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Long term recovery from cattle grazing disturbance and climate impacts at Capitol Reef National Park, Utah

Erin Tessens
University of Northern Colorado

UNIVERSITY OF NORTHERN COLORADO

Greeley, Colorado

The Graduate School

LONG TERM RECOVERY FROM CATTLE GRAZING DISTURBANCE AND CLIMATE IMPACTS AT CAPITOL REEF NATIONAL PARK, UTAH

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Erin Tessens

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Accepted by the Thesis Committee:

Dr. Scott Franklin, Ph.D., Chair

Dr. Emily Holt, Ph.D., Committee Member

Dr. Mitchell McGlaughlin, Ph.D., Committee Member

Accepted by the Graduate School

Jeri-Anne Lyons, Ph.D.
Dean of the Graduate School
Associate Vice President for Research

ABSTRACT

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Cattle grazing has influenced the environment in the western United States since European settlement in the 1800's. Continuous and heavy grazing on arid and semi-arid rangelands has resulted in decreased biodiversity, changes in vegetation structure, and vulnerability to exotic plant invasion. Heavy grazing has also been linked to decreased cryptobiotic soil due to trampling and susceptibility to erosion. With a lack of effective means of successful habitat restoration, there is a rising concern among land managers to maintain these intricate systems, notably under the threat of climate change. Consequently, there is a critical need to understand these system's response to grazing pressure and resilience once released from such pressure, especially on a long-term scale. To address this problem, we studied various attributes (i.e., cryptobiotic soil, vegetation, and soil properties- among seven exclosure locations on the rangeland of Capitol Reef National Park, Utah. These exclosures were built in the 1980's, were monitored for six years, and have not been observed since initial monitoring from 1984-1989. We found observable differences when comparing inside versus outside the exclosures under a variety of grazing histories. Treatment differences included percent ground cover, vegetation trends, soil stability, and cryptobiotic soil attributes. Additionally, we found significant changes in these attributes over time. One of the more notable changes was that of significant increase in cryptobiotic soil cover over all treatments across the park. Finally, we found that drought may have an overarching, greater influence over rangeland communities than grazing or grazing history. Future long-term research on arid/semi-arid landscapes should further examine the relationship of vegetation and cryptobiotic soil under both heavy grazing regimes and long-term drought conditions. Greater understanding of these changes on disturbed lands, especially under the threat of climate change, will better equip land managers to make sustainable and successful landscape decisions.

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CHAPTER I

INTRODUCTION TO THE STUDY SYSTEMS: CATTLE DISTURBANCE, AND RANGELAND COMMUNITY ATTRIBUTES

Summary

Across the United States, 640 million acres of federal land have been set aside for public use. Collectively, the National Park Service (NPS), Bureau of Land Management (BLM), US Fish and Wildlife Service (USFWS), and the US Forest Service (USFS) manage most of these areas, which are located primarily in the Western United States. Public lands provide many ecosystem services. Ecosystem services are the benefits to humans provided by the natural environment and healthy ecosystems. However, tradeoffs exist that may be prioritized differently to different management agencies. One land management practice may promote one service, but reduce another, which has led to conflict (Bennett et al., 2009) on how to utilize, sustain, and restore public lands.

The Colorado Plateau, about 340,000 km^2, is located near and around the Four-Corners region of Arizona, Colorado, New Mexico, and Utah and is an ecosystem almost fully within public land that provides multiple ecosystem services. About 75% of the Colorado Plateau is managed by federal and tribal agencies (Winkler et al., 2018). Being one of North America’s five major desert ecosystems, its ratio of total annual precipitation is less than two-thirds of potential evapotranspiration (Yang et al., 2012; Poitras et al., 2018). Despite water limitation of desert ecosystems, they often support high levels of biodiversity with a large variety of endemic plant and animal species (Stohlgren et al., 2005). Supporting this biodiversity trend, the Colorado Plateau contains the third greatest number of endemic species across all taxonomic groups in North America (Daily et al., 1999). In addition, these cool desert ecosystems offer a variety of ecosystem services, including grazing on 45 million acres of land in Utah alone. Sometimes overlooked, these ‘services’ (from grazing to recreational hiking) are also ecosystem disturbances.

The Disturbance of Cattle Grazing

When disturbances occur over a landscape, the community structure and diversity of an ecosystem changes at all scales (Sousa, 1984). Different disturbance components such as frequency, size, intensity, and severity can each alter landscape responses and resilience to disturbance differently. In water-limited ecosystems such as the Colorado Plateau, recovery is often slow following disturbance (Poitras et al., 2018).

Dryland ecosystems are important in supporting global biodiversity, and also support the majority of the world’s livestock (Yang et al., 2012). Being the historically dominate land use in the western United States (Bigelow & Borchers, 2017), cattle grazing in areas such as the Colorado Plateau have been a point of debate on how to manage and restore areas experiencing grazing disturbances. In the Colorado Plateau, there has been a dominate human impact on the landscape due to grazing over the past two centuries, where overgrazing has led to both short- and long-term negative effects on soils, cryptobiotic soil crust and vegetation (Cole et al., 1997; Ware et al., 2014).

Cryptobiotic Soil Crust

Cryptobiotic soil inhabits the top surface layer of dryland soils. It consists of cyanobacteria, algae, micro fungi, lichens and bryophytes and soil particles interacting together (Concostrina-Zubiri et al., 2019). This living soil plays an important role in the desert community and is associated with higher plant species diversity and richness (Rosentreter and Root, 2019). Cryptobiotic soil can be the biggest source of limiting nutrients, such as nitrogen, for desert communities (Belnap, 2002). It is also associated with greater uptake of other essential elements in plants, such as copper, potassium, magnesium, and zinc (Harper and Belnap, 2001). Areas covered in cryptobiotic soil have higher seed numbers and viability of those seeds (Stohlgren et al., n.d.). In addition, cryptobiotic soil can decrease populations of annual invasive plant species as the lichen in the crust provides a physical barrier against colonization and expansion (Rosentreter and Root, 2019). As a physical barrier, it also reduces soil erosion (Belnap & Gillette, 1998).

Cryptobiotic soil crust has many benefits for plant communities, and thus damage to cryptobiotic soil has become a major concern for land managers. Some of the biggest threats to cryptobiotic soil health is physical damage and altered climate (Young et al., 2016). Cover of cryptobiotic soil are shown to be related to disturbances such as invasion of *Bromus tectorum*, grazing, and fires (Condon & Pyke, 2018). Loss of cryptobiotic soil can result in loss of ecosystem function at a larger scale (Belnap, 2002; Condon & Pyke, 2018).

