions increased, so did the cost of the restoration. The 2021 surveys indicate that a greater investment in restoration methods did not produce a greater benefit when considering species richness, diversity, and total plant cover. However, improvements were found to exist when treatments were compared against the initial site conditions. Follow up research is needed to accurately account for the long-term impacts of the restoration techniques. Assessments of the study site are ongoing and through an adaptive management approach, post restoration treatments will be applied to the study site including prescribed burns, grazing, and herbicide applications.

Literature Review

Prairie Pothole Region Wetlands

The Prairie Pothole Region (PPR) encompasses a large area, approximately 900,000 km^2, between upper central United States and lower central Canada and provides a plethora of valuable ecosystem services (Gleason et al. 2005; Gleason et al. 2008; Gleason et al. 2011; Zilverberg et al. 2014). Wetlands within the PPR are characterized by their formation originating from past Pleistocene glaciation, which created many depressional topographical features (Johnson et al. 2005; Gleason et al. 2011). These wetlands are further distinguished by the grassland species present, supported by the mid-continental climate of the region. The services PPR wetlands provide include wildlife habitat, forage, limiting flooding potential, filtration of water, and recharging groundwater (Kirby et al. 2002a; Kirby et al. 2002b; Mitsch and Gosselink 2007; Gleason et al. 2011; Zilverberg et al. 2014; Guretzky et al. 2017; Hemes et al. 2018), with wetlands of this region supporting 50-80% of North America's ducks (Johnson et al. 2005). Other benefits include the highly valuable ecosystem service of sequestering carbon as well as producing high quality carbon-containing soil (Gleason et al. 2011; Galatowitsch 2012; Zilverberg et al. 2014; Hemes et al. 2018; Biró et al. 2019). Despite debate over whether wetlands have a positive or negative impact on carbon dioxide emissions, Taillardat et al. (2020) concluded the majority of inland wetlands have an overall cooling impact.

One of the most relevant and notable wetland ecosystem services to humans is the regulation of water levels and water quality (Kirby et al. 2002a; Zedler and Kercher 2005; Howell et al. 2012). Wetlands provide this service through taking up nutrients and metals into their soils (Kirby et al. 2002a; Howell et al. 2012; Yang et al. 2016). Vegetation absorbs materials into their tissues and back into the ground once the plant decays (Kirby et al. 2002a).

Through these services, wetlands filter out contaminates which would likely otherwise infiltrate into valuable water sources for humans. Adding to human benefits, wetlands help to decrease the impacts from peak flood events by holding large amounts of water, thus reducing property damages and expenses (U.S. Army Corps of Engineers 1994; Kirby et al. 2002a; Zedler and Kercher 2005; Downard et al. 2014; Yang et al. 2016). By lowering the intensity of peak flow events, residing streambanks are protected more from degradation and erosion caused by high water flows and velocities (Knight 1993). Flood water uptake, however, is dependent on the water holding capacity of the wetland and how much water is currently being held within the wetland.

The Degradation and Conversion of Wetlands

As a result of human demand and oceanic influences, wetlands have been depleted and degraded across the United States creating a great need to restore the systems to their once highly productive states (Dahl 2000; Zedler and Kercher 2005; Dahl 2011; Gascoigne et al. 2011; Qu et al. 2019). It is estimated between 2004 to 2009, wetlands within the US have been reduced by 5,590 ha per year (Dahl 2011). Across the globe wetlands have been reduced by 54-57% (Davidson 2014). Wetlands within the PPR are commonly isolated making them at great risk to impacts and alterations from agriculture (Kirby et al. 2002a; Tiner 2003; Gleason et al. 2005; Gleason et al. 2008; Gleason et al. 2011). Losses in connectivity between wetlands result in declines in ecosystem services and impacts the potential for restoration success (Dahl 2011; Gleason et al. 2011). Tiner (2003) attests isolated wetlands are still able to provide ecosystem services including wildlife habitat and the protection of manmade structures.

Within North Dakota, wetland losses were roughly 49% from the 1780s to 1980s (Dahl 1990; Gleason et al. 2008; Zilverberg et al. 2014). Due to agricultural demands, many wetlands have been degraded or converted for alternative land uses (Tiner 2003; Gleason et al. 2011; Downard et al. 2014; Zilverberg et al. 2014; Bartelt and Klver 2017; Guretzky et al. 2017; Altrichter et al. 2018), often depleting the ecosystem services they provide (Tiner 2003; Zilverberg et al. 2014; Bartelt and Klver 2017; Guretzky et al. 2017, Schultz et al. 2020). Conversions to cropland from Conservation Reserve Programs (CRP) in North Dakota are, in part, due to monetary enticements promoting corn production for biofuel (Dahl 2011). In North Dakota this led to losses of 12.4% of CRP land in 2007 alone. Harmful agents against wetlands often include tile drainage systems and improper usage of chemicals and soil additives. Cropped wetlands can be unsuccessful if hydrology is not controlled due to the variability in water levels diminishing crop yields (Tiner 2003; Zilverberg et al. 2014). The accumulation of naturally occurring salts may also hinder crop yields in wetland systems.

Wetlands have intricate and defining hydrological properties that are often modified (Zedler 2000; Gleason et al. 2011; Downard et al. 2014). Agricultural impacts in the form of tile drainage and other alterations to the water table can have negative impacts on the structure of native plant and animal communities within the wetland system (Dahl 1990; Ratti et al. 2001; Zedler and Kercher 2005; Gleason et al. 2008; Gleason et al. 2011; Hopple and Craft 2013; Buró et al. 2019). Within the Midwest, there has been a 50-90% decline in wetlands due to the impacts of tile drainage (Hopple and Craft 2013). Rare native plants are stifled when the natural hydrology of a wetland is changed, making it a key component in addressing impacts to wetlands (Galatowitsch and van der Valk 1996a; Hopple and Craft 2013; Zilverberg et al. 2014; Bartelt and Klver 2017).

Soil organic matter is lower in altered wetlands previously cultivated and used for crop production when compared to native wetlands, even following restoration (Gleason et al. 2008).

These fluctuations, as reported by Gleason et al. (2008), are between 12% to 26% within the PPR. Alterations to soil organic matter can then carry over into carbon sequestration levels. Gascoigne et al. (2011) predicted losses in native sites within the PPR would result in a net loss of $4 billion in ecosystem services, with specific considerations on reductions in carbon capture, waterbird habitat, and soil runoff capture. Zedler and Kercher (2005) valued wetland ecosystem services across the globe at roughly $13 trillion per year, with Schuyt and Brander (2004) valuing wetland service much lower, at $70 billion annually.

Invasive Species

A common source of degradation to wetland sites is the presence and invasion of introduced nonnative species (Zedler and Kercher 2005; Gleason et al. 2011; Kettenring and Adams 2011; Biró et al. 2019; Schultz et al. 2020). Zedler and Kercher (2004) found although wetlands only make up roughly 6% of earth's land mass, almost 25% of the earth's most notorious invasive species are listed as wetland plants. These species can limit site biodiversity, hinder ecosystem services, and create financial losses (Kettenring and Adams 2011; Biró et al. 2019). Restoration projects looking to control invasive species often struggle to find success due to high costs and difficulty in creating an inclusive plan able to not only control invasive species abundance but also promote native species. Invasive species, in general, establish and dominate due to rapid maturation that facilitates fast population expansions (Rehnánek and Richardson 1996). Within the PPR, invasive species such as *Phalaris arundinacea* (reed canarygrass) and *Typha spp.* (cattail) are often present in natural, restored, and degraded wetlands, creating challenges for restoration (Seabloom and van der Valk 2003a).

Reed Canarygrass

Species such as reed canarygrass tend to dominate degraded wetland sites and pose difficult management problems (Iannone et al. 2008; Strehlow et al. 2017; Weilhoefer et al. 2017; Matthews et al. 2020; Schultz et al. 2020). Agricultural sites may use reed canarygrass for erosion control and forage, aiding in the species spread (Clark and Thomsen, 2020). Reed canarygrass's competitive advantages allow it to stifle out native vegetation and reduce wetland biodiversity (Weilhoefer et al. 2017; Matthews et al. 2020; Sinks et al. 2021), especially within previously overused cropland (Weilhoefer et al. 2016; Clark and Thomsen 2020). This is partially because when competing with native species, reed canarygrass has an increased flexibility in resource allocation underground despite fluctuations in available levels of nutrients (Green and Galatowitsch 2001). Reductions in reed canarygrass help to facilitate dominance of native species, recover energy transfer relationships between organisms, and improve benthic populations (Ebberts et al. 2018; Sinks et al. 2021).

Control of reed canarygrass is necessary to restore or establish a biodiverse, native, wetland ecosystem. Even after management strategies are employed, reed canarygrass can reestablish quickly, minimizing the benefits of restoration efforts (Matthews et al. 2020; Sinks et al. 2021). Therefore, management strategies should consist of varied and combinable approaches able to be applied over an extended time period to have the best shot at decreasing reed canarygrass (Clark and Thomsen 2020; Sinks et al. 2021). The most effective strategies include combinations of native seedings, chemical control applications, prescribed burns, and tilling (Sinks et al. 2021).

Reintroductions of fast-growing native species able to occupy the land for long periods of time must occur immediately following control treatments of reed canarygrass to limit reinvasions (Iannone et al. 2008; Matthews et al. 2020). The use of native soil plugs and haying has exhibited positive impacts on native species establishment following treatments to control reed canarygrass (Clark and Thomsen 2020). This, however, is partially dependent on the native species already present at the site, which help to decrease the ability of reed canarygrass to dominate an ecosystem. Management objectives for controlling reed canarygrass should focus on reducing its competitive advantages to provide an outlet for native species to reestablish themselves (Perry and Galatowitsch 2006; Annen 2011; Sinks et al. 2021). Alterations to specific physical characteristics of a wetland leading to reductions in accessible nutrients, available sunlight, and increases in standing water help support wetland characteristics capable of hindering reed canarygrass productivity (Sinks et al. 2021).

Herbicide usage is a common practice to control reed canarygrass within wetlands (Hillhouse et al. 2010; Bonello and Judd 2020; Sinks et al. 2021). Herbicide applications are typically necessary to manage invasive species and can be used first in a restoration plan to control a large portion of the vegetation present (Aronson and Galatowitsch 2008). Applications of herbicide will likely need to be reapplied throughout the restoration duration due to the presence of invasive species seeds within the wetland system. Bonello and Judd (2020) analyzed the potential to use herbicide to control reed canarygrass over a period of 10 years and found following treatments, relative cover of reed canarygrass was significantly lower than unmanaged sites and plant species richness increased by two times.

To combat reed canarygrass, the use of fire and herbicide can be combined to reduce and/or eliminate the invasive species presence (Clark and Thomsen 2020; Sinks et al. 2021). A fall prescribed burn is one method that can be used in conjunction with herbicide applications to reduce the abundance of reed canarygrass (Ailstock et al. 2001). The use of fire in conjunction with herbicide applications help to reduce reed canarygrass litter and proved to be most successful when a quality native seedbank was present on site (Bonello and Judd 2020). The use of prescribed burning also provided greater benefits to floristic quality when used in addition to herbicide applications. Czarapata (2005) recommends carrying out herbicide applications of 3% glyphosate to decrease the competitiveness of reed canarygrass prior to burning.

The suitability of fall applications of prescribed burns depends on the amount of reed canarygrass present and water levels at the study site (Czarapata 2005). Fall burns carried out for five consecutive years or more are able to limit the amount of reed canarygrass (Hutchinson 1992; Czarapata 2005;). Burning wetlands can be difficult however, due to the potential to have an ineffective burn based on wetland water levels (Czarapata 2005; Bonello and Judd 2020). Further, the study site needs to have native species adapted to fire to compete with the previously burned reed canarygrass. Reinhardt and Galatowitsch (2005) concluded fall burns can reduce the seedbank capacity of reed canarygrass, however, provided little support for the ability of the burn to limit reed canarygrass over a long period of time. Penderrass et al. (1998) stated a fall burn application may decrease the height of reed canarygrass but will not limit its ability to regrow in the future. Prescribed burns play a key role in hindering monocultural stands of reed canarygrass through impairing the species feedback mechanisms that rely, in part, on the species dense litter that coats the soil surface (Zedler 2009; Annen 2011). Dense litter blocks other species from growing, and as more reed canarygrass grows, so does the litter layer, making it harder and harder for other species to establish.

Wetland livestock grazing can be an appealing option to landowners (Hillhouse et al. 2010; Guretzky et al. 2017). Livestock grazing may also be used to help control reed canarygrass if a high intensity grazing regime is established consisting of at least three events of disturbance to reed canarygrass occurring in a single year (Guretzky et al. 2017). Hillhouse et al. (2010) found grazing wetlands for two repeated spring seasons supports a reduction in the dense dead biomass layer reed canarygrass contributes yet did not result in declines of reed canarygrass abundance. Cleys (2019) analyzed the effect prescribed burns and grazing have on reed canarygrass-dominated wet meadows and concluded the use of grazing may limit the cover of reed canarygrass and supported increases in native plant production. However, results also indicated in the absence of reed canarygrass, other invasive species may move into its place if native species are not properly reintroduced. Grazing may be combined with prescribed burn practices to limit the amount of reed canarygrass in restored sites.

Kidd and Yeakley (2015) and Sinks et al. (2021) recommend reed canarygrass focused grazing paired with grazing exclusion for minimal time periods to help reduce reed canarygrass prevalence. Further, grazing can be used in conjunction with herbicide applications to provide better access for the herbicide to connect with live, growing reed canarygrass (Hillhouse et al. 2010). However, more long-term studies are needed to determine the most successful combination of these previously mentioned treatments and the associated costs and benefits of each (Green and Galatowitsch 2001; Hillhouse et al. 2010; Yang et al. 2016; Sinks et al. 2021).

Cattail

Another prevalent species within wetlands is cattail. Across the United States, three main species of cattail exist (Apfelbaum 1985; Bansal et al. 2019). These include *Typha latifolia* L. (broad-leaved), *Typha angustifolia* L. (narrow-leaved), and *Typha domingensis* Persoon (southern cattail), with hybrid variations existing. The hybrid variation of native and introduced cattail, *Typha* × *glauca* (hybrid cattail), is a prevalent invasive dominating many wetlands across North America (Ellstrand and Schierenbeck 2000; van der Valk 2005; Svedarsky et al. 2019).

Cattail's dominance is attributed to the species' ability to quickly spread, adapt, and propagate in new locations, maintain a strong allocation of above ground biomass, and create dense monoculture stands stifling competition (Solberg and Higgins 1993; Zapfe and Freeland 2015; Wilcox et al. 2018; Bansal et al. 2019). Anthropogenic influences account, in large part, for the distributional success cattail has had in asserting dominance within wetlands (Bansal et al. 2019).

Within North Dakota, in the PPR, cattail has been found to be present in roughly 23% to 49% of wetlands assessed (Ralston et al. 2007). Given the tendency of PPR wetlands to be located within and around agricultural operations, specifically farmlands, wetlands are more likely to receive increased inputs of fertilizer and soil runoff which can support cattail growth based on cattail's ability to rapidly allocate excess nutrients (Gleason et al. 2011; Wilcox et al. 2018; Bansal et al. 2019). Increases in sediment quantities within wetlands lower the amount of available water and alter seedling success of vegetation due to lower water levels (Gleason et al. 2011; Bansal et al. 2019). This decrease in water available to plants and corresponding diminished seedling success provides an outlet for cattail (Bansal et al. 2019). Native species within PPR wetlands can be further impacted by sedimentation changes leading to declines in soil carbon-based compounds and compaction (Werner and Zedler 2002; Gleason et al. 2011). Increases in human agricultural impacts and alterations to natural fire and grazing regimes leave North Dakota PPR wetlands more susceptible to invasions of cattail and other introduced species (Apfelbaum 1985; Gleason et al. 2011; Bansal et al. 2019;). Consequently, the ecosystem services of wetlands can be altered based on the changes cattail makes to a wetland's hydric regimes and present species (Bansal et al. 2019).

Cattail can negatively impact a variety of ecosystem services wetlands and their residing components provide (Bansal et al. 2019). Key impact points are centered around alterations to native plant species diversities and compositions throughout wetlands (Linz and Homan 2011; Wilcox et al. 2018; Bansal et al. 2019; Svedarsky et al. 2019). Additional impacts negatively influence wildlife dependent on the presence of consumable nutrient dense wetland species (Apfelbaum 1985; Kantrud 1986). Waterbirds, fish species, and macroinvertebrates reliant on adequate ponded water for habitats within marsh wetlands can be displaced by thick concentrations of cattail. Waterbirds are impacted by a diminished supply of high seed producing hydric species and a reduction in macroinvertebrates limited by the dominance of cattail (Bansal et al. 2019). One of the greatest focal points on influencing the desire to limit cattail presence is the species' impact on waterfowl populations (Apfelbaum 1985; Solberg and Higgins 1993).

Cattail can, however, provide value to human and natural entities through its medicinal, consumable, environmental, and practical usages. (Mitich 2000; Bansal et al. 2019). Cattail has been found to support habitats and cover for muskrats, pheasants, and white-tailed deer, as well as limiting soil erosion (Apfelbaum 1985; Kantrud 1992). The biomass of cattail can be used as a biofuel crop to support ethanol creation (Bansal et al. 2019). Cattail can filter out metals and nutrients specifically, in a process supporting phytoremediation (Bansal et al. 2019; Svedarsky et al. 2019).

When land managers determine control of cattail is needed, a variety of options are available. Synthetic herbicide appropriate for use within aquatic systems can be used to help manage cattail (Bansal et al. 2019). Chemical control methods of spraying are deemed to have the greatest success on cattail when carried out during time periods where cattail is currently in the process of maturing, most often towards the end of the summer (Solberg and Higgins 1993; Bansal et al. 2019; Svedarsky et al. 2019). Rodeo® (trademark of Monsanto Co., Inc., St. Louis, Mo.) glyphosate is a common chemical used in cattail control (Solberg and Higgins 1993; Linz et al. 2011; Bansal et al. 2019; Svedarsky et al. 2019). By limiting cattail success, native vegetation is given an outlet to make use of available water habitats (Solberg and Higgins 1993).

Grazing is another option to control cattail (Sojda and Solberg 1993; Kirby et al. 2002b). Grazing can be carried out on younger cattails to target rhizomes that are not fully matured (Sojda and Solberg 1993). Longer duration grazing in the later seasons can also take place on more developed cattails given coordination with water level controls. Grazing is often paired with prescribed burn treatments and takes place following the burn (Svedarsky et al. 2019). Appropriate pairings of prescribed burns and grazing can be successful in increasing heterogeneity of wetlands. Kirby et al. (2002b) attests wetland grazing paired with other disturbances such as fire, not only lead to production gains but also promotes greater species diversity supporting increases in feed production and value.

Native plants within the Northern Great Plains are well adapted to fire, therefore suppression of cattail by fire gives other plants an advantage in establishing themselves over cattail (Bansal et al. 2019; Svedarsky et al. 2019). Burning requires a focused approach with thorough considerations of water levels (Sojda and Solberg 1993; Svedarsky et al. 2019). For burns to be successful, water levels should be lower in order to reach an adequate fire temperature capable of stunting cattail growth (Apfelbaum 1985; Sojda and Solberg 1993; Bansal et al. 2019). If soils are not thawed enough there may be difficulties in maintaining a successful burn (Sojda and Solberg 1993). Apfelbaum (1985) indicates the value of lowering wetland water levels, carrying out a prescribed burn, and then flooding the area to depths of roughly 20 to 46 cm to reduce the vigor of cattail. An integrated approach using multiple treatment methods is highly encouraged, as cattail can have increased growth if prescribed burns are not carried out in conjunction with other treatments methods and with considerations of timing and water levels (Venne et al. 2016; Bansal et al. 2019; Svedarsky et al. 2019)

Carrying out an adaptive approach by employing multiple control methods based on site needs is typically necessary for successfully limiting the abundance and success of cattail based on the species ability to propagate quickly in unfavorable conditions (Wilcox et al. 2018; Bansal et al. 2019; Svedarsky et al. 2019). Considerations for the impacts management techniques have on the entire system should be made to ensure degradation of the wetland does not occur when employing treatments to control cattail (Apfelbaum 1985).