Once cryptobiotic soil is lost, restoration becomes another major issue. Time required to restore cryptobiotic soils was historically thought to be measured in centuries

(Belnap & Warren, 2002). However, more recent studies in Australia observed passive recovery began to stabilize after 20 years, but sites with past grazing stabilized to a lower cover level (Read et al., 2011). Sites with different types of cryptobiotic soil and other environmental factors will respond differently. For example, compared to other forms of lichen, crustose and squamulose lichens are expected to be more sensitive to trampling (Aquilar et al., 2009). Although there is a recent increase in studies on cryptobiotic soil, successful restoration and long term recovery trends are unknown (Herrick et al., 2001).

Soil

One important indicator of ecosystem health is soil surface stability as it is sensitive to complex changes in physical, chemical, and biological processes (Herrick et al., 2017; Miller, 2005). There is evidence that grazing leads to major changes in physical properties of soil, including decreasing nutrient availability (Belnap and Eldridge, 2003; Hiernaux et al., 1999; Neff et al., 2005). Large ungulates, such as cattle cause physical soil compaction. This can restrict water filtration, root growth, and activity of microorganisms (Herrick et al., 2006). Physical disturbance also enables invasive species to colonize, and sometimes outcompete native species. With fecal pats of cattle have been found to have higher species richness for annual exotic grasses (Bartuszevige & Endress, 2008), it provides additional introduction to exotic species in these disturbed areas. In drought years, annual plants often do not germinate. Coupled with lack of root stability, this leaves the soil barren and vulnerable to erosion (Belnap et al., 2009).

Soil Nutrients

Soil properties on the Colorado Plateau are heavily influenced by their geologic parent material, tectonic faulting, and aeolian processes. Sedimentary rock, such as sandstone, silt, limestone, and shales are the most dominate parent material on this landscape (Duniway et al., 2016; Reynolds et al., 2006). Desert ecosystems are generally characterized by little organic matter, low soil moisture, and high alkalinity. (Gaitán et al., 2018; Noy-Meir, 1973).Studies in areas with urine and dung excreted by livestock have shown enhanced mineral availability by increasing nitrogen cycling and providing soluble nitrogen, that is available for plant growth (Holland et al., 1992; McNaughton et al., 1997). Other studies have found that removal of plants due to overgrazing, which reduced the topsoil layer due to erosion, and have been the main factors for reduction of soil organic matter contents and loss of essential nutrients (Oliveira Filho et al., 2019; Schulz et al., 2016; Tainton et al., 2000. In an arid rangeland excluded from grazing for 17 years, Carbon: Nitrogen, Carbon: Phosphorous, Nitrogen: Phosphorous, and Phosphorous: Potassium ratios had no variation while Calcium^{2+} and Potassium+ increased and the Aluminum^{3+} content in soil decreased with grazing exclusion (Oliveira Filho et al., 2019). As monitoring techniques based on soil properties quantifying integrity of nutrient-related processes have not been fully developed (Havstad et al., 2000), further investigation on changes in soil nutrients may provide novel insight in soil and vegetation recovery.

Vegetation

Since historically the Intermountain West, including the area where our study site is located, occurred without the presence of large ungulate herds, vegetation here lacks some adaptations and resilience to grazing, specifically cattle (Fernandez et al., 2008; Schwinning et al., 2008). Over the past two centuries studies of the Colorado Plateau show a dominate human impact on the landscape through grazing (Cole et al., 1997).

Livestock trample and defoliate individual plants, which decreases plant biomass of native plants (Cook & Child, 1971) and may also negatively affect reproductive success. Plants may convert to less productive growth forms, such as sod forms of grasses (Holechek and Galt, 2000; National Park Service [NPS], 2018; Vallentine, 2001;). Decreased fitness of native plant species combined with an increase in physical disturbance can increase invasion of non-native plant species and decrease native species diversity, changing plant community structure as a whole (Bartuszevige & Endress, 2008). Since the cover and type of vegetation influence soil stability (Okin, 2008), changes in plant communities can have negative implications. Many of the impacts of grazing may be enhanced through negative feedbacks in conjunction with climate change (Belote et al., 2009).

Climate Change

Climate change is expected to increase overall aridity with more extreme and prolonged droughts in the United States desert southwest (Seager et al., 2007). Because the Colorado Plateau lies at the boundaries of two climate zones, it is expected to have more extreme fluctuations in climate compared to other arid regions (Schwinning et al., 2008). With grazing shown to alter the way rangeland communities respond to climate change (Belote et al., 2009; Loesser et al., 2007), the Colorado Plateau may be even more prone to extreme changes over the landscape.

In the Colorado Plateau, weather stations have shown that over time, summer precipitation has decreased (NPS, 2020). Additionally, over the last 30 years, the Colorado Plateau has experienced a 0.2°C to 0.5°C average temperature increase, with warmer temperatures more pronounced in the cold season (NASA, 2019; Schwinning et al., 2008).

Changes in temperature and precipitation may have large effects on native vegetation as total annual primary productivity in perennials are largely influenced by winter precipitation (Caldwell, 1985). Shifts from dominance by cool-season grasses to warm-season grasses as well as increased populations of invasive plant species, have already been observed (NPS, 2018). In areas such as semi-arid grasslands and shrublands with slow growing vegetation, resilience to climate-related disturbances, such as severe droughts, may be low (NPS, 2017). Due to the known negative impacts of grazing in these systems coupled with climate change, land managers are now more focused on changing cattle grazing regimes to minimize their impact.

Ecological Site Descriptions

Ecological site descriptions were developed by the US Department of Agriculture Natural Resource Conservation Service as way to classify land, management, and monitoring systems focused on specific ecological site (Doherty et al., 2011). In term of a rangeland, an ecological site is "a distinctive kind of land where specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation" (USDA-NRCS, 2006). They are based on changes in soil, aspect, topography, and moisture conditions. These descriptions were developed to provide management tools for vegetation, restoration, and risk and assessment and monitoring decisions (Herrick et al., 2006) and provide the framework for understanding and predicting patterns on rangeland (Spiegal et al., 2016).