Wetland Restoration

Successful restoration projects yield improvements in carbon storage, vegetation diversity, and water quality (Schultz and Pett 2014). The restoration of wetlands can carry over into upland systems and increase the quality of vegetation present with regard to floristic quality (Gleason et al. 2008). Ecosystem services of restored cropped wetlands do not always meet the same level as natural undisturbed wetlands (Gleason et al. 2008; Schultz and Pett 2014). Seabloom and van der Valk (2003b) found vegetation compositions of restored wetlands were less heterogeneous at higher and lower elevations when compared to natural wetlands. Upland sites may be impacted by increased homogeneity at higher elevations through the spread and dominance of *Bromus inermis* (smooth brome). Wetland restorations near a natural wetland have increased connections to desirable species and therefore have been found to have higher species richness in comparison to isolated wetlands. However, the benefit of connectivity between degraded and natural wetlands is hindered by declines in water pathways between wetlands and the common dominance of invasive species within soil seedbanks (Galatowitsch 2006; Gleason et al. 2011). Species composition is a key component in the proper functioning of wetlands and their ability to provide ecosystem services, specifically relating to carbon cycling (Schultz et al. 2011). In some cases, species composition can be a stronger indicator of wetland health than soil properties or species diversity, highlighting the necessity for restoration techniques to consider vegetation interactions and make up.

Restoration Cost

The expenses associated with wetland restoration are key points of consideration (Qu et al. 2019). Much of the present management plans pertaining to wetland restoration use passive restoration strategies as they are not as likely to be constrained by cost (Birge et al. 2016). Management plans need to be cost effective while still ensuring restoration goals are met (Yang et al. 2017).

Yang et al. (2017) attributed the cost of wetland restoration over a 12-year evaluation period to be on average $132.4/ha/yr for the full wetland restoration. A modeling system which assessed associated and expected extra expenses, restoration conduction cost, and various logistical components was used. In contrast, the restoration of inland fen and bog wetlands for carbon sequestration services over various time scales fell between $464ha/yr to $37,173 ha/yr with a median value of $1,229ha/yr (Taillardat et al. 2020). Zentner et al. (2003) put wet meadow restoration costs at $40,772/ha for a baseline of grading and planting treatments. The expenses did not include costs of acquiring land, accounts for all the legal aspects of the restoration, or follow up techniques. Strehlow et al. (2017a) indicated the complete cost of restoration treatments totaled out at $1,963/ha for native seeding only; $2,342/ha for native seed and hay mulch; and $5,145/ha for native seed, hay mulch, and transplanted soil plugs, with the cost of hired work was valued at $12/hour and supplementary components added to the evaluated expenses associated with pretreatments. Individual treatments of seeding, hay mulching, and soil plugs were valued at $1,143/ha, $379/ha, and $2,803/ha, sequentially. The seed only had the greatest cost to benefit ratio as no significant differences existed between treatments.

Variations in restoration costs can be attributed, in part, to sampling protocols and the intensity of restoration methods used. Additionally, restoration expenses can be lessened by selecting sites historically greater in species biodiversity (Qu et al. 2019). Taylor et al. (2013) assessed the ability to incorporate grazing, herbicide, and seeding to increase the transitional time period of a site previously used for cropping. Results indicated this moderately lower costing restoration project, $625/ha, promoted a rise in native grasses and reduced the presence of introduced grasses.

Matthews et al. (2020) studied the impact different afforestation restoration methods would have on a wetland and their associated cost benefits. The study site was previously used as cropland and consisted of invasive plants, including reed canarygrass. Fifteen years after the initial restoration, Matthews et al. (2020) found larger initial cost restoration methods such as balled-and-burlapped tree plantings, provided greater ecosystem benefits, including increases in plant densities and richness, as well as a decline in invasive species such as reed canarygrass. Matthews et al. (2020) concluded each $10,000 added to the project's restoration methods, resulted in a subsequent increase in success, therefore warranting the greater startup expenses.

Restoration Treatments

Seeding

Seeding native species, as a method of wetland restoration, can support the rehabilitation of plant communities through improving plant community diversity while also reducing the ability of introduced species to find success within the system (Lindig-Cisneros and Zedler 2002; Hopple and Craft 2013). Restoration techniques focused on the removal of invasive species followed by native species plantings have been found to create restored wetlands more similar to natural wetlands at a faster pace (Galatowitsch 2006). Based on difficulties in the spread of some native species, seeding in combination with the reinstatement of natural water regimes may provide greater success through its more integrated approach (Seabloom and van der Valk 2003a; Hopple and Craft 2013).

Following successful seeding treatments, vegetation propagules can spread beyond their initial planting site to promote an increased presence of the seeded plant species across the site, therefore, reducing the need for additional treatments (Rayburn and Laca 2013). Wetlands with high plant species diversity and a strong presence of native species provide an increase in wetland efficiency and their ecosystem services (Hopple and Craft 2013). This translates into higher speeds of breaking down organic matter, increases in the transfer of nutrients, and greater levels of resiliency to environmental stressors. These benefits also improve fauna health. Biodiverse wetlands providing ample native seed can be a source of revenue for landowners (Johnson 2019).

Aronson and Galatowitsh (2008) deemed seeding of native vegetation crucial for a positive restoration outcomes and recommended seeding at the beginning of the restoration to help support establishment over invasive species. Lindig-Cisneros and Zedler (2002) and Sinks et al. (2021) encourage selecting site-specific seed capable of competing with invasive species, with Sinks et al. (2021) highlighting the native species' abilities to create numerous intricate canopy levels on the soil surface and diverse underground networks. Native species, once established, can thwart off reed canarygrass dominance through their ability to uptake nutrients needed by reed canarygrass (Seabloom et al. 2003; Iannone et al. 2008). Leps et al. (2007) highlights the value of curating seed blends high in diversity, as the larger number of species can counteract failures of species within the mix to establish. However, it is worth noting the greater cost associated with selecting native seeds specific to a certain region given their rarity and limited use in agricultural practices (Rayburn and Laca 2013).

Soil Plugs

Soil seedbanks are a collection of dormant and viable seeds that are often a product of both current and past aboveground vegetation (Roberts 1981; Thilmony and Lym 2017). The soil seedbank present can have a massive impact on the success of a restoration project (Suding et al. 2004; Strehlow et al. 2017b; Thilmony and Lym 2017). To recreate a biodiverse plant community within wetlands, considerations for the present seedbank are important given some degraded wetlands may have diminished available native seeds present (Wienhold and van der Valk 1989; Van der Valk 2013; Schlutz and Pett 2014; Strehlow et al. 2017b). Without adequate seedbanks, restoration efforts may be hindered as seedbanks have the potential to be dominated by invasive nonnative species (Adams and Galatowitsch 2006; Schlutz and Pett 2014). Choices of seed propagates should reflect native vegetation to the region of study (Galatowitsch 2012; Schultz and Pett 2014). Vegetation selected must be able to handle the region's specific biotic and abiotic characteristics while still being financially feasible to obtain (Suding et al. 2004; Galatowitsch 2012; Schultz and Pett 2014). Within wetland restorations, the use of soil plugs and seeding in conjunction with each other can lead to greater native vegetation diversity, richness, and overall treatment success as opposed to the use of seed alone (Middleton et al. 2010; Strehlow 2015).

Haying

Hay additions can supplement restoration actions by acting as an outlet for the transport of native vegetation (Rasran et al. 2009; Foster et al. 2009) that can be potentially hindered by the plant's minimal dispersal abilities (Galatowitsch & Van der Valk 1996). Hay transfer involves transporting plant material from one site to another, with a focus on hay high in viable diverse native seed (Rasran et al. 2009). Haying can also offset the additional costs often associated with applying larger quantities of fertilizer (Foster et al. 2009). Combinations of seeding and haying support in the reduction of weedy species, while also providing valuable cover on the soil surface protecting native seed (Foweler 1988; Török et al. 2012). By covering the soil surface with hay, native plants requiring a greater investment in nutrients for growth are provided an outlet for germination with less potential for losses of moisture and competition from weedy plants due to the reduced light penetration (Foster and Gross 1988; Foweler 1988; Török et al. 2012). Haying also facilitates more consistent soil temperatures, dampening the environmental trigger temperature raises cue for weedy plant growth (Wedin and Tilman 1993; Foster and Gross 1998). Further, research utilizing wetland hay for grassland restoration demonstrated hay applications provide slight improvements in plant community composition (Foster et al. 2009).

Prescribed Burns

In the PPR the use of prescribed burns can help to replicate past historical disturbance regimes present within wetland systems (Venne et al. 2016; Johnson 2019). Prescribed burns support increases in light penetration into the system and improve the availability of nutrients (Venne et al. 2016). Improvements can also occur in plant composition and plant development (Suding et al. 2004; Venne et al. 2016). Johnson (2019) indicates the necessity of utilizing prescribed burns, haying, and grazing in conjunction with each other to have a successful outcome. Suding et al. (2004) and Hopple and Craft (2013) also encourage the use of prescribed burns paired with native seedings to create a more successful outcome than the use of those treatments individually.

Grazing

Wetland grazing is another possible management option, one which provides large amounts of benefits to land managers (Kirby et al. 2002a; Hillhouse et al. 2010; Guretzky et al. 2017). Through limiting the use of drainage and chemical additives by using wetlands for grazing, land managers can maintain the integrity of the wetland, without having to take the land out of commission (Kirby et al. 2002a; Guretzky et al. 2017). Benefits also spread beyond land managers and provide for local wildlife with increased habitat quality (Kirby et al. 2002a; Hillhouse et al. 2010; Guretzky et al. 2017). Livestock disturbances play a dominant role in altering the microclimate and relationships between biota within wetlands, with prevalent impacts on plant dynamics and their gradual change in community structure over time (van der Valk 1981, Zedler and Kercher 2005, Biró et al. 2019). These alterations to the system may promote improvements in heterogeneity across vegetation and habitats (Davidson et al. 2017; Biró et al. 2019). Biró et al. (2019) highlighted the difference between excluded and grazed wetlands, indicating the decline in the number of plant species within excluded wetlands and necessary value livestock can have on promoting higher species richness in wetlands.

Incorporating methods of prescribed burns into grazing regimes help to provide quality forage for livestock through the reduction of vegetation less palatable to livestock (Kirby et al. 2002b). The restoration of wetlands can help to improve the native floristic composition to be more beneficial to livestock grazing through decreases in species such as cattail which have lower digestibility (Kirby et al. 2002b; Johnson 2019). For grazing regimes to be successful they must be tailored to the maturity of vegetation within the wetland, as vegetation is more digestible prior to reaching full development. A commonly recommended grazing practice employs the principal of "take half, leave half" to ensure native and desirable vegetation is not over utilized (Johnson 2019). Targeted grazing (Kidd and Ueakley 2015) and high intensity grazing (Hillhouse et al. 2010) have been shown to increase total average species richness when compared to wetlands excluded from grazing. A key focus of wetland grazing for restoration, especially in invasive dominated systems, is the impact livestock can have on opening the litter layer above the soil surface (Hillhouse et al. 2010). The removal of dead biomass supports the growth of more desirable annual vegetation species and encourages greater rates of plant growth in the absence of excess litter (Wilby and Brown 2001; Williams et al. 2007; Hillhouse et al. 2010).

Challenges in Restoration

Defining the success of a wetland restoration project is often a difficult endeavor (Kentula 2000; Zedler 2000; Seabloom and van der Valk 2003b). Wetland restorations are expensive projects and therefore often require appropriate validation for high costs (Zedler 2000). Increasing ecosystem services is a common goal for restoration projects, however, this is difficult to measure. The ecosystem services a wetland provides following initial vegetation establishment may not equal the same level as the wetland provided prior to degradation (Hossler and Bouchard 2010; Moreno-Mateos et al. 2012). The term "success" is also often arbitrary and ambiguous because there can be many goals and methods associated with defining an effective restoration project (Kentula 2000; Zedler 2000). It is crucial to have a clear understanding of a restoration projects' goals to ensure proper monitoring and data are attained to validate and define project outcomes and statues. Wetland restorations are often assessed on the basis of rehabilitating hydrological regimes and improving plant species biodiversity to further the sites' ability to provide beneficial services to both wildlife and humans. Kentula (2000) argues for a more inclusive and interchangeable study design promoting a looser definition of success to allow for an increased incorporation of adaptive management. This step away from a stringent study design will create an environment more representative of the natural systems fluctuations and will, in theory, set a path towards increased site rehabilitation.

Adaptive and Post Management in Wetland Restoration

There is a great difficulty when it comes to achieving specific management goals, while also accounting for the associated expenses and allotted time (Matthews et al. 2020). Due to this, proponents for passive restoration methods argue this is the most time and cost-effective solution, despite the potentials for the management to be unsuccessful. Passive restoration looks to reduce or stop environmental stressors whereas active restoration employes more intensive management techniques such as seeding or prescribed burns (Morrison and Lindell 2010). There are, however, instances where passive restoration is successful and may be a better option than active restoration.

Wetlands experiencing influences from diminished flooding, geographic separation (Suding et al. 2004; Aronson and Galatowitsch 2008), and/or domination by invasive species (Zedler 2000; Kidd and Yeakley 2005; Matthews et al. 2020), may require more intensive methods for recovery as they may not recover on their own within a reasonable timeframe (Suding et al. 2004; Aronson and Galatowitsch 2008). Wetland restoration sites may be sensitive to changes in vegetation quality following treatments and require long term attention to ensure a regression back their previous state does not occur (Aronson and Galatowitsch 2008; Gleason et al. 2008) Furthermore, dependent on a specific project goal, such as adhering to legal mandates, intensive restoration may increase the likelihood of achieving these goals (Zedler 2000; Matthews et al. 2020).

Natural systems have many functions with a limited potential to be controlled and possible unintentional responses (Kentula 2000; Yang et al. 2016; Birge et al. 2016). This potential for uncertainty within a system can pose a threat to the success of management plans due to the uncertainty of responses it creates (Kentula 2000; Zedler 2000; Birge et al. 2016). Galatowitsch and van der Valk (1996) assessed the potential for wetlands within PPR to be restored by reinstating the natural water regimes through the removal of tile drainage and included no further efforts of revegetation. Comparisons between restored wetlands and natural wetlands demonstrated key difference still existed between the vegetation present. Similarly, Seabloom and van der Valk (2003a) assessed differences in species presence and distribution between natural wetlands and wetlands restored through the reestablishment of hydrological regimes. Results indicated natural wetland and restored wetlands had differences in both plant diversity and distribution, with the restored wetland presenting lower values (Seabloom and van der Valk 2003a).

In contrast, Hopple and Craft (2013) assessed the potential to employ multiple adaptively managed treatment methods including burns, seeding, and chemical additions to improve the biodiversity of wetlands. Results illustrated their restored wetlands had similar species richness and Floristic Quality Assessment Index values to that of natural wetlands yet were still falling short in the quality of the wetland's plant species. Improvements in species richness were hypothesized to be resulting from the adaptive approach applying stress on introduced species not adapted to the area's natural disturbance regimes. Hopple and Craft (2013) deemed more research was needed to address implications of actively managing restoration sites.

Methods

Wetlands provide meaningful ecosystem services and are valuable assets to land managers. The mismanagement and degradation of wetlands threatens to reduce their functionality. Through restoration practices, degraded wetlands may be restored. However, limited research is present accounting for the actual long-term costs associated with various wetland restoration practices and their corresponding effectiveness. The objective of this restoration project is to improve native plant species richness, diversity, and cover, while also reducing the dominance of invasive species. To assess treatment responses, data was collected on total species richness, native species richness, introduced species richness, total cover, reed canarygrass relative cover, native relative cover, and introduced relative cover. Through evaluating response variables, researchers can assess whether treatments are significantly different across multiple metrics to provide insight on the most cost-effective treatments for restoring highly degraded wetlands.

Study Site

The study site is a wetland, previously used as cropland in Richland County (46°31'34.41"N, 97° 7'56.16"W) southeastern North Dakota. The site is 18.86 ha located on the Albert K. Ekre Grassland Preserve, which was historically native tallgrass prairie. The study site is in MLRA 56, Red River Valley of the North (USDA-NRCS 2006). The region supports natural prairie vegetation historically including *Andropogon gerardii* Vitman (big bluestem), *Panicum virgatum* L. (switchgrass), *Sorghastrum nutans* (L.) Nash (Indiangrass), and *Schizachyrium scoparium* (Michx) Nash (little bluestem). Soil orders within this MLRA are dominated by Mollisols and Vertisols with most of the region used for private cropland that accounts for 79% of the land use. Hydrologically the region is mainly influenced by the Red River with further impacts from the Sheyenne River, its biggest tributary.

The McLeod 3E's climate station is the closest to the site. The average annual temperature over the last 30 years (1991-2020) is 5.67°C, with the average maximum temperate being 11.83°C, and the average minimum temperature being -0.44°C (NOAA 2021). Average annual rainfall is 60.93 cm, with peak precipitation occurring during the summer. Flooding is a common occurrence based on the level topography, leading many farmers to utilize drainage technologies (USDA-NRCS 2006; NOAA 2021).

The study site's soils are composed of an Alymer-Rosewood-Serden complex with slopes ranging from 0 to 9% (USDA-NRCS 2021). The Alymer soil is classified as mixed, frigid Aquic Udipsamments with very deep, moderately well drained and quickly permeable soils. The Alymer soil originates from wind outwash plains and delta plains and has a high water table during the spring months. The Rosewood soil is classified as sandy, mixed frigid Typic Calciaquolls with highly limited drainage capacities. The Serden complex is classified as mixed, frigid Typic Udipsamments with excessive drainage potential and high permeability.

Farming on the land took place for decades, however, due to the hydrologic regimes causing large variations of water levels, farming was not a financially viable option in recent years. Drainage has not been used within the study site. As a result of a history of farming and its current absence, the site contained a large abundance of weedy and invasive species able to flourish on the unsowed land.

The site now is used as an experimental wetland restoration site employing various techniques to restore native plant biodiversity and control invasive species. Treatments began in 2013, when a randomized complete block design experiment was created. The study plots cover

11.04 ha separated into nine plots of roughly the same size (1.23ha) and organized by three rows of three (Strehlow 2017a). Each treatment was repeated three times and replicated once per block shown in Figure 3.1. Treatments of 1) Seed, 2) Seed + mulch, 3) Seed, mulch, + transplanted soil plugs, were applied once per block. Impacts of the three treatments were closely monitored to determine their associated costs to analyze their respective success towards reaching restoration goals (Strehlow 2017a).

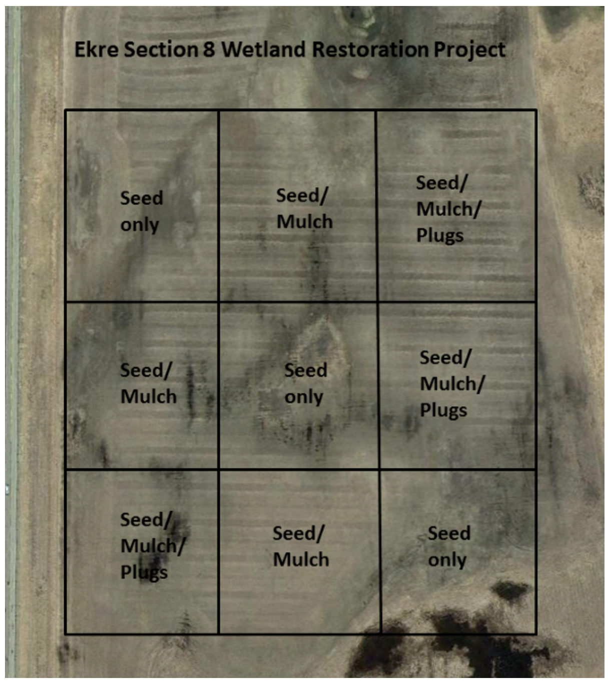

Figure 3.1: Ekre Section 8 wetland restoration treatment configurations (Strehlow 2015).

Prior to installing study plots, the entire site was burned in the spring of 2013 and Salford vertical tilled to reduced hummocks and mounds present within the site. The site was then planted with Roundup Ready ® soybeans during the growing season. Herbicide (Roundup PowerMax®, Monsanto Company, St. Louis, MO, 63167) treatments were also carried out, with one application following the prescribed burn, three applications while Roundup Ready ® soybeans were in cropping, and one following the soybean harvest. Two more applications of herbicide were applied in the spring prior to seeding. This was done to limit the presence of weedy and invasive species, increasing the success of initial treatment applications. Following herbicide use, native seed (Table 3.1) and soil plugs were applied at the study site during the summer and fall of 2014.

Table 3.1: Species mix and percent composition for restoration seeding.

Large Seed Mix **Variety Name**	**Scientific Name[a]**	**Percent Mix**
Certified Rosana Western Wheatgrass	*Pascopyrum smithii*	4.91%
Certified Revenue Slender Wheatgrass	*Elymus trachycaulus*	4.84%
Bison Big Bluestem	*Andropogon gerardii*	20.34%
Pierre Sideoats Grama	*Bouteloua curtipendula*	11.42%
Bad River Blue Grama	*Bouteloua gracilis*	0.62%
Goshen Prairie Sandreed	*Calamovilfa longifolia*	8.30%
Certified Mandan Canada Wildrye	*Elymus canadensis*	5.75%
Sunburst Switchgrass	*Panicum virgatum*	7.10%
Itasca Little Bluestem	*Schizachyrium scoparium*	6.12%
Tomahawk Indiangrass	*Sorghastrum nutans*	9.44%
Red River Prairie Cordgrass	*Spartina pectinata*	4.82%
Needle and Thread	*Hesperostipa comata*	4.14%
Small Seed Mix **Variety Name**	**Scientific Name[a]**	**Percent Mix**
MN Native Purple Prairie Clover	*Dalea purpurea*	47.60%
American Sloughgrass	*Beckmannia syzigachne*	5.29%
AK Native Canada Bluejoint	*Calamagrostis canadensis*	0.60%
SD Native Slimstem Reedgrass	*Calamagrostis stricta*	2.69%
IA Native Prairie Sedge	*Carex prairea*	9.72%
IA Native Plains Oval Sedge	*Carex brevior*	0.34%
SD Native Pale Sedge	*Carex pallescens*	3.97%
WI Native Porcupine Sedge	*Carex hystericina*	4.64%
SD Native Smoothcone Sedge	*Carex laeviconica*	1.04%
MN Native Woolly Sedge	*Carex pellita*	0.56%
Brown Fox Sedge	*Carex vulpinoidea*	3.90%
OR Native Creeping Spike Rush	*Eleocharis fallax*	3.00%
SD Native Reed Manna Grass	*Glyceria maxima*	2.13%
Prairie Junegrass	*Koeleria macrantha*	0.29%
MN Native Green Muhly	*Muhlenbergia ramulosa*	0.45%
SD Native Pale Bulrush	*Scirpus pallidus*	0.32%
SD Native Three Square Bulrush	*Schoenoplectus pungens*	6.35%
SD Native Sand Dropseed	*Sporobolus cryptandrus*	0.62%

[a]Species names are from The PLANTS Database: USDA, NRCS. 2021. The PLANTS Database (http://plants.usda.gov, 29 December 2021). National Plant Data Team, Greensboro, NC 27401-4901 USA.

Vegetation surveys have been conducted annually, from 2015 through 2021. At each plot transects are placed by adding 90, 180, and 270 degrees to the initial bearing, with initial plot bearings generated randomly. Thirteen 1-square meter quadrats were sampled per plot, with three quadrats extended in all four directions, originating from the center point quadrat. The quadrats were placed at 8, 16, and 24 meters from the center of the plot, as shown in Figure 3.2.

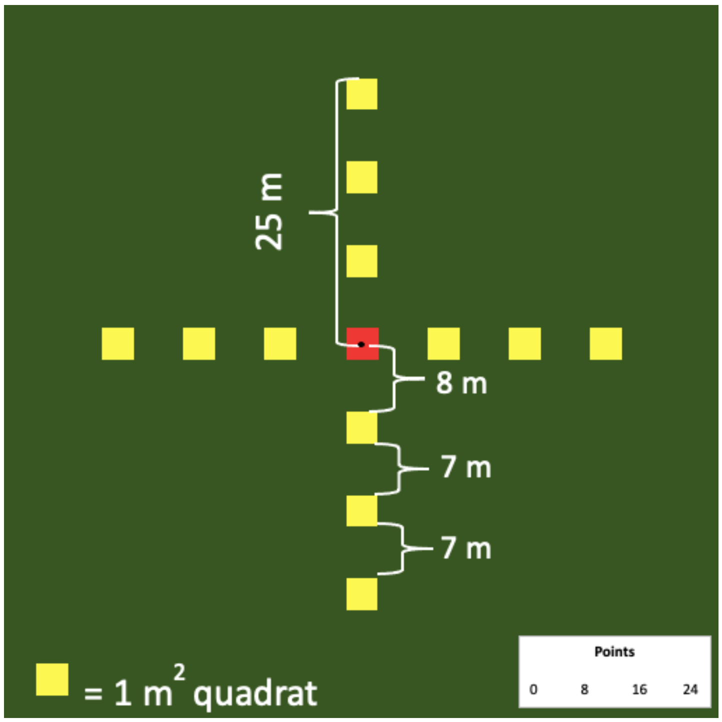

Figure 3.2: Vegetation survey design with quadrats and distancing.

Vegetation surveys were conducted in each 1 m^2 quadrat using cover classifications of each species present in the quadrat beginning in 2015 until 2020. Bareground and litter were also assigned a corresponding cover class for each quadrat. A total of 13 quadrat measurements were conducted per treatment plot. Cover classes were based on Daubenmire (1959) classification codes ranging from one through six with respective percent cover classes being 0-5; 5-25; 25-50; 50-75; 75-95; and 95-100. In the summer of 2021, vegetation surveys were conducting using exact percent cover estimates instead of relative percent cover classes. Vegetation was surveyed in each 1 m^2 quadrat and specific percentages were assigned to each species within the quadrat with additional percentages given for bare ground and litter. Vegetation survey methods were altered in 2021 to exact percent cover estimates in hopes of assessing vegetation with a greater degree of precision to account for smaller differences in individual species presence within quadrats. Both initial site evaluations of the wetland comparing 2016 data alone (Strehlow et al. 2017a; Strehlow et al. 2017b) and more recent evaluation in 2019 accessing data across the full study duration (Durant 2020) have found no significant differences between treatments across all metrics assessed. This analysis focused on the 2021 data to determine specific wetland vegetation responses to treatments.

Data was analyzed with a one-way ANOVA employing Tukey's student range test ((SAS Enterprise Guide 7.1 (Copyright © 2017 by SAS Institute Inc. Cary, NC, USA)). Data was not transformed for the ANOVA analysis based on the skewness being less than zero, plus or minus one. The dependent variables were total species richness, native species richness, introduced species richness, total cover, reed canarygrass relative cover, native relative cover, and introduced relative cover. The independent variable was treatment type (seed, seed + mulch, and seed, mulch + soil plugs).

Using adaptive management, follow up treatments of burning, grazing, and herbicide applications will be applied to the study site to help control reed canarygrass and promote native species. These treatments will be conducted on the basis a large portion of restoration plans do not typically include details of a post management strategy. The post management plan was deemed to only begin following five years after initial treatment. Prominent impacts to the wetland include a wildfire that burned the entire study site on May 9th, 2017 and the implementation of rotationally grazed cow/calf pairs since early 2017.

Results

Total species richness, native species richness, introduced species richness, total plant cover, native species cover, introduced species cover, and reed canarygrass cover variables were summarized across treatments (Table 3.2).

Table 3.2: Total species richness, native species richness, introduced species richness, total plant cover (%), native species cover (%), introduced species cover (%), and reed canarygrass cover (%) across seed only (S), seed + mulch (SM), and seed + mulch + soil plugs (SMP) treatments.

Treatment	Total Species Richness	Native Species Richness	Introduced Species Richness	Total Plant Cover	Native Species Cover	Introduced Species Cover	Reed Canarygrass Cover
S	32.00	24.67	7.33	37.18	63.44	36.56	24.34
SM	28.33	21.33	7.00	32.96	56.20	43.8	16.70
SMP	31.33	25.00	6.33	34.03	73.88	26.12	9.52
Average Across Treatments	30.55	23.67	6.89	34.72	64.51	35.49	16.85

There were no significant differences ($P < 0.05$) in species richness ($F_{2,6} = 1.45$, $P = 0.3065$) (Figure 3.3), native species richness ($F_{2,6} = 2.71$, $P = 0.1452$) (Figure 3.4), introduced species richness ($F_{2,6} = 0.17$, $P = 0.847$) (Figure 3.5), total plant cover ($F_{2,6} = 1.85$, $P = 0.2363$) (Figure 3.5), reed canarygrass cover ($F_{2,6} = 1.546$, $P = 0.287$) (Figure 3.7), native relative cover ($F_{2,6} = 0.87$, $P = 0.464$) (Figure 3.8), or introduced relative cover ($F_{2,6} = 0.87$, $P = 0.464$) (Figure 3.9) between treatments.

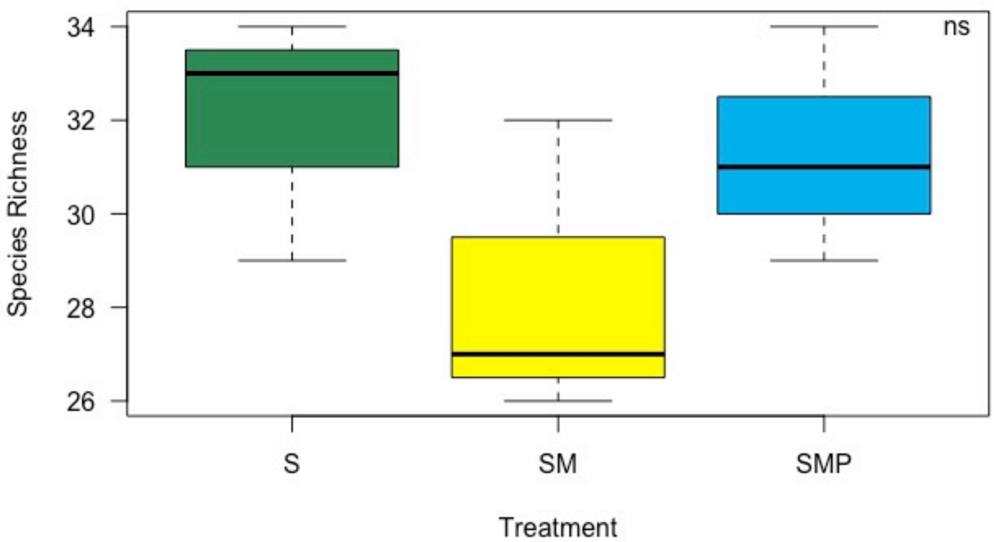

Figure 3.3: Average species richness (± standard error) by restoration treatment. Average species richness was not significantly different across seed, seed + mulch, and seed + mulch + soil plugs treatments in 2021.

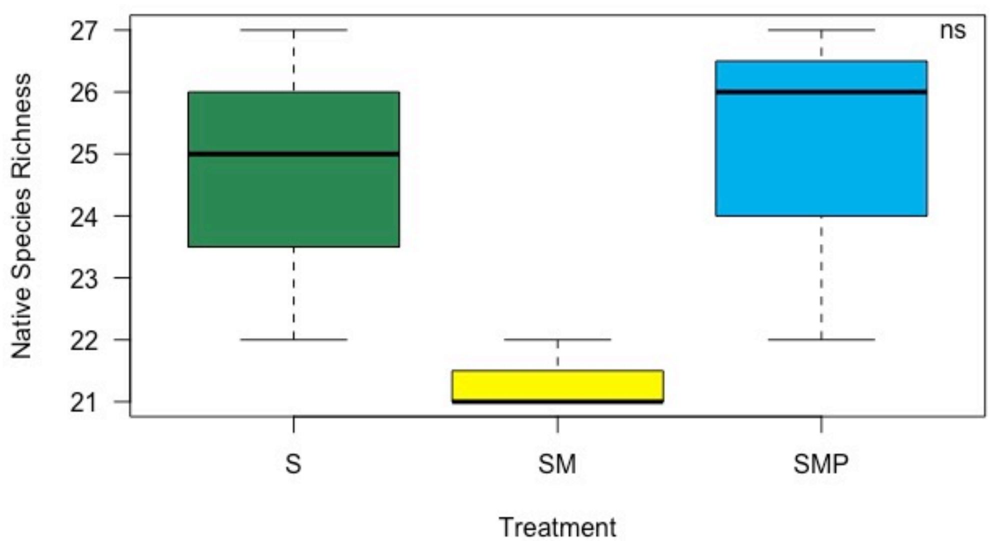

Figure 3.4: Average native species richness (± standard error) by restoration treatment. Average native species richness was not significantly different across seed, seed + mulch, and seed + mulch + soil plugs treatments in 2021.

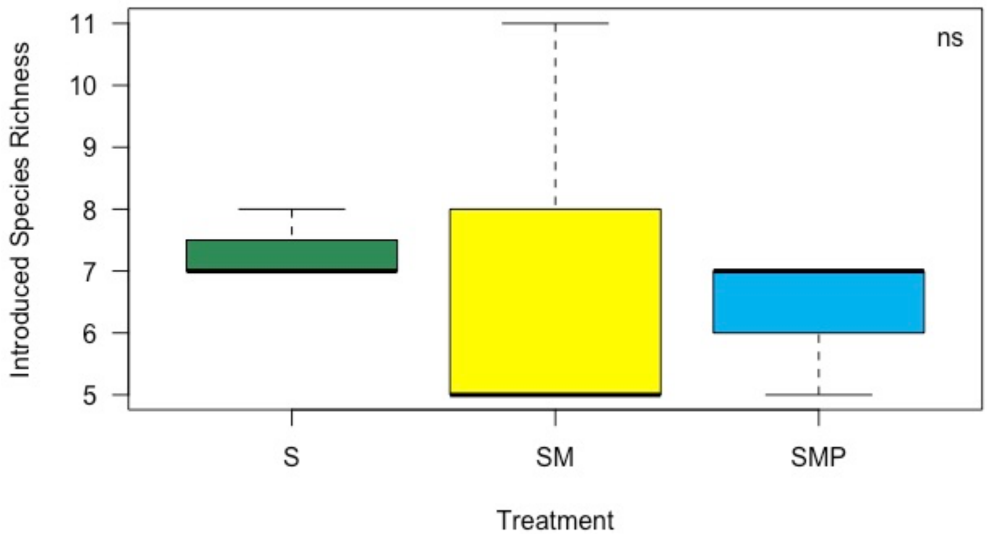

Figure 3.5: Average introduced species richness (± standard error) by restoration treatment. Average introduced species richness was not significantly different across seed, seed + mulch, and seed + mulch + soil plugs treatments in 2021.

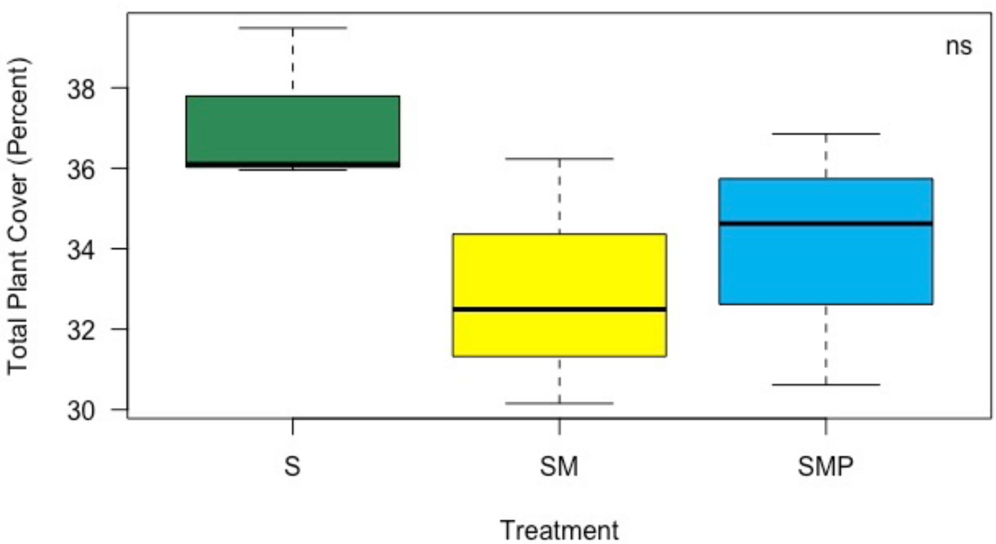

Figure 3.6: Average total plant cover (± standard error) by restoration treatment. Average total plant cover was not significantly different across seed, seed + mulch, and seed + mulch + soil plugs treatments in 2021.

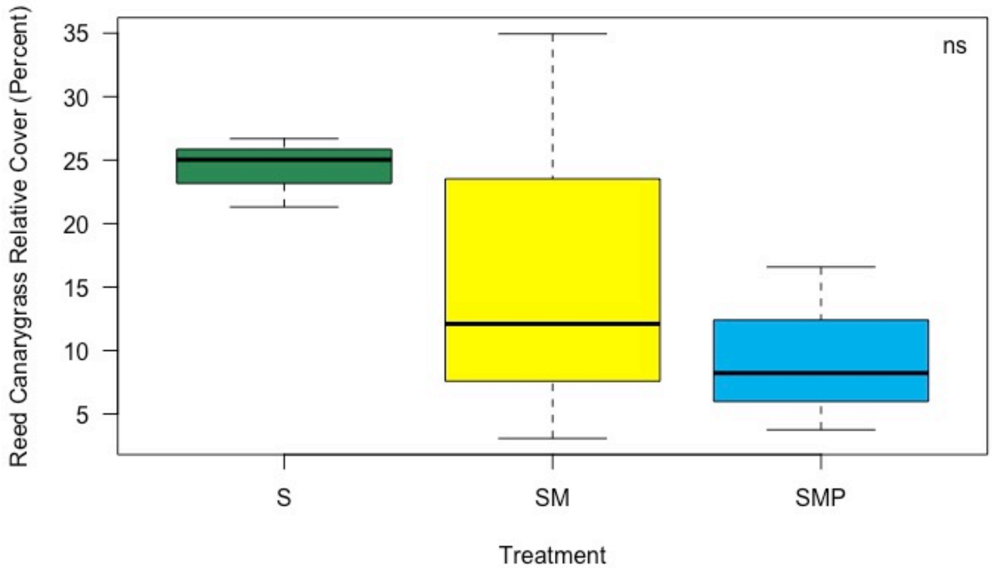

Figure 3.7: Average reed canarygrass relative cover (± standard error) by restoration treatment. Average reed canarygrass relative cover was not significantly different across seed, seed + mulch, and seed + mulch + soil plugs treatments in 2021.

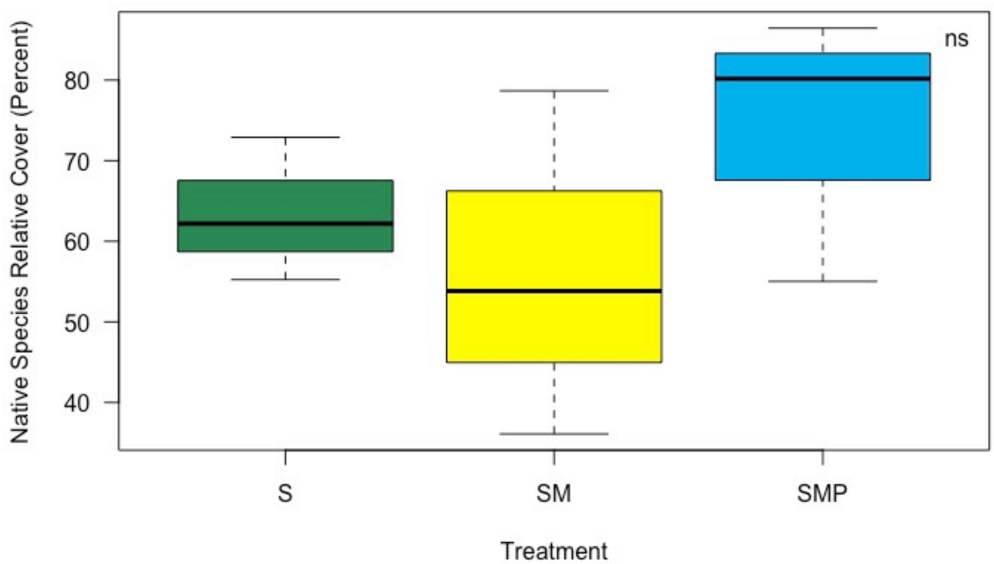

Figure 3.8: Average native species relative cover (± standard error) by restoration treatment. Average native species relative cover was not significantly different across seed, seed + mulch, and seed + mulch + soil plugs treatments in 2021.