In North America, the Reference Community is the vegetation community that existed at the time of European immigration and settlement (USDA-NRCS, 2006). This is the community in dynamic equilibrium with its environment. Natural disturbance and disturbance patterns that occurred here did not displace the plant community. Using Ecological Site Descriptions and comparing them to their Reference Community can give us an understanding of how different locations have or have not deviated from their ideal conditions.

Restoration

Shifts in plant community composition due to grazing is a slow process, but recovery back to its original composition can be just as long (Fernandez et al., 2008). Despite rapid development of research on restoration techniques in dryland ecosystems, current methods are unsuccessful on the rangeland and our understanding of how to restore these ecosystems remains poor (Schwinning et al., 2008; Winkler et. al., 2018). Additionally, there are no standard strategies on how to restore these landscapes (Bernstein et al., 2014; Winkler et al., 2018,). Since two-thirds of rangelands do not respond to current management practices (Peters et al., 2006), understanding these changes during and after disturbance are vital to the health of the landscape. In the absence of long-term post-disturbance and recovery data (Anderson et al., 2008; Bennett et al., 2009; Reich and Lake, 2015), it is hard to determine how communities recover and thus what land managers can do to facilitate healthy recovery. The focus on recovery and healthy rangelands includes three main attributes: soil/site stability, hydrologic function, and biotic integrity (Pyke et al., 2002) To measure these changes, we focused on observation of soil, cryptobiotic soil crust, and vegetation at Capitol Reef National Park.

Capitol Reef National Park

Within the Colorado Plateau lies one of Utah's five National Parks, Capitol Reef National Park, established in 1971. Covering approximately 242,000 acres, Capitol Reef is home to a wide range of environments and accompanied diverse flora and fauna.

In the 1870's, Mormon pioneers began to settle in lands within and near where Capitol Reef National Park is now located. By 1890, there were over 2800 cattle and about 60,000 sheep in the area. When the area officially became part of a national park, much of the livestock numbers were reduced, however, some ranchers maintained their grazing rights on the lands that are now managed by the National Park Service (Snow, 1953). Livestock grazing and trailing, i.e., moving livestock across park lands between winter and summer ranges on adjacent Federal lands, of both sheep and/or cattle have had influence on the land in the Park (Frye, 1998), with 19 grazing allotments created when it reached National Park status (NPS, 2018). Over time, the National Park Service purchased many of the grazing permits from the rancher permitters. By the 1990's, most of the allotments in Capitol Reef ceased seasonal grazing except for two allotments, Hartnet and Sandy 3 (NPS, 2018).

In March of 2018, the grazing permit for the Hartnet Allotment, one of the two remaining park grazing allotments, was purchased by a non-governmental organization, ceasing grazing within this area (NPS, 2018). Since 1954, the Hartnet allotment had provided winter grazing which occurred mid-October through May every year (NPS, 2018; Williams, 1989). Before 1954, it was used for year-round grazing (Williams et al., 1995). This left the Sandy 3 allotment as the only active allotment in Capitol Reef, currently grazed in the winter months between October and May (NPS, 2018). Although continuous, seasonal winter grazing has been removed from much of the park, some cattle trailing, still occurs throughout the park and Hartnet allotment. (NPS, 2018).

Between 1983 and 1986, seven grazing exclosures were built in various allotments around the park: Surprise Canyon, Cathedral, Muley Twist, Hartnet, Red Slide, The Post, and South Desert. These grazing exclosures were fenced in barbwire squares (33m x 33m) that prevented cattle from accessing the area inside. They were placed among different allotments (i.e., areas of grazing) with different grazing histories. These exclosures were paired with an identical layout just outside the fence, accessible to grazers. Their purpose was used as a comparison to aid in determining how climate and grazing influenced plant community dynamics. Data on these exclosures were collected between 1984-1993. However, after 1993, the exclosures data collection ceased and were not observed until the present study.

Current Data and Research Questions

Using data from 1984-1993, Belote et al. (2009) performed a study on vegetation of three exclosure sites located in Capitol Reef National Park; Surprise, Hartnet and Cathedral to investigate how grazing and climate influenced shift in species composition and relative community stability. Although this study was never published, they found that grazing can alter the way the rangeland communities respond to climate. Grazing appeared to change relative compositional stability in response to climate pulses and suggest that grazed sites tend to be less resilient than non-grazed locations.

The Northern Colorado Plateau Network monitors much of the land in the park, including climate, invasive exotic plants, and landscape dynamics. From 2009-2018, the Cathedral allotment was dominated by warm-season grasses, low levels of invasive exotic species and relatively good soil (NPS, 2020). From past, but limited historical data, this showed good recovery from past grazing in this area.

The Hartnet allotment, which was recently released from grazing pressure in 2018, did not show the same trend. Cool season grasses had low frequency, while invasion of exotic species was high. Although it showed improvement in soil parameters, it still contains high potential for erosion (United States Department of Agriculture et al., 2014).

Given the current observations in Capitol Reef and need for further understanding of post-disturbance effects on the landscapes overall, especially long-term effects, this study aims to evaluate how communities in semi-arid desert ecosystems passively recover after long term, heavy grazing. We evaluated these effects using data sampled seven times, unevenly, over 36 years. By using the exclosures constructed in the 1980's, examining changes inside versus outside the exclosures and over time, we hypothesized:

H1 Through time, areas inside exclosures will show higher diversity of species, different community structure, greater cryptobiotic soil cover and greater bare ground cover than areas outside.

H2 In 2020, areas inside the exclosures will show higher cryptobiotic soil darkness, different cryptobiotic soil morphology, greater soil stability, and higher levels of soil nutrients than areas outside of exclosures.

H3 In 2020, inside exclosures will show soil and vegetation ratings more similar to Ecological Site Description Reference State values for these locations than outside exclosures.

We tested these hypotheses by comparing various attributes inside versus outside of each of the seven exclosures as well as over time. In the summer of 2020, we revisited these exclosures to collect the current data, following the previous data collection protocol of the 1980's (Graham, 1987). Measurements taken were percent cover of various attributes, plant species diversity, shrub cover, and vegetation height. In addition to these protocols, we conducted soil stability tests using a Soil Stability Kit and its protocol (Herrick et al., 2001) to determine the soil's resistance to erosion. Soil core samples were also obtained to collect information on bacterial and fungal species to observe soil diversity inside versus outside of the exclosures.