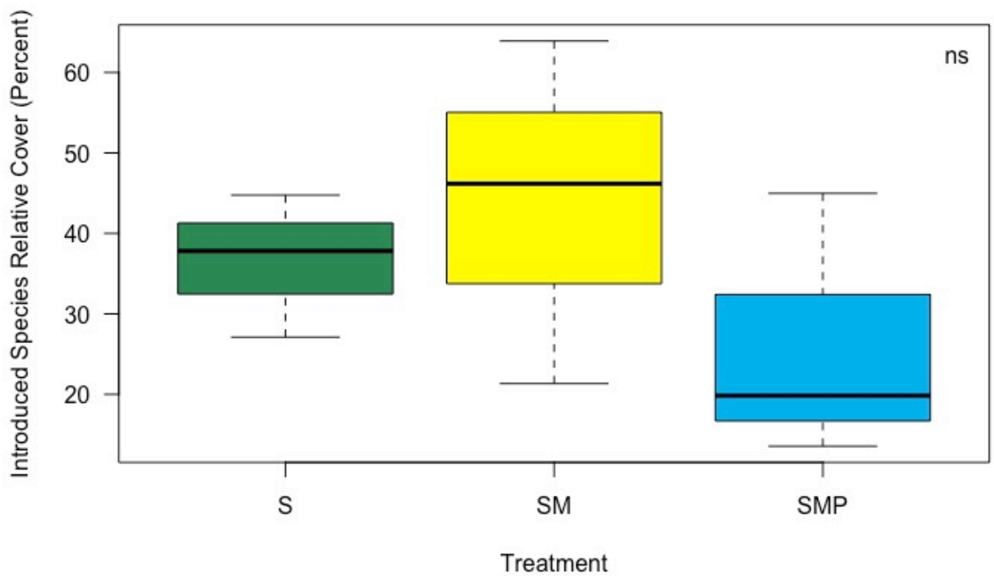

Figure 3.9: Average introduced species relative cover (± standard error) by restoration treatment. Average introduced species relative cover was not significantly different across seed, seed + mulch, and seed + mulch + soil plugs treatments in 2021.

Discussion

Creating a restoration plan capable of reaching specific goals can be extremely difficult when working in an ecosystem that is not only isolated but also dominated by invasive species (Galatowitsch and van der Valk 1996; De Steven et al. 2010). Therefore, it is increasingly important to understand why restoration projects may not be reaching desired outcomes to provide future land managers metrics to base their plans on when working in similar conditions. The goal of the wetland restoration project was to improve native plant species richness, diversity, and cover while also reducing the presence of invasive species using high intensity treatments. However, higher intensity treatment applications have yet to display any significant differences amongst treatments in the eight years since restoration began. It should be noted, wetland conditions have moved away from reed canarygrass and cattail monocultures prior to restoration, towards greater diversity, cover, and richness of native species presently. Improvements away from initial site conditions highlight the value of treating reed canarygrass and cattail dominated wetland sites with a spring prescribed burn, followed by planting Roundup Ready ® soybeans, and issuing seven rounds of herbicide applications prior and during the restoration (Strehlow 2015). The study site can still make marked improvements into the future as treatments continue to impact the landscape through their interactions spatially and temporally (van der Valk, 1981; Biró et al. 2019).

Given ecological restoration works in dynamic systems requiring time to progress successional characteristics, patience is needed (van der Valk 1981; Howell et al. 2012; Biró et al. 2019). As a study site shifts further into an altered state, the duration of time needed to get back to pre-disturbance conditions increases (Jørgensen 1994; Zedler and Kercher 2005). Wetlands with poor starting conditions could require decades to reach project goals (Jørgensen 1994; Zedler and Callaway 1999; Wilkins et al. 2003; Hilderbrand et al. 2005; Zedler and Kercher 2005). Aronson and Galatowitsch (2008) indicate the 12 years following initial restoration actions are the most valuable in establishing a desired vegetation make up, as this provides time for species to establish with few plants being added to the wetland system.

In contrast, Galatowitsch and Bohnen (2021), reiterate systems with a limited ability to handle stress may not transition to the desired recovery state simply through the progression of time. This wetland study site can, in this context, be considered one with a limited resilience given the dominance of invasive species, and based on this theory, time may not be a valid reason for the lack of differences between treatments. The strong presence of invasive species, especially reed canarygrass, within the wetland restoration has established a state of positive feedback for the invasive species and reduced the ability of treatments to perform to their full potential and produce the desired outcomes (Zedler and Kercher 2005; Galatowitsch and Bohnen 2021). Reed canarygrass specifically, only requires three years to bounce back from a controlling treatment, resulting in a greater need to employ further time, money, and resources towards reducing its levels to a self-managing state (Bohnen and Galatowitsch 2005; Lavergne and Molofsky 2006; Wilcox et al. 2007; Aronson and Galatowitsch 2008; Stinks et al. 2021). Under conditions of increased soil saturation, reed canarygrass has a greater tendency to establish in degraded wetlands (Zedler and Kercher 2005). Given the wetland is not in a position to self-mend through lower input restoration efforts, a higher intensity approach employing multiple rounds of adaptive treatments may be required to break the invasive species feedback cycles (Zedler and Kercher 2005; Galatowitsch and Bohnen 2020).

A key component influencing the duration of time for a restoration project to reach its goals is the presence of invasive species within it the restoration site (Galatowitsch and Van Der Valk 1996; De Steven et al. 2010). The wetland has an average introduced species relative cover of 35.5%, with reed canarygrass making up 16.9% relative cover. This is a marked improvement from the initial conditions of the study site, where the vast majority of the site was monocultures of reed canarygrass and cattail based on visual assessments prior to treatment applications. Across treatments, however, there are no significant differences in the relative cover of both introduced species or reed canarygrass. Although not significant, trends show the seed, mulch + soil plugs treatments as having the lowest values for both introduced species relative cover and reed canarygrass relative cover when compared to the other two treatments. Further, albeit still not significant, the seed, mulch + soil plugs treatment, as expected, has the highest totals for native relative cover. Previous years of data collection have not seen any significant differences across treatments or years (Durant 2020). However, the seed, mulch + soil plugs treatment may be shifting towards improved native species relative cover compared to other treatments. Therefore providing a possible outlet to warrant the increased expenses and labor.

The wetland site is likely highly sensitive to invasive plants due to its land use history. It receives higher inputs of nutrients from surrounding agricultural operations, has experienced impacts to soil health from its previously farmed state, and has large fluctuations in water levels, all leading to a less hospitable environment for native species (Galatowitsch et al. 1999; Zedler and Kercher 2004; Gleason et al. 2011; Wilcox et al. 2018; Bansal et al. 2019; Schultz et al. 2020). The study site resides directly adjacent to a highway, which could be a possible explanation for the continued presence of invasive species; as highways commonly aid in the spread of invasive species (Joly et al. 2011; Lemke et al. 2018). Roadside ditches commonly accumulate water and salt, which aids in the transport of invasive hydrophytic vegetation able occupy niches native salt intolerance vegetation cannot survive in (Galatowitsch et al. 1999; Zelder and Kercher 2004). Zelder and Kercher (2004) highlight the correlation between road density and invasive species spread. Invasive species likely have a greater dominance within newly restored sites because they do not require as many specific site properties to establish themselves (Zedler and Kercher 2005). Thus, seedings without additional treatments may not reach intended outcomes due to more tolerant invasive species dominating less hospitable sites (Aronson and Galatowitsh 2008; Sinks et al. 2021). The wetland has been isolated and previously cropped leading it to possibly be experiencing impacts from an overabundance of nutrients, water, accumulating soil, and invasive species plant material traversing down the landscape into the wetland. Zedler and Kercher (2005) state a change in elevations of only ten cm can have the ability to hinder or promote certain plant species ability to be competitive in systems altered by anthropogenic changes.

The geographic location of the wetland study site may also play a key role in its ability to recover (Zedler and Kercher 2004). Wetlands are located within the landscape in a depressional position making them act as landscape sinks, often collecting contaminated materials and invasive plant propagules. Additionally, the isolated nature of the wetland study site limits the potential for native propagules to be disseminated from other wetlands, thus diminishing the speed of native plant establishment (Galatowitsch and van der Valk 1996; De Steven et al. 2010). Adding to the difficulty of this specific restoration project is the common climatic fluctuations in the Northern Great Plains, producing intense droughts and periods of high precipitation (Woodhouse and Overpeck 1998; Mitsch and Jørgensen 2004; Johnson et al. 2005). Instances of intense weather events creating both drying and wetting circumstances can support more variation in seedbanks as a result of exposure during dry conditions and inundation during wetter conditions (van der Valk and Davis 1978; Johnson et al. 2005; Galatowitsch 2006). Intense variations can then produce drastically different plant communities making management decisions difficult. Understanding and utilizing these influential regional components can help produce a more diverse wetland ecosystem with greater outputs.

Although the restoration treatments included seeding, hay mulch spread, and soil plug additions, it is possible the ability of the site to revegetate was hindered by past land use practices and requires further adaptive treatments to control dominant invasive species within the site. Hilderbrand et al. (2005) state sites may not be equipped to support specific seedings if early colonizing species have not altered the system enough to be more suitable for later successional species. The vegetation makeup of an ecosystem is the result of processes carried out over decades to centuries, therefore, to evaluate the success of a restoration project, long-term assessments are needed (Jørgensen 1994; Aronson and Galatowitsch 2008; Strehlow 2015). The application of site-specific treatments both initially and adaptively may support an increased speed of plant community succession, however, time is needed to see the impacts. There is great importance in evaluating restoration principles from other projects. This method however has the potential to incorrectly account for differences between similar wetland sites (Holling 1995, Holling and Meffe 1996; Hilderbrand et al. 2005).

Assessments of the study site beyond vegetation surveys could prove to be a valuable addition, especially when comparing locations within the study site displaying either a greater dominance of invasive species or a shift towards improved native biodiversity. Soils play an important role in restoration projects given their impact on water movement and quality, nutrients, vegetation establishment and containment, and microbial health. There is a strong likelihood soil characteristics were altered during initial farming operations within the study site. It would be a worthwhile investment to consider microbial activity, nutrient contents, and the soil seedbank to evaluate functionality levels (Zedler and Kercher 2005; Hilderbrand et al. 2005). This may help direct management efforts, as considerations for the vegetation species alone can result in a system with less resemblance to desired natural conditions (Galatowitsch 2006).

Through evaluating the cost of each treatment, results continue to show over eight years following initial treatments, the most cost-effective treatment is seeding alone (Strehlow et al. 2017b, Durant 2020). This result, although counter to the study hypothesis, is promising because there is a greater acceptance towards employing restoration techniques lower in expense that are still capable of producing the same level results as those with higher expenses (Török et al. 2011; Rayburn and Laca 2013). Despite numerous papers illustrating the value of multiple treatment combinations producing a greater chance of restoration success (Foweler 1988; Suding et al. 2004; Galatowitsch 2006; Török et al. 2012; Hopple and Craft 2013; Matthews et al. 2020; Sinks et al. 2021), that was not the case for this wetland restoration. This is often the result within dynamic systems facing alterations from both abiotic and biotic functions of the landscape, making even the most meticulous of plans unsuccessful in achieving specific goals. Hilderbrand et al. (2005) cautions at defining the success of a restoration project based on the elimination of invasive species, as their presence in many systems may be too engrained to be completely removed.

As adaptively managed treatments continue to be applied to this wetland restoration project, it will be important to evaluate alternative goals and applications capable of supporting the site in improved resiliency and diversity, even if that means invasive species are still present within the site. Future management goals should continue to focus on bringing populations of reed canarygrass to a self-manageable level. Within the wetland restoration project, work will be needed to promote diverse, native vegetation communities capable of improving wetland function and value in a self-sustaining state (Maitland and Morgan 2002; Cardinale et al. 2012; Schlutz and Pett 2014; Manton et al. 2016).

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CHAPTER 4: USING GRAZING MANAGEMENT AS A RESTORATION TOOL IN PRAIRIE STREAMS

Abstract

A major threat facing riparian systems is the potential for entrenchment and incision of stream channels. Using theory based on riparian complex state-and-transition model concepts for prairie streams, an experiment was created to assess the potential for riparian grazing to facilitate stream succession towards a more stable state. Treatments included grazing exclusion (GE), rotational grazing (RG), and high intensity short duration grazing (HI). Cross-sections were collected from 13 reaches, across three streams in Southeastern North Dakota, with baseline data collected in August 2020 and in August 2021 following initial treatments. Geomorphic characteristics across stream reaches were categorized based on the Rosgen Stream Classification System (E, C, B), entrenchment ratio (ER), bank height ratio (BHR), width-to-depth ratio (WDR), meander-width ratio (MWR), valley width (VW), sinuosity, slope, and channel material size (D^{50}). Plant community components (PCC) and vegetation cover within greenline plant communities were assessed using Global Positioning System (GPS) and Line Point Intercept (LPI), respectively. Results from multivariate analysis of variance (MANOVA) analyzing the connection between stream geomorphic characteristics, produced no significant differences. Analysis of variance (ANOVA) produced two significant changes across geomorphic characteristics between years. There was a significant difference in percent change of ER between the HI and RG treatments, with the RG reaches having a lower ER. A similar trend was observed for MWR. This is not surprising since both are metrics of floodplain accessibility. No significant differences were observed in PCC, vegetation, bareground, soil surface, or basal cover. The results indicate a single season of RG or HI grazing within an

adaptively monitored system may have longer term use opportunities towards restoring prairie streams. Grazing treatments and monitoring will continue in the future to evaluate influences treatments many have on stream geomorphic parameters and PCC over a longer period.

Literature Review

Riparian Ecosystem Management

Riparian systems, the land surrounding streams with a dependent connection to the stream's hydrology, provide a plethora of ecosystem services, with effects present beyond the riparian system itself (Gregory et al. 1991; Moerke and Lamberti 2004; Holland et al. 2005; Swanson et al. 2015; Meehan and O'Brien 2019; Meehan et al. 2021). Ranchers and land managers benefit from riparian systems through improved forage, habitat, water quality and control, and biodiversity (Moerke and Lamberti 2004; Holland et al. 2005; Kidd and Yeakley 2005; Swanson et al. 2015; Meehan and O'Brien 2019; Meehan et al. 2020). However, with improper management, land managers may also be negatively affected by a reduction in the ecosystem services they rely on (Oles et al. 2017; Meehan et al. 2021).

Human actions such as recreation and agricultural practices have harmed many riparian systems across the United States, and within the United States, riparian grazing has long been a controversial topic. The mismanagement of riparian grazing has led to the degradation of over 80% of stream systems in the western United States (Clary and Kinney 2002). Subsequently, vast amounts of research account for the negative impacts riparian grazing has on stream health and the ecosystem services provided (Kauffman and Krueger 1984; Flenniken et al. 2001; Clary and Kinney 2002). Despite the large amount of research pertaining to the detrimental implications surrounding grazing in riparian systems (Belsky et al. 1999; Kauffman et al. 2004; Kidd and Yeakley 2005; Blanks et al. 2006), there are limited studies on the potential management strategies in place to reduce the possible negative impacts (Lucas et al. 2004; Agouridis et al. 2007). Even fewer research studies show the potential to use riparian grazing to restore Midwest prairie streams while accounting for geomorphic principals (Clary and Kinney 2002). Grazing durations and intensities have the potential to be altered based on stream health needs (Ehrhart and Hansen 1997; Swanson et al. 2015). This approach, although more hands on, promotes an improved relationship between livestock and riparian systems while also providing support for ecosystem services.

Riparian Degradation

A major threat facing riparian systems is the potential for entrenchment and incision of stream channels. Incised streams are streams with a restricted floodplain due to entrenchment, resulting in the abandonment of the original floodplain (Rosgen 1997; Simon and Rinaldi 2006). Incised stream channels are distinguished based on their increased ability to contain flow events in comparison to other channels in their system that are not incised (Simon and Rinaldi 2006). High flow events degrade stream channels as water energy is unable to be absorbed as readily by the stream's floodplain. Incised streams will either be characterized by tall streambanks or a confined valley (Rosgen 1997). Simon and Rinaldi (2006) state when sediment input and transfer are thrown out of equilibrium a channel becomes incised. An incised stream may experience negative effects regarding faster soil erosion along banks, which in turn limits available habitat for presiding species and may degrade water quality both at the site and beyond. The health of the riparian system may decrease along with land productivity. Stream degradation results in increased streambank soil loss, and commonly causes the stream to become unstable (Rosgen 2006). Degradation often can result from dam or reservoir outflows being redirected, alterations to water flow patterns, channelization of streams impacting gradient, head cutting from downstream adjustments, or stream confinement structures (Shafroth 2002; Rosgen 2006). These impacts can be noted through reduced width to depth ratios and raises in bank height ratios, with bank height ratios also acting as an initial alarm for incision that may lead to entrenchment and the abandonment of floodplain access (Rosgen 2006; Meehan and O'Brien 2019). Alterations to stream attributes creating a departure from a balanced equilibrium can increase rates of sediment movement, depletion of land, reductions of belowground water levels, diminished site yields, declines in available lotic habitat, and a lessening of ecosystem services.

Riparian Vegetation

Vegetation along streams provide valuable functions including streambank stability, sediment trapping through roots, improved water infiltration, stabilization of water temperatures, flooding control, and reductions in soil loss (Swanson et al. 2015; Derner et al. 2018; Hecker et al. 2019). Native greenline plants are also valuable resources for livestock when used with adequate consideration for stocking rate, stocking density, and season of use (Swanson et al. 2015). The greenline is defined as the initial area adjacent to a stream, within bankfull elevation, containing perennial plants and directly connected to stream flow and the local water table (Coles-Ritchie et al. 2007; Meehan et al. 2016). Grazing systems have the capacity to help restore certain species within the greenline and promote species with greater soil stabilizing capacities (Rosgen 2006: Derner et al. 2018). Contrary to upland areas, flora within riparian areas maintain nutrient values longer and contribute higher quality and greater quantities of forage for livestock (Unterschultz et al. 2004; Holland et al. 2005; Stringham and Repp 2010), making riparian grazing an important asset to ranchers (Unterschultz et al. 2004).

Riparian areas are vulnerable to change, making them more sensitive to grazing practices. An influential cycle exists between the stream water quality and the presiding vegetation, where both rely on the other to maintain their health (Jeffery et al. 2014). Mismanaging riparian grazing can lead to decreased plant material and cover, increased erosion, reductions in water quality, and reductions in stream biota (Belsky et al. 1999; Kauffman et al. 2004; Kidd and Yeakley 2005; Blank et al. 2006). Therefore, a fine balance is needed to maintain the health of riparian ecosystems, while also allowing livestock to utilize the riparian area's higher quality forage (Oles et al. 2017). It is important to consider alterations to riparian systems can also be carried over into the upland and vice versa. Riparian areas should be managed outside of upland areas based on the major differences in their ecosystem needs (Stringham and Repp 2010; Jeffrey et al. 2014; Swanson et al. 2015). The primary difference between riparian systems compared to upland systems originates from the riparian systems hydrological connection (Gregory et al. 1991; Meehan and O'Brien 2019; Meehan et al. 2021). Management strategies aimed to protect upland grazing usage ultimately still have the potential to harm the presiding riparian habitat and in turn can limit forage quality and productiveness of stream systems (Oles et al. 2017).

Riparian Grazing

While studies have been done on the economic impact grazing or grazing exclusion has on riparian areas, limited studies have been carried out on the potential economic feasibility of riparian grazing for restoration. Even fewer studies have been carried out in Northern Great Plains. Agouridis et al. (2005) values cow/calf pairs for marketable purchases at $40.5 billion, with livestock production as a whole providing $200 billion in revenue from sales based on a 1997 report. The total shared cost between government and land managers across the United States towards improved grazing management practices in 2003 was $2.5 billion. The value of grazing is well documented and measured, however, the value of riparian ecosystems is not as closely monitored (Jeffrey et al. 2014).