CHAPTER II

INVESTIGATION OF THE IMPACTS OF CATTLE GRAZING DISTURBANCE ON RANGELAND COMMUNITIES IN CAPITOL REEF NATIONAL PARK, UTAH

Introduction

When disturbances occur over a landscape, the community structure and diversity of an ecosystem changes at all scales (Sousa, 1984). Different disturbance components such as frequency, size, intensity, and severity can each alter landscape responses and resilience to disturbance differently. In water-limited ecosystems such as the Colorado Plateau, recovery is often slow following such disturbance (Poitras et al., 2018).

Dryland ecosystems are important in supporting global biodiversity, and the majority of the world's livestock (Yang et al., 2012). Being the historically dominate land use in the western United States (Bigelow & Borchers, 2017), cattle grazing in areas such as the Colorado Plateau have been a point of debate on how to manage and restore areas experiencing grazing disturbance. In the Colorado Plateau, overgrazing has led to both short- and long-term negative effects on soil erosion, cryptobiotic soil crust and vegetation (Cole et al., 1997; Ware et al., 2014).

Consisting of cyanobacteria, algae, micro fungi, lichens and bryophytes in various amounts and soil particles interacting together (Concostrina-Zubiri et al., 2019), cryptobiotic soil plays an important role in desert communities as ecosystem engineers (Rosentreter and Root, 2019). Often, it is largest source of nitrogen for desert communities (Belnap, 2002). It also plays an important role in soil stability and can decrease populations of annual invasive species by acting as a physical barrier (Belnap & Gillette, 1998; Rosentreter and Root, 2019).

As physical damage to cryptobiotic soil is one of its bigger threats, and loss of cryptobiotic soil can result in loss of ecosystem function at a large scale (Belnap, 2002; Condon and Pyke, 2018; Young et al., 2016), damage to this ecosystem component has become a major concern for land managers. Historically, restoration was thought to be measured in centuries (Belnap and Warren, 2002). However, more recent studies in Australia observed passive recovery began to stabilize after 20 years, albeit sites with past grazing stabilized to a lower cover value (Read et al., 2011). Although there are recent increases in studies on cryptobiotic soil, successful restoration and long-term recovery trends are still unknown (Herrick et al., 2001).

One important indicator of ecosystem health is soil surface stability as it is sensitive to complex changes in physical, chemical, and biological processes (Miller, 2005; Herricks et al., 2017). Grazing leads to major changes in physical properties of soil, including decreasing nutrient availability (Belnap and Eldridge, 2003; Hiernaux et al., 1999; Neff et al., 2005) and physical soil compaction by large ungulates. This can restrict water filtration, root growth, and activity of microorganisms (Herrick et al., 2006), and physical disturbance also enables invasive species to colonize, and sometimes dominate the landscape. In addition, fecal pats of cattle have been found to have higher species richness for annual exotic grasses (Bartuszevige & Endress, 2008).

Livestock on the rangeland trample and defoliate individual plants, which decreases the biomass of native plants (Cook & Child, 1971) and may also negatively affect reproductive success. Plants may convert to less productive growth forms, such as sod forms of grasses (Holechek and Galt, 2000; NPS, 2018; Vallentine, 2001). Decreased fitness of native plant species combined with an increase in physical disturbance can increase invasion of non-native plant species and decrease native species diversity, changing plant community structure (Bartuszevige & Endress, 2008). Since the cover and type of vegetation influence soil stability (Okin, 2008), changes in plant communities can have negative implications.

To classify and predict changes in plant communities and other attributes, such as soil, ecological site descriptions were developed by the US Department of Agriculture's Natural Resource Conservation Service. This provides a way to classify land, manage, and monitor systems on specific ecological sites (Doherty et al., 2011). For a rangeland, an ecological site is "a distinctive kind of land where specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation" (USDA-NRCS, 2006). These descriptions were developed to provide management tools for vegetation, restoration, and risk assessment and monitoring decisions (Herrick et al., 2006), and they provide the framework for understanding and predicting patterns on rangeland (Spiegal et al., 2016). In North America, the Reference Community is the vegetation community that existed at the time of European immigration and settlement (USDA-NRCS, 2006). Using Ecological Site Descriptions and comparing them to their Reference Community can give us an understanding of how different locations have or have not deviated from their more natural conditions.

Our understanding on how to restore arid/semi-arid ecosystems remains poor with no standard strategies on how to restore these landscapes (Bernstein et al., 2014; Schwinning et al., 2008; Winkler et. al., 2018). With two-thirds of rangelands not responding to current management practices (Peters et al., 2006), understanding the changes during and after disturbance are vital to the health of the landscape. In the absence of long-term post-disturbance and recovery data (Anderson et al., 2008; Bennett et al., 2009; Reich and Lake, 2015), it is hard to determine how communities recover and thus what land managers can do to facilitate recovery. To measure such changes, we focused on observation of soil, cryptobiotic soil crust, and vegetation at Capitol Reef National Park.

When Capitol Reef National Park was designated as a park in 1971, some ranchers who previously used the area for cattle grazing maintained their grazing rights (NPS, 2018; Snow, 1953; Williams, 1989). Heavy grazing influences here had been present for about 150 years (Frye, 1998). By the 1990's, most of the allotments in Capitol Reef ceased seasonal grazing except for two allotments, Hartnet and Sandy 3 (NPS, 2018). In March of 2018, grazing was also removed from the Hartnet allotment (NPS, 2018).

Given the current observations in Capitol Reef and need for further understanding of post-disturbance effects on the landscapes overall, especially long-term effects, this study aims to evaluate how communities in semi-arid desert ecosystems passively recover after long term, heavy grazing. We evaluated these effects using data sampled seven times, unevenly, over 36 years. By using the exclosures constructed in the 1980's, examining changes inside versus outside the exclosures and over time, we hypothesized:

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We tested these hypotheses by comparing various attributes inside versus outside of each of the seven exclosures as well as over time. In the summer of 2020, we revisited these exclosures to collect the current data, following the previous data collection protocol of the 1980's (Graham, 1987). Measurements taken were percent cover of various attributes, plant species diversity, shrub cover, and vegetation height. In addition to these variables, we conducted soil stability tests using a Soil Stability Kit and its protocol (Herrick et al., 2001) to determine the soil's resistance to erosion.