Conventional grazing is a widely used grazing practice where livestock are able to roam freely within an enclosure for the entire duration of a growing season (Raymond and Vondracek 2010). This grazing style can lead to degraded stream banks, increased sediment loading and compaction, lower water and habitat quality, and can inhibit the many ecosystem services riparian systems provide. Increases in soil compaction contribute to decreased water movement through soil leading to greater amounts of run off, potentially carrying pollutants. Furthermore, stream banks and channels can be altered as a result of livestock eroding soil (Kauffman and Kreuger 1984). Jeffery et al. (2014) states the impact of riparian grazing over a period of six weeks or more on land grazing excluded for four years prior, will experience increased erosion in the stream valley. Stream widening may occur within a system where cattle are grazing on previously ungrazed land, in turn altering the stream geomorphology (Grudzinski and Daniels 2017).

Evidence suggests grazing riparian wetlands at the appropriate stocking rate, which varies from system to system, can lead to raises in species richness in comparison to sites without grazing (Green and Kauffman 1995; Bullock and Pakeman 1996; Jutila 1999; Humphrey and Patterson 2000; Hoover et al. 2001; Krzic et al. 2004; Austin et al. 2007; Kidd and Yeakley 2005). Grazing plans accounting for the dynamic and complex characteristics of riparian systems can support a raise in species richness due to the removal of dominant forage species by livestock, allowing for other less competitive species to grow in their place (Green and Kauffman 1995; Clarke et al. 2005; Kidd and Yeakley 2005). It is worth noting, grazing systems produce a variation of impacts across riparian systems due to differences in climate, fluvial geomorphology, species composition, and grazing characteristics (Kauffman and Krueger 1984; Belksey et al. 1999; Kidd and Yeakley 2005). Stocking rates and time within riparian systems are main influential components effecting the success of riparian grazing (Swanson et al. 2015; Oles et al. 2017).

Riparian systems may experience improvements relating to channel health and vegetation diversity through the use of minimal to moderate early season grazing practices (Clary 1999). Rotational grazing practices have the potential, given adequate soil types, to alter stream channel types and reduce the amount of soil erosion and runoff (Magner et al. 2008). Through channel evolutionary processes, stream channels can widen and floodplains may be altered. Once the change in land surpasses the potential of the stream, a change in stream type occurs (Rosgen 1997). With proper management, livestock are able to support and provide positive improvements to rangelands (Oles et al. 2017). Rotational grazing may increase the structural integrity of stream valleys and provide improved riparian health (Raymond and Vondracek 2010). Spring rotational grazing has been noted to provide easier management of livestock due to the livestock's lower desire to preside in the riparian system due to higher moisture and greater forage coverage throughout both the upland and greenline (Clary and Kinney 2002). Rosgen (2007) highlights the potential for appropriate grazing to enhance stream edges and greenline plants in comparison to sites devoid of grazing which experienced raises in bank soil loss, however, this is highly dependent on stream type. Magner et al. (2008) highlights the need to analyze current and changing characteristics of riparian geomorphology as a result of riparian grazing to account for potential soil losses and alterations to streamline habitats. To accommodate for the environmental pressure of grazing, actions must be taken to protect the varied functions riparian systems provide through riparian grazing standards (Oles et al. 2017).

Given the varied approaches to riparian grazing, adaptive management needs to be employed within grazing riparian systems (Ehrhart and Hansen 1997). This is due to the site-specific nature of riparian grazing. Ehrhart and Hanson (1997) addressed the effects of riparian grazing on over 70 stream reaches and employed a variety of grazing practices from less than eight days to season long. Their overarching conclusions were that there are no all-encompassing management strategies they recommend. Riparian grazing management must be addressed in a stream-to-stream manner to ensure all the specific characteristics of the stream such as vegetation, soil, water movement, and geomorphology are accounted for (Wyman et al. 2006; Swanson et al. 2015; Oles et al. 2017).

Stream Geomorphology

Geomorphology is a contributing factor to a variety of stream characteristics, from abiotic and biotic features to species abundance (Frothingham et al. 2001). The geomorphology of a stream is the interaction between the traveling water and the topography surrounding and within the stream. The interaction of water and geology paired with impacts from wind, sediment movement, and climate help form a stream's geomorphology (Ruhe 1954; Schaetzl and Anderson 2005). This interaction plays a key role in many of the defining characteristics of a stream. There is a connection between the complexity of a stream's geomorphology, and the abundance of species in a stream, as well as the total volume of organisms (Frothingham et al. 2001; Stringham and Repp 2010). Changes in geomorphology can lead to shifts in the structure of a stream, in turn, altering the stream's biological traits. During times of increased water flow, the structure of a stream changes and has positive or negative effects on the species present (Feminella and Gangloff 2006). Further, the landscape in which a stream is located plays a key role in the way water moves across the land's surface. Water movement creates a specific structure for species to inhabit through its unique characteristics (Porter et al. 2003).

Species have fundamental niches they can reside in and require a habitat that meets those needs to be able to sustain themselves (Porter et al. 2003). As a result of this, the hydrologic conditions set into place through a stream's geomorphology influence the abundance of species present and the biotic integrity (Porter et al. 2003; Meehan et al. 2021). Different locations have different fluvial characteristics stemming from the physical location they are at. The geographic features can impact the speed at which water travels downstream, as well as the variations in stream water depth (Porter et al. 2003). By understanding these interactions and influencing factors, changes in species abundance and habitats can be better understood when analyzing livestock grazing impacts (Porter et al. 2003).

Through the use of ecological site descriptions (ESD) and state-and-transition models (STM), land managers and researchers are able to better inform their decisions on rangeland stream management practices to help maintain the health and productivity of their land, however, more research is needed to understand the impacts cattle have on riparian sites and phases (Ratcliff et al. 2018). An ESD defines site characteristics of soil, plant species presence, and land features, with applications present in riparian systems given a further analysis of hydrological influences and stream geomorphology (Stringham and Repp 2010; Ratcliff et al. 2018; Meehan and O'Brien 2019). Connecting these factors is pivotal for developing riparian vegetation descriptions in accords with a STM, as it helps account for the more dynamic states streams have compared to upland sites (Ratcliff et al. 2018; Meehan and O'Brien 2019). The STMs are used to explain hydrogeomorphy within a given system and help to categorized channel states for specific areas with accounts for their response to disturbances (Bestelmeyer 2009; Ratcliff et al. 2018; Meehan and O'Brien 2019). Pairing ESDs with STMs provide context on areas of a landscape displaying similar attributes and spectrums of geomorphic states to aid in land management decision making processes.

Within North Dakota, riparian complex ecological site descriptions (RCESD) are being developed by utilizing framework from the Rosgen Stream Classification System (RCS) (Meehan and O'Brien 2019). Stream geomorphology and hydrology are defining characteristics in the classification of natural streams within RCS (Meehan et al. 2021). RCESDs utilize classifications of valley type and stream type from the RCS (Meehan and O'Brien 2019). Valley type is determined by evaluating measurements of valley positioning, aspect, slopes, confinement, fluvial features, and material and can give an idea of what type of stream channels may be present (Rosgen and Silvey 1996). Valley type influences stream channel movement, grade, and longitudinal profile (Dunne and Leopold 1978; Rosgen 1994). The most often observed valley types in North Dakota are alluvial and lacustrine valleys (Meehan and O'Brien 2019). Alluvial valleys are characterized as wide with slight relief and common glacial and alluvial terraces that provide for streamline plants (Rosgen and Silvey 1996). Alluvial valleys also contain streams separated from their floodplains and dominated by upland vegetation. Lacustrine valleys are characterized by their formation from old lake beds creating lacustrine sediment, very wide floodplains, and common flooding.

The RCS assesses stream channel entrenchment, slope, sinuosity, width, and depth to denote stream type (Rosgen 1994; Meehan et al. 2016; Meehan et al. 2021) (Figure 4.1). The most influential determinants in stream analyses when considering the stability of a channel are entrenchment ratio (ER), bank height ratio (BHR), and meander width ratio (Meehan and O'Brien 2019). Prevalent streams within prairie ecosystems include E, C, B, F, G stream types, all of which can characterize the stream as having varying levels of structural integrity within their landscape making them more or less stable.

E stream channels have the highest level of stability with a strong connection to their floodplain and prevalent riparian vegetation along their banks resulting from a high-water table (Rosgen and Silvey 1996; Meehan et al. 2016). C streams have a minimal gradient, expansive valleys, slight entrenchment, and are considered stable as long as streambanks are prominently vegetated. Sediment supply from C streams along their streambanks following high flow events create terrace features with material deposited and aggregated on opposite sides of stream bends. B streams have moderate entrenchment, slope, and width to depth ratios with a narrow and parabolic shape. B streams are considered stable to stabilizing, often due to previous disturbances that require further floodplain formations. F streams are classified as unstable resulting from their deep entrenchment, high width to depth ratios, high rates of erosion, and lack of access to their floodplains. G streams, also called gullies, have no access to their floodplain, experience extreme downcutting, and commonly form step/pools. In North Dakota, G streams experience exacerbated impacts from erosion based on the high erodibility of their channel material causing a faster shift towards becoming a F stream.

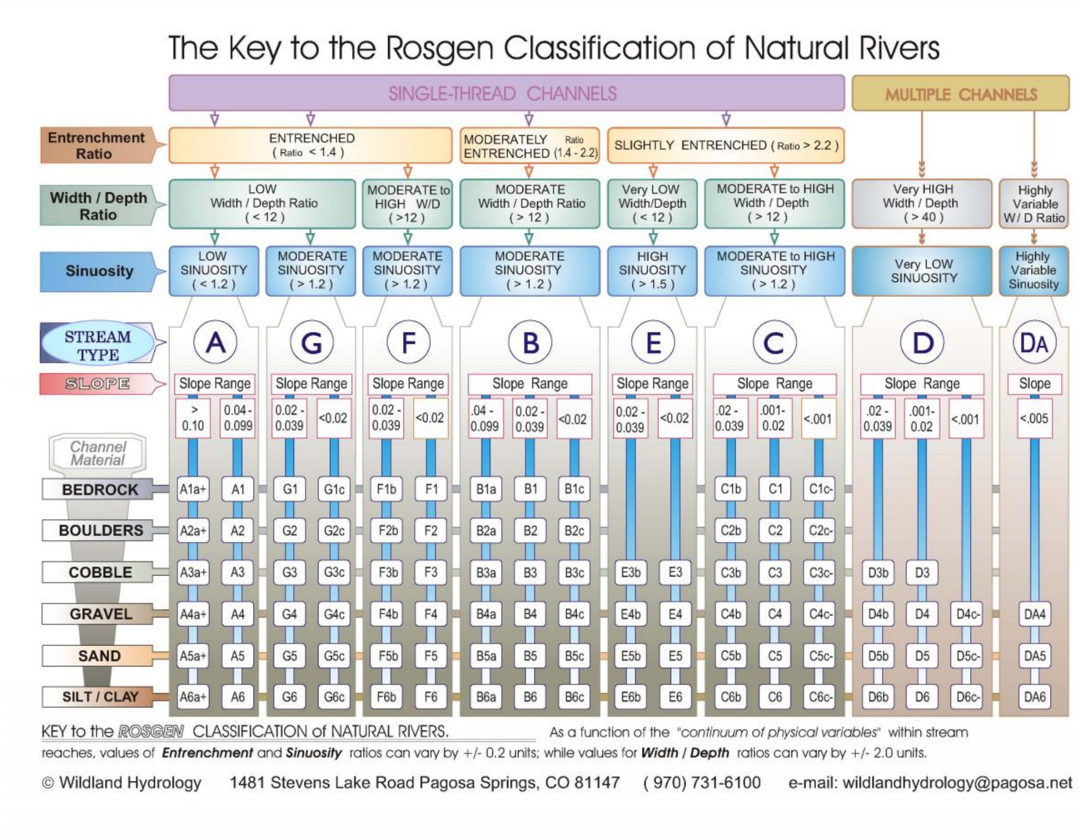

Figure 4.1: Rosgen stream classification of natural rivers dictating entrenchment ratio, width to depth ration, and sinuosity as categories for classifying stream type (Figure from Rosgen and Silvey 2006). Stream type is further classified by slope and channel material.

Within stream classifications, indications of BHR can be valuable characteristics to define a streams' ability to flow over into their floodplain (Rosgen 2006; Meehan et al. 2021). This stream attribute is determined by dividing the low bank height by the bankfull discharge height (Rosgen 2006). The relationship between the stream channel and its residing floodplain is connected to the streams ability to dissipate hydrological stress (Rosgen 2006). This connection plays a central role in the stability of streambanks, as the greater connection the stream has to their floodplain, the greater ability it has to absorb excess water impact to the streambank (Rosgen 2006; Swanson 2015). The greater the bank height ratio, the less connection is present between the stream channel and its floodplain, thus diminishing stream's stability in its current state (Rosgen 2006; Meehan et al. 2021). Plant species along streambanks may also be negatively impacted by a high BHR given the additional stress. The reduction in vegetation creates a compounding impact to the streambanks security as the vegetation coverage declines, water stress is no long absorbed as greatly, and the streambank is impacted even more (Ward et al. 2003; Hooke 2007; Engelhardt et al. 2012; Swanson et al. 2015; Derner et al. 2018; Meehan et al. 2021).

By analyzing channel types (RCS) and making connections to channel evolution models (Simon and Rinaldi 2006), four states have been created to illustrate transitions between stable and unstable stream types to support a RCESD (Figure 4.2) (Meehan and O'Brien 2019). The model illustrates the impact incision has on triggering a transition from a stable state to an unstable state. Incision can lead to stream bank failure subsequently increasing stream channel width and causing sediment to build up along the collapsed banks further widening the channel (Aaland 2010). As the speed of erosion raises, the channel deposits sediment and recreates its floodplain to once again reach a stable state. Stable reaches are attributed to having greater riparian areas and are associated with a stronger connection to their residing floodplains and limited confinement.

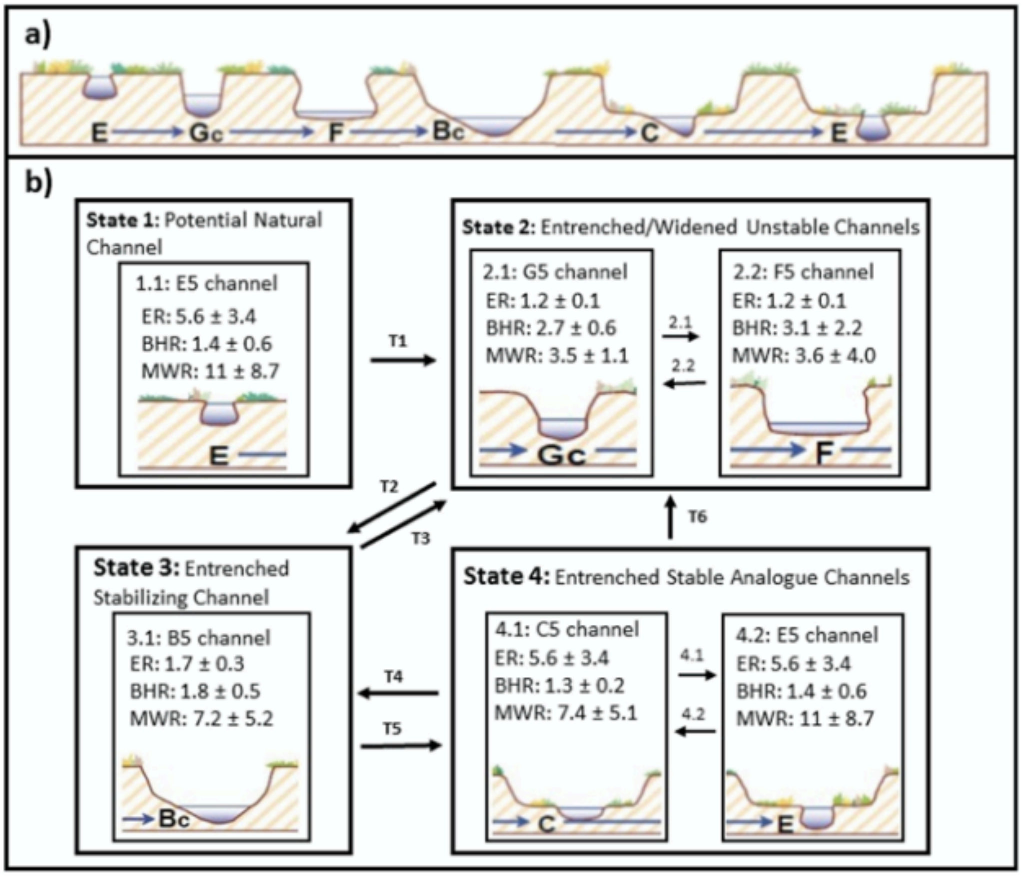

Figure 4.2: a. Illustration of example channel evolutionary phases between stable and unstable states b. State-and-transition model (STM) based on components from Riparian Complex Ecological Site Descriptions (RCESD) and Rosgen Stream Classification System (RSC) (Figure from Meehan and O'Brien 2019).

Additionally, RCESD require a variety of plant community components (PCC) in order to delineate riparian systems (Meehan et al. 2016). The PCC is a valuable tool for evaluating flood patterns, water table height, and predominant stream soils. There are a minimum of two and maximum of five PCC, all of which are dependent on stream type (Figure 4.3). Species present in a PCC are highly dependent on geomorphic characteristics (Atkinson et al. 2018), most prominently ER given its relevance to a stream's floodplain area (Meehan and O'Brien 2020; Meehan et al. 2021). Meehan and O'Brien (2020) determined stream channels were more stable when they were supported by larger PCC2 and PCC3 areas. Geomorphic properties and

PCC are linked, meaning alterations to one factor can subsequently alter the other. However, changes to ecological status are often more related to alterations to geomorphic parameters than PCC.

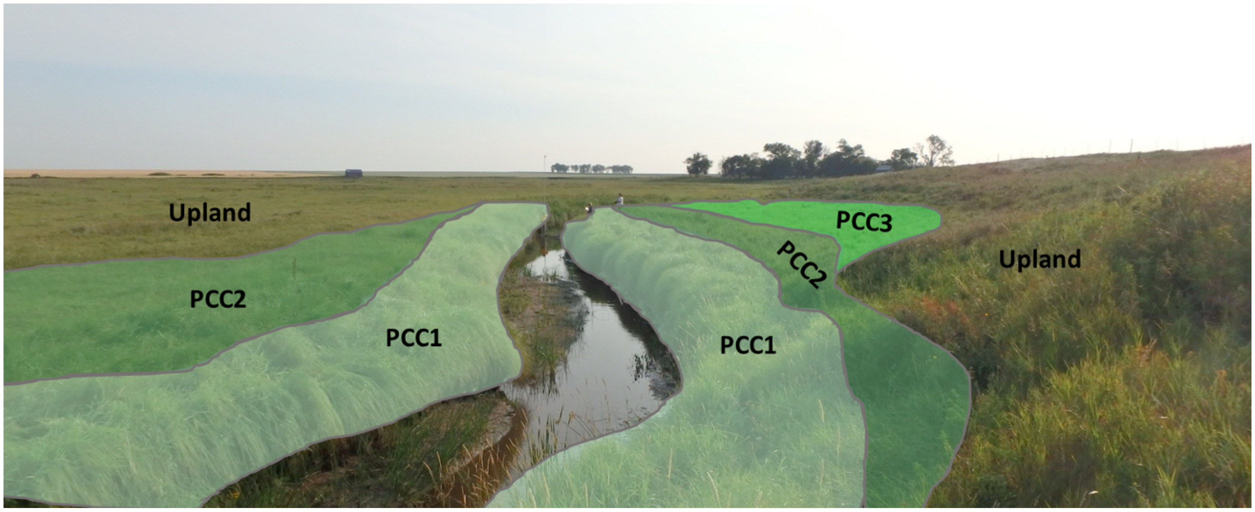

Figure 4.3: Plant community components (PCC) associated with an E stream channel, with the upland system extending beyond the edge of illustrated PCC (Figure from Meehan and O'Brien 2020).