Methods

Study Area

The study area was located within Capitol Reef National Park which lies in southcentral Utah near the town of Torrey (Figure 1a). Capitol Reef National Park is part of the Colorado Plateau desert ecosystem that encompasses about 340,000km^2 (242,000 acres) of the western United States. It is the second largest park in Utah. Elevation of the park ranges from 1219m nears the southern tip of the park, Halls Creek, to 3353m on the north boarder near Thousand Lake Mountain. Average precipitation from the weather station, located near the visitor's center (central area of the park), is 20.3 cm annually with most precipitation occurring July through September, Capitol Reef's monsoon season (NPS, 2018). Temperatures average a high of 30ºC in July to a low of 3ºC in January (NPS, 2018). With this diversity of elevation and precipitation, soils and vegetation are also diverse across the park. Out of the 175 vegetation community types identified at Capitol Reef, 58 of those are woodlands and saltbush shrublands (NPS, 2020). Pinyon-Juniper/ Mesic Shrubs Woodlands Complex account for the most frequent vegetation map class (NPS, 2020).

Exclosure Description

Between 1983 and 1986, seven grazing exclosures were built within the park: Surprise Canyon, Cathedral, Muley Twist, Hartnet, Red Slide, The Post, and South Desert (Figure 1b; Table 1). These exclosures were built from barbwire fencing to exclude cattle from the area. There were signs of other grazers having access inside the exclosure fence. Rabbit and elk dung were found in some locations. Immediately adjacent to each exclosure is a replicate design, not surrounded by a fence (i.e., exposed to grazing). These locations at the northern and southern ends of the park, where grazing historically and currently occurs. Each exclosure is 33m x 33m, except for Cathedral and Surprise which were enlarged with two additional exclosures each immediately adjacent to the original. Data for these adjacent-enlarged plots were combined when doing calculation to one location for each treatment (each site is a replicate; n=7). These two larger exclosures were intended to study a variety of range improvements, but those studies were never implemented (Sandra Borthwick, pers. comm.).

At each site, 20 1m^2 plots were established randomly; ten inside the exclosure and ten outside, Cathedral being an exception with eleven plots, and Cathedral and Surprise each having three exclosures. These plots are marked by conduit in the southwest corner of the plot, except Surprise whose markers are in the southeast corner. In addition, four

30m line-intercept transects were established, with two inside the exclosure and two outside also marked by conduit (Figure 2).

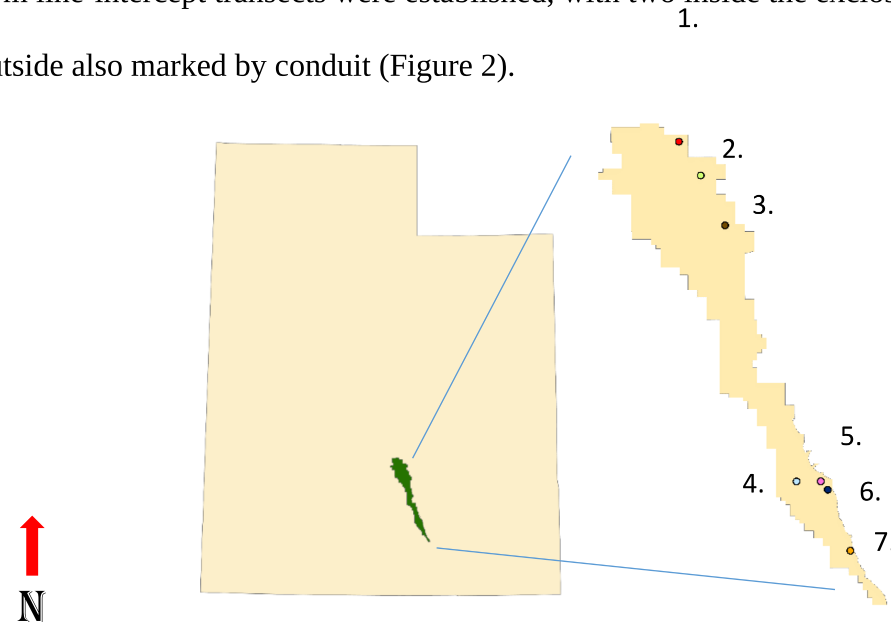

Figure 1: Capitol Reef National Park's location within the state of Utah, United States of America, and locations of exclosure sites within the boundary of Capitol Reef National Park: 1. = Cathedral Valley, 2 = Hartnet, 3 = Lower South Desert,4 = Muley Twist, 5 = Surprise, 6 = Post, 7 = Red Slide. Maps by Blackford, Anna.

Table 1: UTM locations of exclosures within Capitol Reef National Park, Utah, Zone 12S. Cathedral being the most northern plot moving southward to Red Slide exclosure.

Exclosure Name	UTM Easting	UTM Northing
Cathedral	476,785	4,261,190
Hartnet	480,473	4,253,901
Lower South Desert	484,558	4,243,118
Surprise	500,846	4,187,967
Muley Twist	496,701	4,187,895
The Post	502,021	4,186,154
Red Slide	505,911	4,173,026

Table 2: The year each exclosure was constructed and the years they had data collection in Capitol Reef National Park, Utah. Cathedral, Hartnet and Muley Twist were constructed in 1983, however, data collection did not begin until 1984.

Exclosure Name	**Year Constructed**	**Data Collected**						
		1984	1985	1986	1987	1988	1989	2020
Cathedral	1983	x	x	x	x	x	x	x
Hartnet	1983	x	x	x	x	x	x	x
Lower South Desert	1986			x	x	x	x	x
Surprise	1984	x	x	x	x	x	x	x
Muley Twist	1983	x	x	x	x	x		x
The Post	1985		x	x	x	x		x
Red Slide	1985		x	x	x	x		x

Table 3: Known grazing history at each exclosure site in Capitol Reef National Park, Utah.

Exclosure Name	*Start Grazing*	*End Grazing*	*Total Years Grazed*	*Density of Allotment (AUM[1])*
Cathedral	1870-1880s	~80% AUM's off 1989, 100% off 1999	119-129	~300-500
Hartnet	1870-1880s	5/1/2018	138-148	1141
Lower South Desert	1870-1880s	5/1/2018	138-148	1141
Surprise	1870-1880s	currently grazed	~150	unknown
Muley Twist	1870-1880s	currently grazed	~150	unknown
The Post	1870-1880s	currently grazed	~150	unknown
Red Slide	1870-1880s	1989	109-110	unknown

Table 4: Description of each site location, including its elevation, ecological site name, ecological site number, and soil type in Capitol Reef National Park, Utah. Total is the number of years of grazing. Density refers to the amount of animal units, which are defined as a cow and calf pair.

Exclosure Name	Elevation	Ecosite Name	Ecosite Classification	Soil Type
Cathedral	1447m	Desert Alkali Sandy Loam	R035XY101UT	Monue-Myton-Uzona complex
Hartnet	1447m	Desert Alkali Sandy Loam	R035XY101UT	Monue-Myton-Uzona complex
Lower South Desert	1426m	Desert Very Shallow Gypsum (Torrey's Jointfir)	R035XY142UT	Goblin-Ivanpatch complex,
Surprise	1447m	Semidesert Gravelly Loam (Shadscale)	R035XY242UT	Begay, saline-Querencia, saline-sodic complex
Muley Twist	1446m	Semidesert Shallow Loam (Utah Juniper-Pinyon)	R035XY221UT	Reef-Rizno-Rock outcrop complex
The Post	1410m	Semidesert Gravelly Loam (Shadscale)	R035XY242UT	Begay, saline-Querencia, saline-sodic complex
Red Slide	1448m	Sandy Bottom	R035XY015UT	Sandyranch-Radnik-Riverwash complex

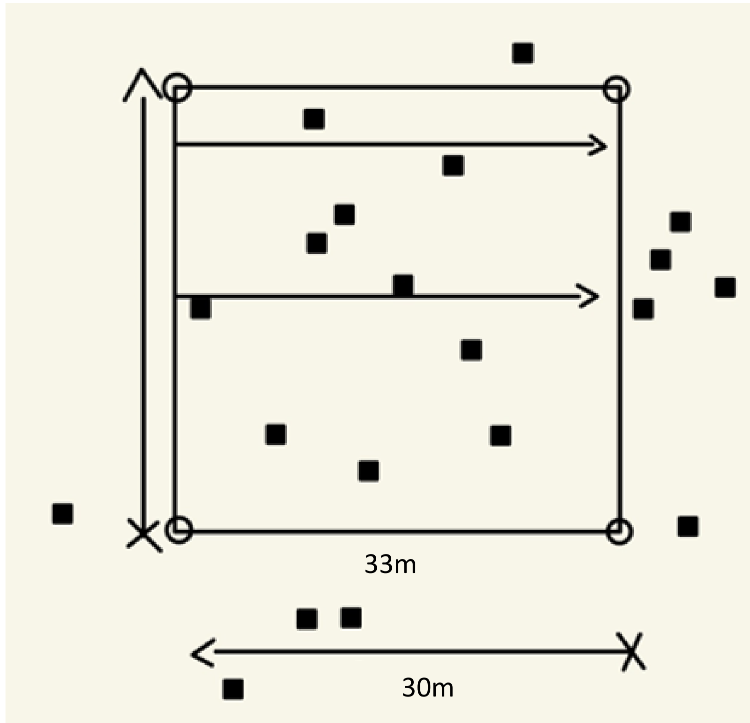

Figure 2: Example site setup of plots and transects inside and outside of the exclosures at Capitol Reef National Park, Utah. The large square with open dots on each corner represents the 33m x 33m exclosure. Black squares represent the 10 randomly assigned 1m x 1m plots, inside and outside of the exclosure. “X” is the start and arrow heads represent the end of the 30m transect, with two transects inside and two outside.

Field Measurements: Original Dataset

Data collected between 1984-1989 followed protocol from Graham (1987). In 2020, we continued to follow the original protocol.

At every exclosure site, we recorded GPS coordinates (Table 1) along with photos of the area from the southwest perspective. In compliance with the National Park Service, before any data collection was conducted, a threatened and endangered species survey was completed. In addition, each study area was cleared by an archeologist before soil data were collected.

Measurements within each of the $1m^2$ plots marked by conduit (10 inside, 10 outside), we used a $1m^2$ PVC plot frame. The frame was positioned so the sides were parallel to the fence line of the exclosure. Data was collected following Graham's (1987) protocol.

Transects were already monumented with conduit pipe on each end. To record data along transects, we ran a meter tape from one end the other. Data along the transect, described below, was taken on the right side of the transect with the observer on the left at all locations. This prevented disturbance on the side of the transect where it could affect soil stability ratings.

Cover and Vegetation Measurements

In each of the twenty $1m^2$ plots, ten inside and ten outside of the exclosure, visual estimates of the percent cover were taken. Percent cover was estimated for live individuals of each plant species (rooted inside the frame or not), cattle dung, cryptobiotic soil, ant hill, and bare ground. From the transect line, the start and end position of a shrub on the line were recorded for each shrub species, live or dead.

If a plant could not be identified in the field, Assessment Inventory and Monitoring's methods (Toevs et al., 2011) were followed. The unidentified plant was given a generic code name based on the plant type: AF= annual forb, PF= perennial forb, G=grass/graminoid, SH=shrub, T=tree. It also received a unique number in that category. For example, AF01, SH01, G03. Pictures of each unknown plant, a detailed description, as well as a specimen were collected. Once identified and keyed to species, the specimen was discarded, per the permit requirements from the National Park Service. Identifying information about the unknown plant included- date found, exclosure location, potential name of identification, and a description of the specimen.

Additional Measurements

In 2020, additional methods were added to the protocol. These included cryptobiotic soil darkness, cryptobiotic soil morphology, site soil stability, and soil core samples.

Cryptobiotic Soil

When cryptobiotic soil was present in any of the 1m² plots we also rated the dominant cryptobiotic soil darkness present on a scale of 1-6 using darkness levels for the Colorado Plateau (Belnap et. al., 2007) Darkness indicates the level of cyanobacteria present in the cryptobiotic soil (Belnap et al., 2007). The crust was also rated morphologically as smooth, rugose, pinnacled or rolling.

Soil Stability

Following Assessment Inventory and Monitoring techniques (Toevs et al., 2011) surface soil samples were collected along each transect in 3m increments using Jornada Experimental Range Soil Stability Test Kit (usda-ars.nmsu.edu/JER/Moni_Assess/PDF_files/SoilAggStabit.pdf). At each collection point, the sample was taken about 15cm from the line. If there was vegetation canopy that covered at least 50% of the sample area, “cover” was recorded. No canopy cover was labeled as “no cover”.