Methods

Numerous restoration projects have been undertaken to help restore riparian ecosystems, however, there are limited studies analyzing the potential to use riparian grazing as a restoration tool, with even fewer studies having been conducted in the Northern Great Plains (Jeffrey et al. 2014). The purpose of this study is to understand and analyze the effectiveness of different riparian grazing treatments towards the goal of prairie stream restoration in the Northern Great Plains. Reaches along three intermittent tributaries of the Sheyenne River located in Ransom and Richland counties of Southeast North Dakota were assessed. Stream reaches are located on Iron Springs Creek, Evanson Creek, and Bird Creek. Stream reaches fall within the Souris-Red-Rainy Region watershed and are within the Sheyenne Valley (North Dakota Hydrological Units 2020).

Treatment responses are monitored using RCS (E, C, B), entrenchment ratio (ER), bank height ratio (BHR), width-to-depth ratio (WDR), meander-width ratio (MWR), valley width (VW), sinuosity, slope, channel material size (D^{50}), plant community components (PCC), and vegetation cover within greenline plant communities. Assessed metrics provide a clear view of the current state each stream is in and how characteristics have changed between years to help understand how best to manage livestock in riparian systems, specifically for channel restoration. RCS and PCC both provide numerous metrics for evaluating dominate driving factors and stream system responses to grazing, making them valuable assessment tools.

Much of the history of the land relating to the Sheyenne River was ultimately influenced by the Pleistocene glaciations that changed the shape of the land (NRCS 2006; Severson and Sieg 2006). The regional accounts from 1862 state the land along the Sheyenne River, near what is known as Kindred, ND today, persisted of excellent pasture lands and fully stocked wooded forests featuring densely populated *tilia americana* (basswood), *populus deltoides* (cottonwood), and *quercus macrocarpa* (oak) (Severson and Sieg 2006). Agricultural practices had a large impact on the land and the degradation of native grassland and stream habitats. Riparian systems in North Dakota are a major location for woodland habitats and provide services such as improved water quality, wildlife habitat, and increased stream bank stability (Holland et al. 2005; Severson and Sieg 2006). Sandy soils are also prominent in this area with highly fluctuating water characteristics (Strehlow et al. 2017). Meehan et al. (2019) and Meehan et al. (2021) indicate streams within prairie ecosystems of the Northern Great Plains have the capacity to bounce back from detrimental impacts and may only be in unstable states for a minimal period of time.

Iron Spring Creek and Evanson Creek are located within the Sheyenne National Grasslands, a unit designated as part of the Dakota Prairie Grasslands (DPG) in 1998 (US Forest Service Dakota Prairie Grasslands). The Sheyenne National Grasslands cover 28,449 ha, with the land publicly owned, and managed across varied private, state, and federal entities with the U.S. Forest Service being the prominent administrator. Management of the DPG is focused on multiple uses, including range, agriculture, conservation, outreach, recreation, ecosystem services, and education.

Iron Springs Creek originally had a direct connection to its floodplain (Figure 4.4). However, following flow alterations and livestock grazing within the riparian system the channel experienced massive downcutting (Figure 4.5), leading to its current state (Figure 4.6). A central focus of the DPG is the proper implementation of grazing practices, with a specific lens regarding riparian and woodland health (Kimbell 2006). These practices consider the type, length, and intensity of grazing systems to support ecosystem services of riparian systems (Kimbell 2006). As stated by the U.S. Forest Service, grazing within the DPG will be done so in a manner that will support the continued proper functioning condition of stream ecosystems while also protecting stream water quality.

Figure 4.4: Conditions of Iron Springs Creek prior to extreme downcutting. Photo received from U.S. Forest Service via Rangeland Specialist, Stacy Swenson. No date or photo credits were made available.

Figure 4.5: Photograph by E. Podoll taken on August 7th, 1969 of Iron Spring Creek within Richland County ND Section 19. Original photo caption reads, “Iron Springs Creek. One of the few flowing streams in southeast North Dakota and the only one in the Sand Hills area. Future plans may call for excluding livestock from this area and development for picnicking and recreation” (USDA-Soil Conservation Service, 1969)

Figure 4.6: Current conditions of Iron Springs Creek as of August 10th, 2020. Photo taken by Dr. Miranda Meehan.

Bird Creek is located within the Albert K. Ekre Grassland Preserve in Richland County North Dakota. Bird Creek is roughly 6 miles north of the DPG, where Iron Spring Creek and Evanson Creek are located. The Albert K. Ekre Grassland Preserve was donated to the North Dakota State University Development Foundation in the 1990s for use towards furthering research and education (Norland 2012). The land is used for varied types of crops, grazing, and ecological studies.

All study stream reaches are located within MLRA 56, Red River Valley of the North (NRCS 2006). The region is located in the tallgrass prairie ecosystem and contains species between the border of bluestem prairie and wheatgrass-bluestem-needlegrass prairie. The region provides for natural prairie vegetation including *Andripogon gerardii* (big bluestem), *Panicum virgatum* L. (switchgrass), *Sorghastrum nutans* (Indiangrass), and *Schizachyrium scoparium* (little bluestem). Soil orders are dominated by Mollisols and Vertisols with most of the region used for private cropland that accounts for 79% of land use. As the elevations change, so do the predominant species and soil types (Stroh 2002). Prevalent soil types include Fluvaquentic Haploborolls and Udifluventic Haploborolls at elevations of 300 meters (plus or minus five meters) and Mollic Udifluvents at elevations of 295 meters (plus or minus five meters).

The 30-year (1991-2020) climate averages of this region, based on the McLeod 3E climate station, are characterized by an average annual temperature of 5.67°C, with the average maximum temperate being 11.83°C, and the average minimum temperature being -0.44°C (NOAA 2021). Average annual rainfall is 60.93 cm, with peak precipitation occurring during the summer, with June attributing 11.50 cm of precipitation.

Grazing treatments are separated across three different treatments; rotational grazing (RG), high intensity short duration grazing (HI), and grazing exclusion (GE). There are a minimum of three replications of each treatment spread across the three study sites. Iron Spring Creek has 3 treatments (Figure 4.7); GE with two cross sections, RG with three cross sections, and two treatments of HI with two cross sections each. Evanson Creek has one RG treatment with one cross section (Figure 4.8). Bird Creek has one GE, one RG, and one HI treatment each with one cross section (Figure 4.9) Streams with more than one treatment were separated by fencing.

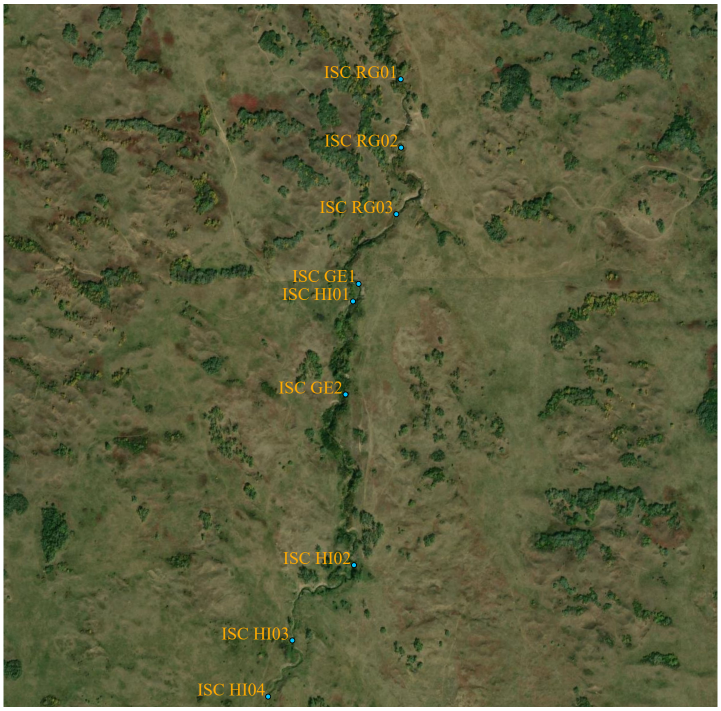

Figure 4.7: Iron Spring Creek riparian grazing treatments of grazing exclusion (GE), rotational grazing (RG), and high intensity short duration grazing (HI) cross section locations within Dakota Prairie Grasslands.

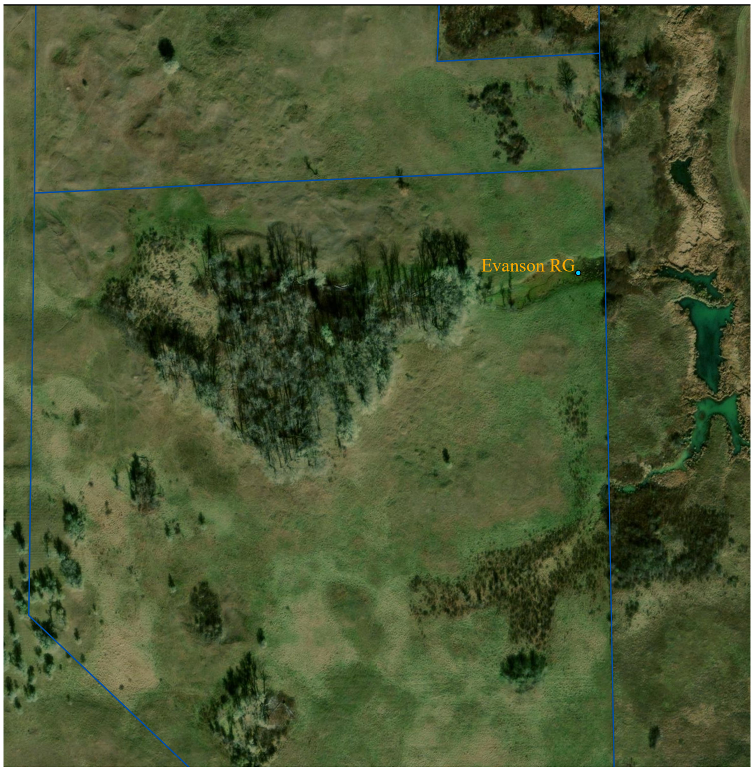

Figure 4.8: Evanson allotment rotationally grazed (RG) treatment and cross section location within Dakota Prairie Grasslands.

Figure 4.9: Bird Creek riparian grazing treatments of grazing exclusion (GE), rotational grazing (RG), and high intensity short duration grazing (HI) cross section locations within the Ekre Grassland Preserve. Fences separating treatments are outlined in white.

Reconnaissance of stream channels were carried out in May 2020 to determine the pretreatment state of each reach and assign corresponding treatments with the help of the U.S. Forest Service. Stream reconnaissance was conducted through a complete foot survey along the length of each stream assessed (Hecker et al. 2019). Surveys accounted for stream geomorphology, land use, plant species, and disturbances. Observations were input into Trimble Geo 7X GPS as points and reflected stream channels similar in type and characteristics across the watershed observed. Treatments were also assigned based on rancher acceptance of practices, as well as stream accessibility by livestock to ensure treatments were carried out successfully.

Ranchers were responsible for turning over livestock in adherence with assigned riparian grazing treatments.

Baseline data was collected in August 2020 and included monitoring of the study stream reaches using RCS (Rosgen 1994; Meehan and O'Brien 2019), plant community composition and cover within the riparian plant communities (Hecker et al. 2019 and Meehan and O'Brien 2019), plant community mapping (Meehan and O'Brien 2020), and photo points along all study reaches. Assessments were conducted to collect baseline data on stream channel characteristics and plant communities to identify and compare the effect of HI, RG, and GE on vegetation cover and stream morphology within Northern Great Plains intermittent prairie streams.

Cross-sections were taken at each treatment, covering the complete stream valley spanning the range of the stream and the corresponding water table (Rosgen 1985; Rosgen 1994). A laser level and survey rod were used to assess the heights of the stream channel and valley with specific notes on flood-prone areas, bankfull height, stream edges, and thalweg. Decisions of where to begin the cross-section analysis were based on riffles. Assessed stream characteristics included streamline (left and right), greenline (left and right), stream edge (left and right), and flood prone (left and right) were collected using Trimble Geo 7X GPS (Trimble, Inc., Sunnyvale, CA) supplemented with ESRI ArcPad 10.2 (Esri, Redlands, CA) and Trimble Positions extension software (Trimble, Inc., Sunnyvale, CA). Geomorphic characteristics across stream reaches were categorized based on the RCS, entrenchment ration (ER), bank height ratio (BHR), width to depth ratio (WDR), meander width ratio (MWR), valley width (VW), sinuosity, slope, and channel material size (D^{50}). BHR was calculated for each treatment to analyze stream channel stability and floodplain linkages. To determine BHR low bank height was divided by bankfull discharge height (Rosgen 2006).

Plant community composition and cover within the riparian plant communities were assessed using the line-point intercept (LPI) method (Herrick et al. 2005; Coulloudon et al. 1999). Measurements assessed present top canopy species, lower canopy layers, and soil surface. One hundred measurements on a single transect parallel to the stream channel bankfull within the greenline community were taken by dropping a pin flag at evenly spaced intervals. All plants intercepted or contacting the pin flag were documented. Soil surface was marked as either plant species, rock, litter, organic litter, moss, lichen, or soil to calculate basal cover (Herrick et al. 2005). On stream reaches where 100 intervals did not fit on a single side, both left and right greenline communities were measured to total 100 intervals. Observed vegetation was categorized by wetland indicator status into 1) obligate (OBL), 2) facultative wetland (FACW), 3) facultative (FAC), 4) facultative upland (FACU), and 5) upland (UPL) (Reed 1988; Lichvar et al. 2012) to evaluate changes in vegetation composition based on grazing practices.

Mapping of riparian PCC took place using a Tremble Geo GPS (Trimble, Inc., Sunnyvale, CA) supported with ESRI ArcPad (Esri, Redlands, CA) and a Trimble Positions extension software (Trimble, Inc., Sunnyvale, CA). Data was collected by creating polygons encompassing each riparian PCC by walking the desired section (Meehan and O'Brien 2020). Each PCC was assessed using geomorphic indicators and further analyzed in ArcMap to ensure consistent polygons were created including only riparian PCC. Riparian PCC were clipped using the clip function in ArcMap to the shortest reach analyzed for each replication. Polygons were clipped perpendicular to the thalweg, making polygon edges spread across a 90-degree angle covering all PCC measured (Meehan and O'Brien 2020). Normalizing polygons to create a consistent length aided in the analysis of PCC areas. PCC area was calculated using the calculate “geometry function” in ArcMap. Mapping took place to monitor changes in riparian PCC and assess potential area shifts.

The HI treatment stocking density ranged from 6.23 cows/ha to 13.17 cows/ha. The stocking rate ranged from 50.11 Animal Unit Days (AUD)/ha to 162.10 AUD/ha. Animal Unit Equivalents (AUE) ranged from 1.15 AUE to 1.23 AUE. Grazing of the Bird Creek HI treatment took place from June 23rd, 2021, to July 2nd, 2021 totaling ten days and grazing of the Iron Spring Creek HI treatment began June 26th, 2021, and lasted until July 2nd, 2021 totaling seven days. The RG treatment stocking density ranged from 0.80 cows/ha to 3.29 cows/ha. The stocking rate ranged from 30.29 AUD/ha to 30.31 AUD/ha with AUE ranging from 1.15 AUE to 1.4 AUE. Grazing at the RG treatments took place between July to August across the three different pastures.

Duration of grazing was based on the utilization of key forage species with a target utilization of 50%, which was assessed using the North Dakota Grazing Monitoring Stick (Meehan et al. 2021). Measurements of both grazed and ungrazed pastures averages were taken of the grazed height and divided by the ungrazed height averages. The number was then subtracted from one and divided by 100 to understand the percent of forage removed. Percent forage removed was compared to a height to weight utilization chart to assess if the desired level of utilization had been achieved. Animals were removed at 50% utilization of key species.

To evaluate impacts to stream channel geomorphology between baseline data collection (2020) and one year following treatment applications (2021), one-way analysis of variance (ANOVA) was completed using R 4.1.1 software (RStudio 2021). Pairwise comparisons were employed through a post-hoc Tukey’s honesty significant difference (HSD) test using package “agricolae” (de Mendiburu 2017). Potential treatment impacts were further evaluated using an

ANOVA to assess change and percent changes between years. Each dependent variable from baseline data collected in 2020 was subtracted from the 2021 data and recorded as an absolute value. Percent change was calculated by dividing the change between years by the original baseline value. Multivariate analysis of variance (MANOVA) was used to analyze the relationship between stream geomorphic characteristics, with Principal Components Analysis (PCA) producing a visualization of geomorphic parameter data, both employing RStudio 4.1.1, "vegan" package (Oksanen et al. 2018). To evaluate trends in greenline vegetation characteristics, Non-metric Multidimensional Scaling (NMDS) was used to create an ordination. Dependent variables of analyses included ER, WDR, BHR, MWR, VW, slope, sinuosity, D_{50}, PCC, basal cover, bareground, and soil surface. The independent variables were treatment type (GE, RG, and HI) and year (2020 and 2021).

Results

Stream channels were classified for each cross section assessed. Classification from baseline 2020 data indicated seven cross-sections were stream type E, five cross-sections were stream type B, and one cross-section was stream type C (Table 4.1). Classifications from 2021 illustrated a change in one stream type, with now six cross-sections classified as stream type E, six cross-sections classified as stream type B, and one cross-section classified as stream type C (Table 4.2). The change in stream type classification occurred on a RG reach within Iron Spring Creek.

Channels were classified as E when displaying entrenchment ratios greater than 2.2, with width to depth ratios less than 12, and a sinuosity greater than 1.5. This indicates the stream channel is both narrow and deep with high sinuosity and slight entrenchment. B channels are classified based on entrenchment ratios between 1.4 to 2.2, width to depth ratios greater than 12, and sinuosity reaching greater than 1.2. These characteristics indicate the B channels are moderately entrenched with a moderate width to depth ratio, and a moderate sinuosity, creating a channel with a parabolic shape. C channels are classified as slightly entrenched requiring a value greater than 2.2. They have a moderate to high width to depth ratio of greater than 12 and a moderate to high sinuosity reaching greater than 1.2. C channels are characterized by movement laterally within their channel and are both wider and less deep than E channels.

Channel slopes for E streams ranged from 0.079 to 0.45 in 2020, with data from 2021 indicated E stream slopes ranged from 0.058 to 0.39. B channel slopes ranged from 0.43 to 0.098 and 2.1 to 0.13 in both 2020 and 2021. C channel slopes were 0.83 in 2020 and 0.92 in 2021. Between both 2020 and 2021, channel material within cross-sections indicated the dominant substrate was comprised mainly of sands, classified as material between 0.062 mm to 2 mm in size. Channel material in 2020 ranged from 0.12 mm to 0.19 mm and in 2021 channel material ranged from 0.08 mm to 0.21 mm.

Analysis of variance (ANOVA) identified two significant changes across geomorphic parameters between years. When assessing percent change between years, ER was significantly different when comparing the HI treatment to the RG treatment ($F_{2,10} = 6.726$, $P = 0.0141$, Figure 4.10), with the HI reaches having a significantly higher ER percent change. A similar, trend was observed for MWR, with the HI treatments having a significantly higher percent change than the RG and GE treatments ($F_{2,10} = 10.51$, $P = 0.00348$, Figure 4.11).

Table 4.1: Baseline stream channel cross-section parameters from 2020 across Iron Springs Creek (ISC), Ekre, and Evanson sites using Rosgen's classification of natural streams. Treatments included grazing exclusion (GE), high intensity short duration grazing (HI), and rotational grazing (RG).

Site	Treatment	Rep	Stream Class	Entrenchment Ratio	Width/ Depth Ratio	Bank/ Height Ratio	Meander/ Width Ratio	Sinuosity	Slope	D_{50} (mm)
Ekre	GE	1	E	2.8	6.4	1.03	1.38	1	0.45	0.18
Ekre	HI	1	E	2.3	7.6	1.02	0.52	1	0.18	0.17
Ekre	RG	1	E	4.1	2.2	1.01	0.93	1	0.095	0.16
Evanson	RG	1	C	5.4	34	1	1.04	1.05	0.83	0.17
ISC	GE	1	E	2.4	6.4	1.05	1.76	1.05	0.92	0.19
ISC	GE	2	B	1.6	11.6	1.02	1.58	1.027	0.43	0.17
ISC	HI	1	B	1.8	9.4	1.06	0.60	1	0.42	0.17
ISC	HI	2	E	3.3	5.3	1.05	0.54	1.01	0.079	0.18
ISC	HI	3	B	2	6.3	1.26	0.86	1	0.17	0.18
ISC	HI	4	B	2.1	7.6	1.13	2.65	1.11	0.098	0.12
ISC	RG	1	B	1.9	7.7	1.05	1.95	1.09	0.26	0.13
ISC	RG	2	E	3.2	6.2	1.03	3.39	1.13	0.18	0.16
ISC	RG	3	E	2.9	5	1.05	2.04	1.01	0.11	0.18

Table 4.2: Stream channel cross-section parameters from 2021 across Iron Springs Creek, Ekre, and Evanson sites using Rosgen's classification of natural streams. Treatments included grazing exclusion (GE), high intensity short duration grazing (HI), and rotational grazing (RG).

Site	Treatment	Rep	Stream Class	Entrenchment Ratio	Width/ Depth Ratio	Bank/ Height Ratio	Meander/ Width Ratio	Sinuosity	Slope	D_{50} (mm)
Ekre	GE	1	E	2.3	9.2	1	0.60	1.001	0.39	0.09
Ekre	HI	1	E	2.3	7.7	1	0.93	1	0.058	0.08
Ekre	RG	1	E	3.7	4	1.03	1.12	1.001	0.16	0.08
Evanson	RG	1	C	2.8	56.1	1	0.34	1.02	0.92	0.08
ISC	GE	1	E	2.4	11.1	1	1.15	1.068	0.16	0.21
ISC	GE	2	B	1.5	10.8	1.02	0.90	1.005	0.27	0.17
ISC	HI	1	B	1.9	7.3	1.03	0.95	1.001	0.17	0.18
ISC	HI	2	E	3.2	6.7	1.07	1.30	1.002	0.13	0.19
ISC	HI	3	B	2.1	7.2	0.99	1.61	1.033	0.17	0.18
ISC	HI	4	B	2	6.3	1.05	2.65	1.061	0.13	0.17
ISC	RG	1	B	1.6	9.3	1.03	1.22	1.032	2.1	0.16
ISC	RG	2	E	2.6	7.3	1.06	3.63	1.063	0.31	0.13
ISC	RG	3	B	2.1	9.5	1.08	1.96	1.021	0.19	0.16

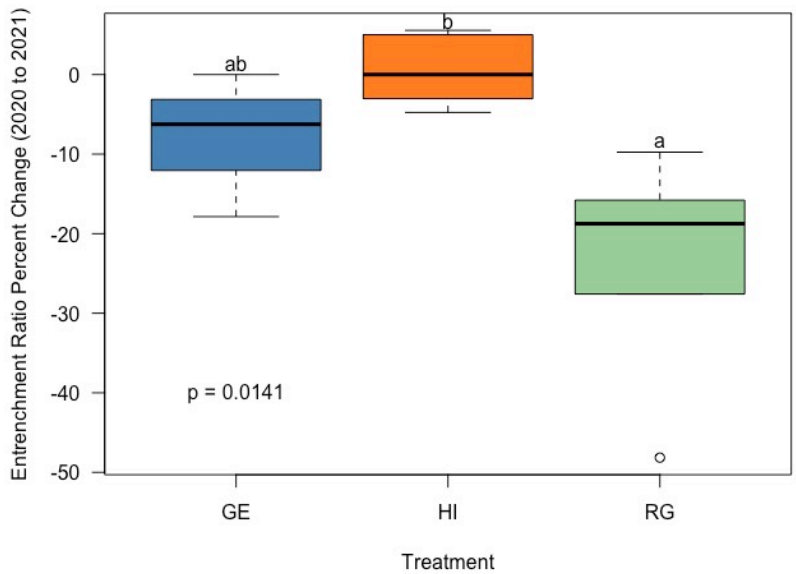

Figure 4.10: Percent change in entrenchment ratio by restoration treatments across 2020 and 2021. Entrenchment ratio was significantly different across high intensity short duration (HI) and rationally grazed (RG) treatments. A statistical difference at α = 0.05 is indicated by different letters.

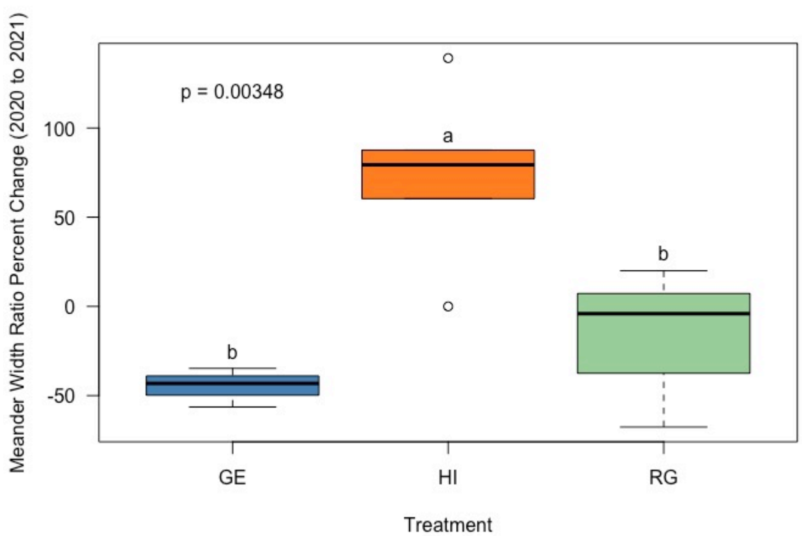

Figure 4.11: Percent change in meander width ration (MWR) by restoration treatments across 2020 and 2021. MWR percent change was significantly different when comparing treatments of grazing exclusion (GE), high intensity short duration (HI), and rotational grazing (RG). A statistical difference at α = 0.05 is indicated by different letters.

The ANOVA indicated no significant differences were present in percent changes for plant community characteristics of bareground ($F_{2,10}$ = 0.543, P = 0.597, Figure 4.12a), soil surface ($F_{2,10} = 0.132$, $P = 0.877$, Figure 4.12b), D^{50} ($F_{2,10} = 0.492$, $P = 0.625$, Figure 4.12c), and basal cover ($F_{2,10} = 0.953$, $P = 0.418$, Figure 4.12d) across treatments.

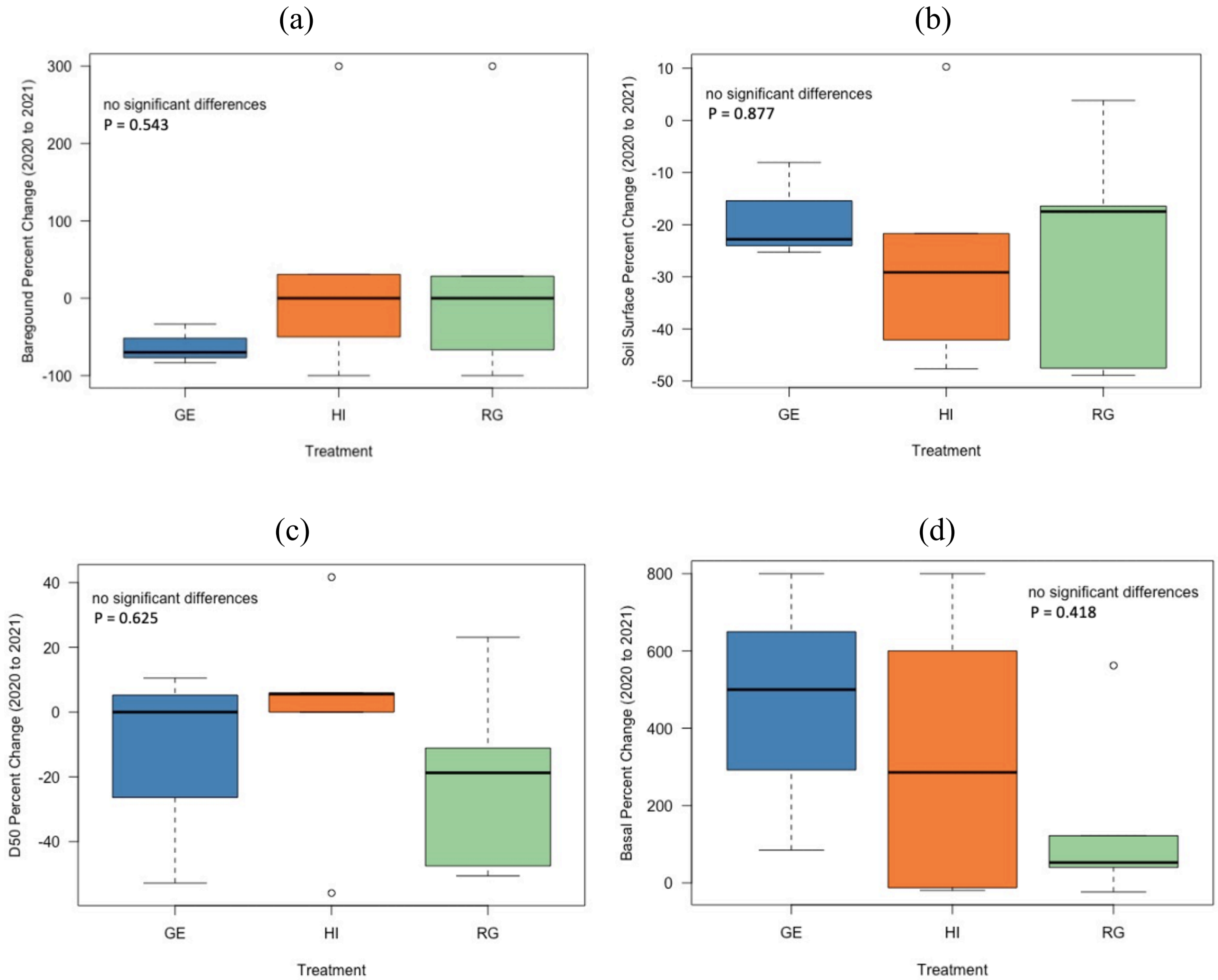

Figure 4.12: Greenline plant community characteristics of bareground (a), soil surface (b), D^{50} (c), and basal cover (d) percent change between 2020 and 2021 were not significantly different across treatments of grazing exclusion (GE), high intensity short duration (HI), and rotational grazing (RG).

Percent change in the area of individual PCCs did not differ between treatments ($F_{2,10} = 0.408$, $P = 0.675$, Figure 4.13). However, although insignificant ($F_{2,23} = 3.72$, $P = 0.0398$, Figure 4.14), there was a noted difference between RG treatments compared to the GE and HI treatments, with RG treatments have a larger total PCC area.

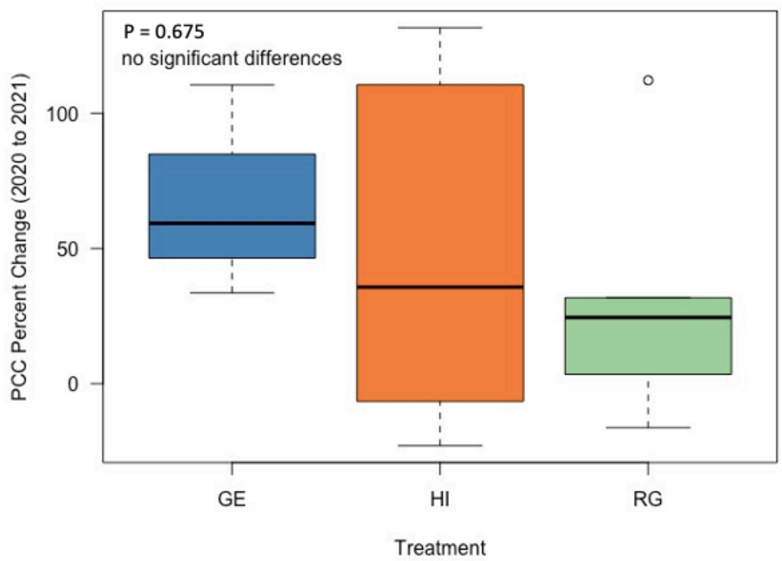

Figure 4.13: Plant community components (PCC) percent change between 2020 and 2021 was not significantly different across treatments of grazing exclusion (GE), high intensity short duration (HI), and rotational grazing (RG).

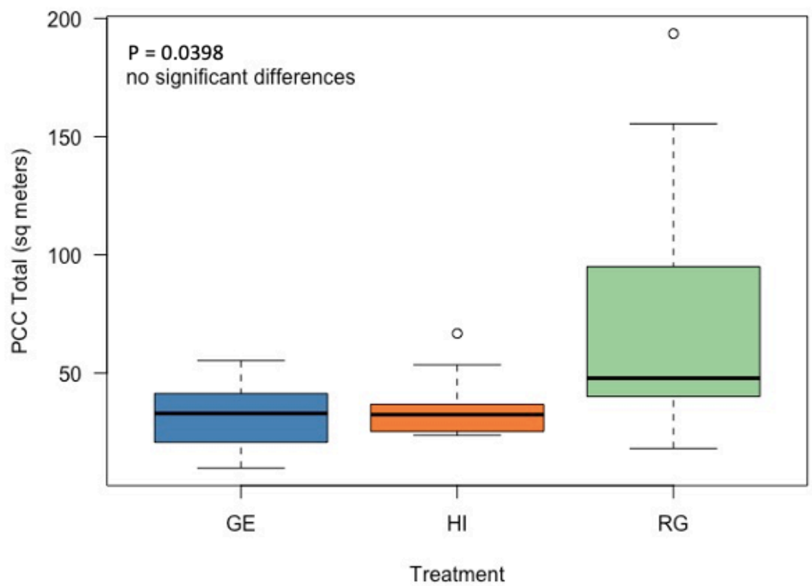

Figure 4.14: Total plant community components (PCC) area (sq meters) was not significantly different across treatments of grazing exclusion (GE), high intensity short duration (HI), and rotational grazing (RG) in 2021.

Evaluations of stream geomorphic characteristic (ER, BHR, WDR, MWR, VW, sinuosity, slope, and D^{50}) connections produced no significant differences across treatments (P = 0.1606), year (P = 0.2396), or treatments x year (P = 0.2824) when using MANOVA (Figure

4.15). The first two principal components explained 51% of the variation with the parameters measured, with PC1 accounting for 29% of the variation and PC2 accounting for 22% of variation.

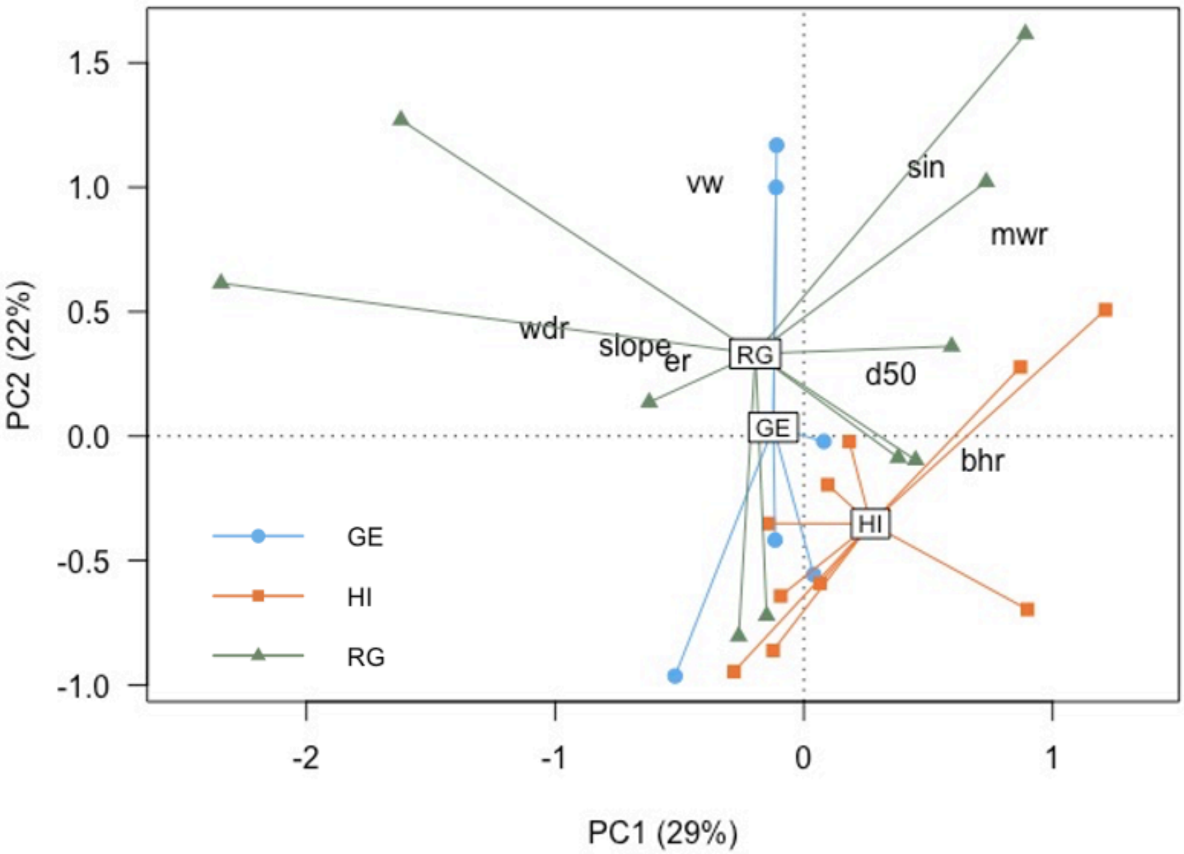

Figure 4.15: Ordination of geomorphic parameters assessed within cross sections between 2020 and 2021. MANOVA indicated no significant drivers between geomorphic characteristics across year and treatments.

Seventy-seven different plant species were identified within the greenline plant community across the 13 stream reaches. Each plant species was placed into its respective wetland indicator group (FAC, FACU, FACW, OBL, UPL, and None). A NMDS was used evaluate trends in greenline vegetation characteristics and found no connections between wetland indicators across treatments.

Discussion

The purpose of this study was to analyze the effectiveness of different riparian grazing treatments towards the goal of stream channel restoration on prairie streams in the Northern Great Plains. Treatments included RG, HI, and GE applied during the summer of 2021. The information attained will aid in land management decisions regarding livestock grazing for the purpose of increasing floodplain connectivity and stream channel resiliency.

Entrenched streams have limited access to their floodplain and a reduced ability to dissipate energy from high flow events (Rosgen 1997; Simon and Rinaldi 2006). As time progresses, an entrenched stream will rebuild its floodplain in a narrower corridor through channel evolutionary processes. To aid in the formation of floodplains for entrenched streams, it was predicted livestock grazing could be utilized to help create and widen floodplain access. By allowing livestock to graze riparian systems livestock can utilize the high valued forage adjacent to streams (Unterschultz et al. 2004; Holland et al. 2005; Stringham and Repp 2010) and potentially level out the confining valley slopes to encourage a quicker transition between stream evolutionary states. Although some stream channels started out in State 4, as E and C channels, their floodplain access was still limited by a confined valley. A primary objective of this study was that following livestock disturbance, stream access to their residing floodplains would increase, even if that meant transitioning back to a less stable state. Prairie streams within North Dakota have alluvial and lacustrine material, making transition times between stages quicker and recovery potential greater (Meehan et al. 2016). Processes of high erosion and channel widening in States 2 and 3 are necessary to support in the recreation of floodplains to improve stream stability. Streams are highly dynamic and capable of responding to disturbances altering their states (Swanson et al. 2015; Ratcliff et al. 2018; Meehan and O’Brien 2019).

Influences on stream ER and MWR follow the predicted channel evolutionary states indicated in the RCESDs STMs for North Dakota prairie streams (Meehan and O'Brien 2019). Within the RG treatments, percent change between 2020 and 2021 illustrated a reduction in ER as well as a slight reduction in MWR when compared to the HI treatments. Alternatively, the HI grazing treatment had an increase in ER and MWR between years. Although no significant differences were present when comparing changes between years, percent change between years was significantly different between the HI and RG treatments, with the HI reaches having a greater percent change in ER. A similar trend was noted with MWR percent change, with the HI reaches having a greater percent change than the RG and GE treatments. This trend makes sense given the connection ER and MWR variables have on influencing floodplain accessibility (Meehan and O'Brien 2020; Meehan et al. 2021). One RG treatment transitioned from being classified as an E channel with an ER of 2.9 to being classified as a B channel with an ER of 2.1. As indicated in the STM, reductions in ER and MWR are connected to changes in stream type and states. B streams are listed as entrenched and stabilizing channels, with C and E streams listed in state 4 as entrenched stable analogue channels. O'Callaghan et al. (2018) indicated changes in stream type are a common occurrence following riparian grazing, however, Williamson et al. (1992) highlighted alterations in channel morphology can often be short lived and pose minimal impacts on the long-term state of a stream channel. Often, larger streams are more resilient to grazing practices, with smaller streams experiencing greater reductions in bankfull width and stability (Williamson et al. 1992; O'Callaghan et al. 2018).