To collect the sample, a small trench 10-15mm deep was dug. If litter was resting over the sample site, it was carefully removed. The top layer of soil was then collected with the sample scoop. The soil aggregate sample was 2-3mm thick and 6-8m in diameter. The sample was placed upright in a drive sieve and put in the appropriate cell of the dry soil stability box. If a sample was unable to be collected because the aggregate was too weak to collect, it was recorded as “1”. If soil was covered by cryptobiotic soil, it was not collected and recorded as “6”, per AIM protocols (Toevs et al., 2011). If a sample occurred at a plant base, the collection was taken within the base, or as close as possible. To test the soil aggregate samples, AIM methods were followed (Toevs et al., 2011).

Soil Nutrients

Every 10m along each transect, a soil core 7.62cm diameter x 20cm deep was collected. Samples were taken 20cm from the transect line to account for soil stability testing. Soil along each transect was combined, air-dried, mixed thoroughly, and sifted through a 2X2 mm sieve. Part of the soil was used to determine bacterial and fungal diversity and the remainder sent to A&L Agricultural Laboratories for nutrient analysis, including percent organic matter (OM), pH, estimated nitrogen reserve (ENR), cation exchange capacity (CEC), phosphorus, nitrate-N, and both parts per million (ppm) and percent of base saturation estimates for potassium (K), magnesium (Mg) and Calcium (Ca).

Ecological Site Descriptions

Each location was matched to its respective ecological site description using SoilMaps, WebSoil Surveys, and ArcMap layers provided by Capitol Reef biologists (https://casoilresource.lawr.ucdavis.edu/gmap/; https://websoilsurvey.sc.egov.usda.gov/App/WebSoilSurvey.aspx; Table 4). Ecological sites descriptions were obtained from the Natural Resources Conservation Service database (https://edit.jornada.nmsu.edu/catalogs/esd) and compared to our observed data.

Ecological sites were not confirmed in the field, as soil pits were not dug due to permit limitations.

Data Analyses

All analyses were run under two separate datasets. To address the first hypothesis, we used data from all site locations from the seven time points (i.e., 1984-1989 and 2020). We partitioned this dataset into two subsets. The first subset included the 1984 through 1989 data for all sites to examine trends through a continuous time frame. The second subset included all years sampled (i.e., 1984-1989, 2020); however, Muley Twist and Red Slide sites were excluded because in both locations the exclosure fence was missing in 2020. They were intact until at least 1989, however, the year the fences were removed is unknown. The second dataset, which helped to address the second and third hypotheses included only the 2020 data from all sites.

Within each treatment (inside or outside) at each location, the average percent cover of each variable (e.g., cover of individual plant species, cover of bare ground) was calculated. The primary data matrix included the average percent cover of each plant species within each treatment (two levels) for each site (seven total) across six years of historical data and one year of present data. The secondary matrix included environmental data. The nine variables were cover of bare ground, cover of cryptobiotic soil, cover of dung, site number, treatment, year collected, drought index value for that year, elevation, and whether the site location was in the northern or southern area of the park.

The original vegetation matrix data had 86 species and 84 plots (i.e. seven sites matched inside and outside exclosures over six years). We evaluated if deleting any rare species, species that only occur in one plot (i.e., twelve species), would reduce variability among sample units (measured as reductions in beta diversity) or dispersion around the mean (measured as reductions in the coefficient of variation or CV). Since these 12 species comprise 14% of the dataset and their removal did not notably reducing the CV or beta diversity, keeping the original matrix was ideal. Outlier test for site location was not conducted because we wanted to retain all sites for an even treatment comparison for each location.

Our plots were grouped by both time (i.e., year data was collected) and by treatment (i.e., inside versus outside the exclosure). To determine whether these groups were defined better by grouping variables than random chance within the data, we used Multi-response Permutation Procedures (MRPP; Mielke, 1984). We used multivariate cover data (all 86 plant species) and univariate cover data (cryptobiotic soil, bare ground, and dung) for these analyses.

To evaluate relationships among plots in species space, a Non-Metric Multidimensional Scaling (NMS; McCune and Grace, 2002) was used. NMS was selected because our data did not follow linear, parametric assumptions and zero values were common within our dataset. This is common within community data and an NMS provides a statistical method able to handle these absences (McCune and Grace, 2002). We used autopilot mode with the “Slow and Thorough” setting, using Sørenson distance measure as it is also preferable for community data. Multivariate analyses were performed using PC-ORD Version 7.08 (McCune and Mefford, 2018).

Using the full vegetation matrix, Shannon’s and Simpson’s diversity indices were calculated for each treatment, blocked by site for each year collected. We then compared diversity estimates inside versus outside the exclosures using a Linear-Mixed Model (LMM), blocked by site, in SAS on Demand (SAS Institute Inc., 2014). LMM was chosen because the data was not continuously gathered for every successional year. A LMM was also used to calculate vegetation species evenness and richness averages for each treatment in every location using the same procedures above.

To determine if cryptobiotic soil cover was different inside versus outside the exclosures, a Linear-mixed Model, blocked by site was used. Darkness and morphology were analyzed using a blocked by site Chi-Square Test using SAS on Demand (SAS Institute Inc., 2014).

Using the soil aggregate stability ratings, an ANOVA, blocked by site was run using SAS on Demand (SAS Institute Inc., 2014). Soil nutrients were examined using Principle Component Analysis ordination (PCA; Jolliffe and Cadima, 2016) within PC-ORD Version 7.08 (McCune and Mefford, 2018). To determine whether soil nutrients were different inside versus outside treatments, a Permutational Multivariate Analysis of Variance (PERMANOVA) was performed using Euclidian distances.

Results

The subsequent section addresses our first hypothesis, using the full dataset across all years.

Vegetation

For data between 1984 and 1989, Shannon and Simpson’s diversity indices both showed treatment effects ($p= 0.010$; $p = 0.027$, respectively; Table 5), but no time main effect or treatment X time interaction (Table 5). Inside versus outside treatments showed opposite trends to each other. With Muley Twist and Red Slide excluded and including the 2020 data, time and treatment separately had a strong effect on diversity (Table 5; Figure 3b, 3d), with diversity decreasing in both treatments during 2020. In both diversity indices, inside treatment ended with a greater diversity than outside.