Although no significant differences were found in PCC between years, it was noted the RG treatment displayed the greatest size in riparian PCC. This was surprising given ER is a predictor of PCC size (Meehan and O'Brien 2020) and the ER for the RG decreased between years (although not significantly), making the increase in PCC area counter to expected trends. Between years there was a noted increase in PCC size across all treatments, which is promising to see given PCC, especially PCC1, is highly sensitive to improper grazing practices (Belsky et al. 1999; Kauffman 2004; Meehan et al. 2016; Oles 2017; Kauffman et al. 2022). Further larger PCC2 and PCC3 areas are associated with more stable reaches (Meehan and O'Brien 2020). Trends of increased PCC area within this study, albeit not significant, support the hypothesis that riparian grazing has the potential to increase floodplain size given the increase in PCC area. For this study only PCC1 and PCC2 were mapped given stream incision hindered the support of additional PCCs.

In a similar study Derose et al. (2020) assessed 46 grazed stream cross sections to evaluate influences of grazing regimes on stream health. Derose et al. (2020) found stream health was not significantly impacted by stocking rate but was impacted by the intensity of management (i.e. time on site fencing). Although grazing may alter vegetation species composition, channel stability, stream geomorphology, water movement, and available niches (Belsky et al. 1999; Agouridis et al. 2005; Derose et al. 2020), high intensity management practices can limit the impact grazing has on these stream attributes (Swanson et al. 2015; O'Callaghan et al. 2018; Derose et al. 2020). On streams with limited access to floodplains due to deep entrenchment, overhanging banks may be eroded by livestock, however since these upper banks are not directly connected to the stream, channel erosion is not raised significantly (Williamson et al. 1992). Variation in stream channel characteristics, such as morphology and soil types, can account, in part, for the conflicting research conclusions on impacts livestock may have on stream channels (O'Callaghan et al. 2018).

We observed no significant differences between years when comparing vegetation composition, bareground, soil surface, or basal cover. Danvir et al. (2018) evaluated differences in bareground and vegetation cover within riparian corridors between strategically managed ranches, continuously stocked, and rested pastures, and found sites strategically managed had the greatest coverage of greenline vegetation and lower bareground. Danvir et al. (2018) highlighted the value of adaptively managing riparian systems with higher stocking rates to include lower frequencies of grazing and raising rest times. Rotationally grazing riparian systems can lead to reduced upland bareground (Jacobo et al. 2006; Teague et al. 2010; Danvir et al. 2018) and improved greenline vegetation coverage (Booth et al. 2012; Swanson et al. 2015; Danvir et al. 2018). This supports our findings that riparian grazing did not alter species composition, following one year of treatments. A common concern surrounding riparian grazing is the impact grazing may have on increasing bareground and erosion (Belsky et al. 1999; Kauffman et al. 2004; Unterschultz et al. 2004; Kidd and Yeakley 2005; Blank et al. 2006) and altering riparian vegetation composition (Swanson et al. 2015; Kauffman et al. 2022). Sluis and Tandarich (2004) states plant communities facing the highest level of disturbances will subsequently have the lowest species richness. With successful management, Swanson et al. (2015) states grazing management plans can benefit and improve riparian plant communities.

Disturbances to upland watersheds carry over impacts to stream channels (Zedler and Kercher 2004). Iron Spring Creek has faced flow alterations caused by weir and stock pond washouts, lake discharge and outlet systems (Devils Lake), channel diversions (Sheyenne River), and excess drain flow from tile drainage systems (Stacy Swenson Personal Communication). Increased flow drainage from tile drains and Devils Lake can carry pollutants, sediment, and salts into the Sheyenne River and its tributaries (Iron Spring Creek, Evanson Creek, and Bird

Creek) (Shabani et al. 2017). With pumping from Devils Lake into the Sheyenne River expected to persist (Shabani et al. 2017), continued raises in flow velocities can cause serious stream degradation (Zedler and Kercher 2004; Shabani et al. 2017), especially if streams do not have adequate floodplains and deep-rooted riparian vegetation to help secure banks and absorb increased flow energy (Swanson et al. 2015; Derner et al. 2018; Hecker et al. 2019; Meehan et al. 2021). Even if riparian grazing was removed from Iron Spring Creek, Williamson et al. (1992) attests active channels will continue to be impacted by erosion. Stream channels with high functioning riparian systems have increased recovery potential, resiliency, and can support well managed grazing regimes (Swanson et al. 2015). Riparian vegetation can be highly resilient to short duration grazing treatments and can handle alterations to valley shape and channel form given adequate rest periods.

Duration of grazing treatments were limited during the 2021 grazing season due to severe and extreme drought conditions across the state (National Drought Mitigation Center 2021). Continuing grazing for a longer time could have resulted in greater differences between treatments, however, given forage availability was limited, treatment durations were shortened to account for livestock nutrient demands and the greater sensitivity of the land (Sedivec et al. 2017). Additional years of treatment applications will be needed to facilitate stream succession towards a more stable state and further test the research hypotheses. Disturbances within this system are common and until the root of these points of stress are addressed, it is likely downcutting, incision, and degradation of the assessed stream channels will continue. Restoration efforts increasing stream floodplain connectivity and resilience will help support streams in handling the impact of future high flow events.

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APPENDIX: FULL SPECIES LIST ACROSS ALL PROJECTS

Scientific Name[a]	Common Name	Origin[a]	Life	Physiognomy	Study Site
Abronia fragrans	Snowball sand verbena	Native	P	Forb	Riparian
Acer negundo	Boxelder	Native	P	Tree	Wetland, Riparian
Achillea millefolium	Common yarrow	Native	P	Forb	Camp Grafton
Agropyron cristatum	Crested wheatgrass	Introduced	P	Grass	Camp Grafton
Agrostis gigantea	Redtop	Introduced	P	Grass	Wetland, Ekre Grassland, Riparian
Agrostis hyemalis	Winter bentgrass	Native	P	Grass	Wetland
Agrostis scabra	Rough bentgrass	Native	P	Grass	Wetland
Alopecurus aequalis	Shortawn Foxtail	Native	P	Grass	Wetland
Ambrosia psilostachya	Cuman Ragweed	Native	P	Forb	Wetland, Camp Grafton, Riparian
Amphicarpaea bracteata	American hogpeanut	Native	A	Forb	Riparian
Andropogon gerardii	Big bluestem	Native	P	Grass	Wetland, Camp Grafton, Ekre Grassland
Andropogon hallii	Sand bluestem	Native	P	Grass	Ekre Grassland
Artemisia absinthium	Absinthium	Introduced	P	Forb	Camp Grafton, Riparian
Artemisia campestris	Field sagewort	Native	B	Forb	Camp Grafton
Artemisia frigida	Prairie sagewort	Native	P	Shrub	Camp Grafton
Asclepias incarnata	Swamp milkweed	Native	P	Forb	Riparian
Asclepias ovalifolia	Oval-leaf milkweed	Native	P	Forb	Riparian
Asclepias speciosa	Showy milkweed	Native	P	Forb	Wetland, Camp Grafton
Astragalus canadensis	Canadian milkvetch	Native	P	Forb	Wetland, Camp Grafton
Astragalus flexuosus	Flexible milkvetch	Native	P	Forb	Camp Grafton
Bolboschoenus fluviatilis	River bulrush	Native	P	Sedge	Riparian
Bouteloua curtipendula	Sideoats grama	Native	P	Grass	Wetland, Camp Grafton, Ekre Grassland

Scientific Name[a]	Common Name	Origin[a]	Life	Physiognomy	Study Site
Bouteloua gracilis	Blue grama	Native	P	Grass	Wetland, Camp Grafton, Ekre Wetland
Bromus inermis	Smooth brome	Introduced	P	Grass	Wetland, Camp Grafton, Ekre Grassland
Calamagrostis stricta	Slimstem reedgrass	Native	P	Grass	Wetland, Riparian
Calamovilfa longifolia	Prairie sandreed	Native	P	Grass	Wetland, Ekre Grassland
Calla palustris	Water arum	Native	P	Forb	Riparian
Carex atherodes	Wheat sedge	Native	P	Sedge	Riparian
Carex brevior	Shortbeak sedge	Native	P	Sedge	Wetland
Carex pellita	Woolly sedge	Native	P	Sedge	Wetland, Riparian
Carex praegracilis	Clustered field sedge	Native	P	Sedge	Riparian
Carex sychnocephala	Manyheaded sedge	Native	P	Sedge	Wetland
Carex vulpinoidea	Fox sedge	Native	P	Sedge	Wetland, Riparian
Chamaesyce glyptosperma	Ribseed sandmat	Native	A	Forb	Camp Grafton
Chamaesyce serpyllifolia	Thymeleaf sandmat	Native	A	Forb	Riparian
Chenopodium album	Lambsquarters	Introduced	A	Forb	Camp Grafton
Cirsium canescens	Prairie Thistle	Native	P	Forb	Riparian
Cirsium flodmanii	Flodman's thistle	Native	P	Forb	Wetland
Cirsium vulgare	Bull thistle	Introduced	B	Forb	Wetland
Conyza canadensis	Canadian horseweed	Native	A	Forb	Wetland
Cornus sericea	Redosier dogwood	Native	P	Shrub	Riparian
Dalea candida var. candida	White prairie clover	Native	P	Forb	Camp Grafton
Dalea purpurea var. purpurea	Purple prairie clover	Native	P	Forb	Wetland
Dichanthelium leibergii	Leiberg's panicum	Native	P	Grass	Ekre Grassland
Echinochloa crus-galli	Barnyardgrass	Introduced	A	Grass	Wetland

Scientific Name[a]	Common Name	Origin[a]	Life	Physiognomy	Study Site
Echinocystis lobata	Wild cucumber	Native	A	Forb	Riparian
Eleocharis compressa	Flatstem spikesedge	Native	P	Sedge	Riparian
Eleocharis palustris	Common spikerrush	Native	P	Sedge	Wetland, Riparian
Elyhordeum macounii	Macoun's barley	Native	P	Grass	Wetland
Elymus canadensis	Canada wildrye	Native	P	Grass	Wetland, Camp Grafton, Riparian
Elymus lanceolatus	Thickspike wheatgrass	Native	P	Grass	Camp Grafton
Elymus repens	Quackgrass	Introduced	P	Grass	Wetland, Camp Grafton, Riparian
Elymus trachycaulus	Slender wheatgrass	Native	P	Grass	Wetland, Camp Grafton
Equisetum fluviatile	Water horsetail	Native	P	Fern	Riparian
Equisetum laevigatum	Smooth horsetail	Native	P	Fern	Wetland, Riparian
Erigeron philadelphicus	Philadelphia Fleabane	Native	B	Forb	Wetland
Euphorbia esula	Leafy spurge	Introduced	P	Forb	Wetland, Camp Grafton, Riparian
Euthamia graminifolia	Flat top goldenrod	Native	P	Forb	Wetland
Fragaria virginiana	Virginia strawberry	Native	P	Forb	Riparian
Gaillardia aristata	Blanketflower	Native	P	Forb	Camp Grafton
Galium aparine	Stickywilly	Native	A	Forb	Riparian
Glyceria grandis	American mannagrass	Native	P	Grass	Riparian
Helianthus	Sunflower	Native	P	Forb	Riparian
Hesperostipa comata	Needle-and-thread	Native	P	Grass	Wetland, Ekre Grassland
Hesperostipa spartea	Porcupinegrass	Native	P	Grass	Ekre Grassland
Heterotheca villosa	Hairy false goldenaster	Native	P	Forb	Camp Grafton
Hordeum jubatum	Foxtail barley	Native	P	Grass	Wetland, Camp Grafton
Impatiens capensis	Jewelweed	Native	A	Forb	Riparian

Scientific Name[a]	Common Name	Origin[a]	Life	Physiognomy	Study Site
Iva annua	Annual marsh elder	Native	A	Forb	Riparian
Juncus arcticus	Baltic rush	Native	P	Forb	Wetland, Riparian
Juncus dudleyi	Dudley's rush	Native	P	Forb	Wetland, Riparian
Juncus interior	Inland rush	Native	P	Forb	Wetland
Juncus torreyi	Torrey's rush	Native	P	Forb	Wetland, Riparian
Koeleria macrantha	Prairie junegrass	Native	P	Grass	Camp Grafton, Ekre Grassland
Lappula occidentalis	Flatspine stickseed	Introduced	A	Forb	Camp Grafton
Lycopus americanus	American water horehound	Native	P	Forb	Wetland, Riparian
Lycopus asper	Rough bugleweed	Native	P	Forb	Riparian
Lysimachia hybrida	Lowland yellow loosestrife	Native	P	Forb	Wetland
Medicago lupulina	Black medick	Introduced	P	Forb	Wetland, Camp Grafton
Medicago sativa	Alfalfa	Introduced	P	Forb	Camp Grafton
Melilotus alba	White sweet clover	Introduced	A	Forb	Camp Grafton, Riparian
Melilotus officinalis	Yellow sweet clover	Introduced	A	Forb	Wetland, Camp Grafton, Riparian
Mentha arvensis	Wild mint	Native	P	Forb	Riparian
Muhlenbergia asperifolia	Scratchgrass	Native	P	Grass	Riparian
Muhlenbergia cuspidata	Plains muhly	Native	P	Grass	Ekre Grassland
Muhlenbergia racemosa	Marsh muhly	Native	P	Grass	Wetland
Nassella Viridula	Green needlegrass	Native	P	Grass	Camp Grafton
Oenothera biennis	Common evening primrose	Native	B	Forb	Wetland
Panicum capillare	Witchgrass	Native	A	Grass	Ekre Grassland
Panicum virgatum	Switchgrass	Native	P	Grass	Wetland, Ekre Grassland
Pascopyrum smithii	Western wheatgrass	Native	P	Grass	Wetland, Camp Grafton, Ekre Grassland
Paspalum setaceum	Thin paspalum	Native	P	Grass	Ekre Grassland

Scientific Name[a]	Common Name	Origin[a]	Life	Physiognomy	Study Site
Phalaris arundinacea	Reed canarygrass	Native	P	Grass	Wetland, Ekre Grassland, Riparian
Phleum pratense	Timothy	Introduced	P	Grass	Wetland
Plantago major	Common plantain	Introduced	P	Forb	Riparian
Poa compressa	Canada bluegrass	Introduced	P	Grass	Camp Grafton
Poa palustris	Fowl bluegrass	Native	P	Grass	Camp Grafton, Riparian
Poa pratensis	Kentucky bluegrass	Introduced	P	Grass	Wetland, Camp Grafton, Ekre Grassland, Riparian
Polygonum amphibium	Water smartweed	Native	P	Forb	Riparian
Polygonum arenastrum	Oval-leaf knotweed	Native	A	Forb	Camp Grafton
Polygonum hydropiperoides	Swamp smartweed	Native	P	Forb	Wetland
Polygonum lapathifolium	Curlytop knotweed	Native	A	Forb	Wetland
Populus deltoides	Cottonwood	Native	P	Tree	Wetland, Riparian
Portulaca oleracea	Little hogweed	Introduced	A	Forb	Camp Grafton
Potentilla norvegica	Norwegian cinquefoil	Native	A	Forb	Wetland, Riparian
Potentilla rivalis	Brook cinquefoil	Native	A	Forb	Wetland
Prunus virginiana	Chokecherry	Native	P	Shrub	Riparian
Pseudoroegneria spicata	Bluebunch wheatgrass	Native	P	Grass	Camp Grafton
Ranunculus pensylvanicus	Pennensylvania buttercup	Native	A	Forb	Wetland
Ratibida columnifera	Upright prairie coneflower	Native	P	Forb	Camp Grafton
Rhamnus cathartica	Common buckthorn	Introduced	P	Shrub	Riparian
Rhus glabra	Smooth sumac	Native	P	Shrub	Riparian
Ribes aureum	Golden current	Native	P	Shrub	Riparian
Rubus idaeus	American red raspberry	Native	P	Shrub	Riparian
Rudbeckia hirta	Blackeyed susan	Native	B	Forb	Wetland
Rumex crispus	Curly dock	Introduced	P	Forb	Riparian
Rumex mexicanus	Mexican dock	Native	P	Forb	Wetland, Riparian

Scientific Name[a]	Common Name	Origin[a]	Life	Physiognomy	Study Site
Salix amygdaloides Andersson	Peachleaf willow	Native	P	Tree	Wetland, Riparian
Salix bebbiana	Bebb willow	Native	P	Shrub	Wetland, Riparian
Salix interior Rowlee	Sandbar willow	Native	P	Shrub	Wetland, Riparian
Salix petiolaris	Meadow willow	Native	P	Shrub	Wetland
Salsola tragus	Prickly Russian thistle	Introduced	A	Forb	Camp Grafton
Schedonorus pratensis	Meadow fescue	Introduced	P	Grass	Wetland
Schizachyrium scoparium	Little bluestem	Native	P	Grass	Wetland, Camp Grafton, Ekre Grassland, Riparian
Schoenoplectus pungens	Common threesquare	Native	P	Sedge	Riparian
Scirpus pallidus	Cloaked bullrush	Native	P	Sedge	Wetland
Setaria pumila	Yellow foxtail	Introduced	A	Grass	Ekre Grassland
Solidago canadensis	Canada goldenrod	Native	P	Forb	Wetland, Camp Grafton, Riparian
Solidago missouriensis	Missouri goldenrod	Native	P	Forb	Wetland, Riparian
Sonchus arvensis	Field sow thistle	Introduced	P	Forb	Riparian
Sorghastrum nutans	Indiangrass	Native	P	Grass	Wetland, Ekre Grassland
Sparganium eurycarpum	Broadfruit bur-reed	Native	P	Forb	Riparian
Spartina pectinata	Prairie dordgrass	Native	P	Grass	Wetland, Ekre Grassland, Riparian
Sporobolus cryptandrus	Sand dropseed	Native	P	Grass	Ekre Grassland
Symphoricarpos albus	Common snowberry	Native	P	Shrub	Riparian
Symphoricarpos occidentalis	Western snowberry	Native	P	Shrub	Riparian
Symphyotrichum ericoides	White heath aster	Native	P	Forb	Riparian
Symphyotrichum laeve	Smooth blue aster	Native	P	Forb	Riparian
Symphyotrichum lanceolatum	White panicle aster	Native	P	Forb	Wetland, Riparian

Scientific Name[a]	Common Name	Origin[a]	Life	Physiognomy	Study Site
Taraxacum officinale	Common dandelion	Introduced	P	Forb	Wetland, Camp Grafton, Riparian
Thinopyrum intermedium	Intermediate wheatgrass	Introduced	P	Grass	Camp Grafton
Trifolium pratense	Red Clover	Introduced	P	Forb	Wetland, Riparian
Trifolium repens	White clover	Introduced	P	Forb	Wetland, Riparian
Typha x glauca	Hybrid cattail	Introduced	P	Forb	Wetland
Urtica dioica	Stinging nettle	Native	P	Forb	Riparian
Verbena bracteata	Prostrate verbena	Native	A	Forb	Camp Grafton
Verbena stricta	Hoary verbena	Native	P	Forb	Wetland, Riparian
Vicia americana	American vetch	Native	P	Forb	Riparian
Viola canadensis var. rugulosa	Creepingroot violet	Native	P	Forb	Riparian
Zigadenus elegans	Mountain deathcamas	Native	P	Forb	Riparian
Zizia aptera	Meadow aptera	Native	P	Forb	Riparian

[a] Scientific names and species origins are from The PLANTS Database: USDA, NRCS. 2021. The PLANTS Database (http://plants.usda.gov, 21 September 2021). National Plant Data Team, Greensboro, NC 27401-4901 USDA. Plant origins are based on native or introduced status within the contiguous United States.