Table 5: Shannon and Simpson's diversity LMM results for 1984-1989 and 1984-2020 at exclosure sites in Capitol Reef National Park, UT. In 1984-2020 analysis, Muley Twist and Red Slide sites were excluded.

Shannon's Diversity Index 1984-1989				
Effect	*Df*	*Den Df*	*F Value*	*Pr > F*
Treatment	1	6	13.36	0.0106
Time	5	22	1.2	0.3402
Interaction	5	22	0.96	0.4606

Shannon's Diversity Index 1984-2020				
Effect	*Df*	*Den Df*	*F Value*	*Pr > F*
Treatment	1	4	6.1	0.069
Time	6	18	5.62	0.0019
Interaction	6	18	1.64	0.1946

Simpson's Diversity Index 1984-1989				
Effect	*Df*	*Den Df*	*F-value*	*P-value*
Treatment	1	6	8.51	0.0267
Time	5	22	1.78	0.1597
Interaction	5	22	0.66	0.6559

Simpson's Diversity Index 1984-2020				
Effect	*Df*	*Den Df*	*F-value*	*P-value*
Treatment	1	4	6.01	0.0704
Time	6	18	5.09	0.0032
Interaction	6	18	1.01	0.448

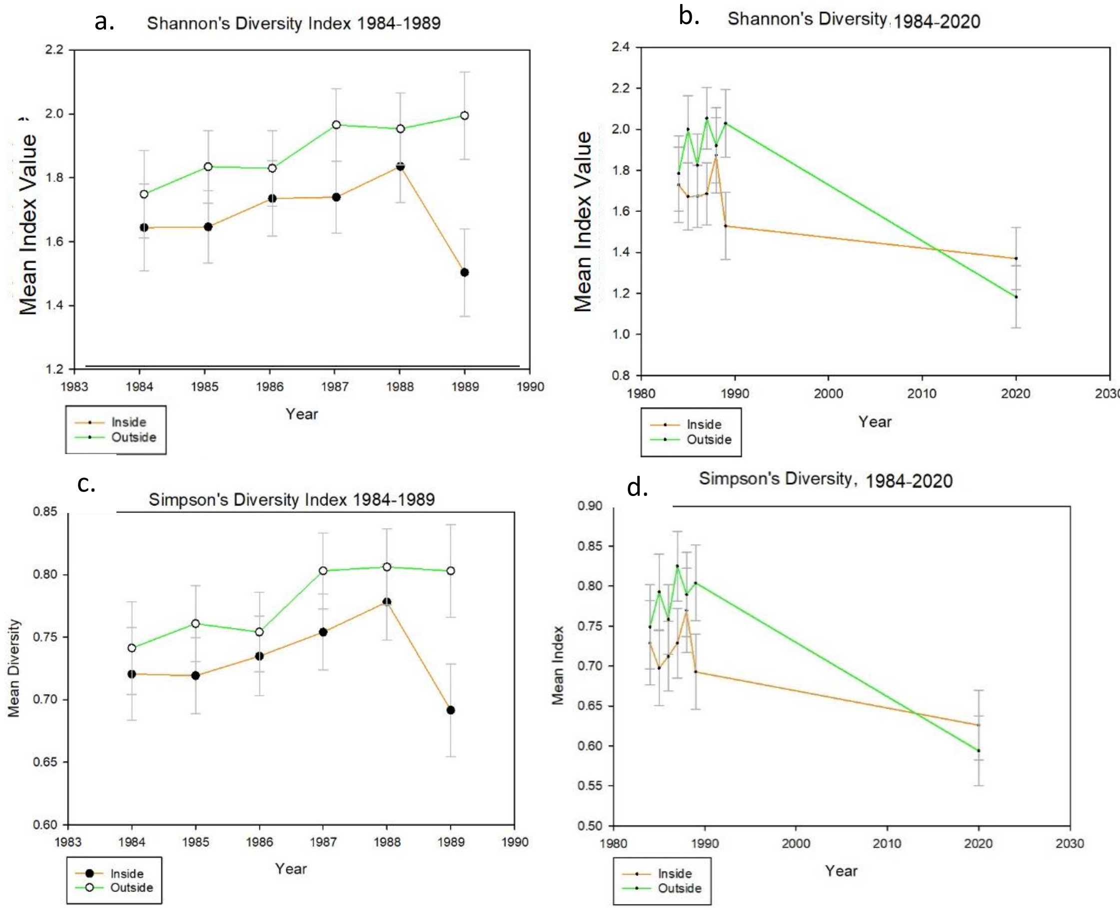

Figure 3: Shannon and Simpson's Diversity Indices for (a, c) 1984-1989 and (b, d) without Muley Twist and Red Slide sites but including 2020 data.

Vegetation species evenness from 1984-1989 only showed a treatment effect (p = 0.019; Table 6). With Muley Twist and Red Slide excluded yet including all years, plots inside the exclosure still had significantly less evenness than the outside (p = 0.037; Table 6). Time main effects and the treatment X time interaction were not significant in either data subset.

Vegetation species richness from 1984-1989 showed a slight significance (p = 0.049; Table 6) inside versus outside. With no Muley Twist or Red Slide sites but including 2020 data, treatment was no longer significant (p = 0.372; Table 6), but time was a significant factor (Figure 4b, 4d; p = 0.041), with richness at their lowest observed values by 2020.

Table 6: Vegetation species evenness and richness for exclosures between 1984-1989 and 1984-2020 at Capitol Reef National Park, UT. In 1984-2020 analyses, Muley Twist and Red Slide exclosures were excluded.

Evenness 1984-1989

Effect	*Df*	*Den Df*	*F Value*	*Pr > F*
Treatment	1	6	10.050	0.019
Time	5	22	0.810	0.555
Interaction	5	22	1.000	0.443

Evenness 1984-2020

Effect	*Df*	*Den Df*	*F-value*	*P-value*
Treatment	1	4	9.54	0.037
Time	6	18	1.8	0.155
Interaction	6	18	0.56	0.759

Richness 1984-1989

Effect	*Df*	*Den Df*	*F-value*	*P-value*
Treatment	1	6	6.060	0.049
Time	5	22	1.810	0.153
Interaction	5	22	0.390	0.853

Richness 1984-2020

Effect	*Df*	*Den Df*	*F-value*	*P-value*
Treatment	1	4	1.010	0.372
Time	6	18	2.810	0.041
Interaction	6	18	0.680	0.664

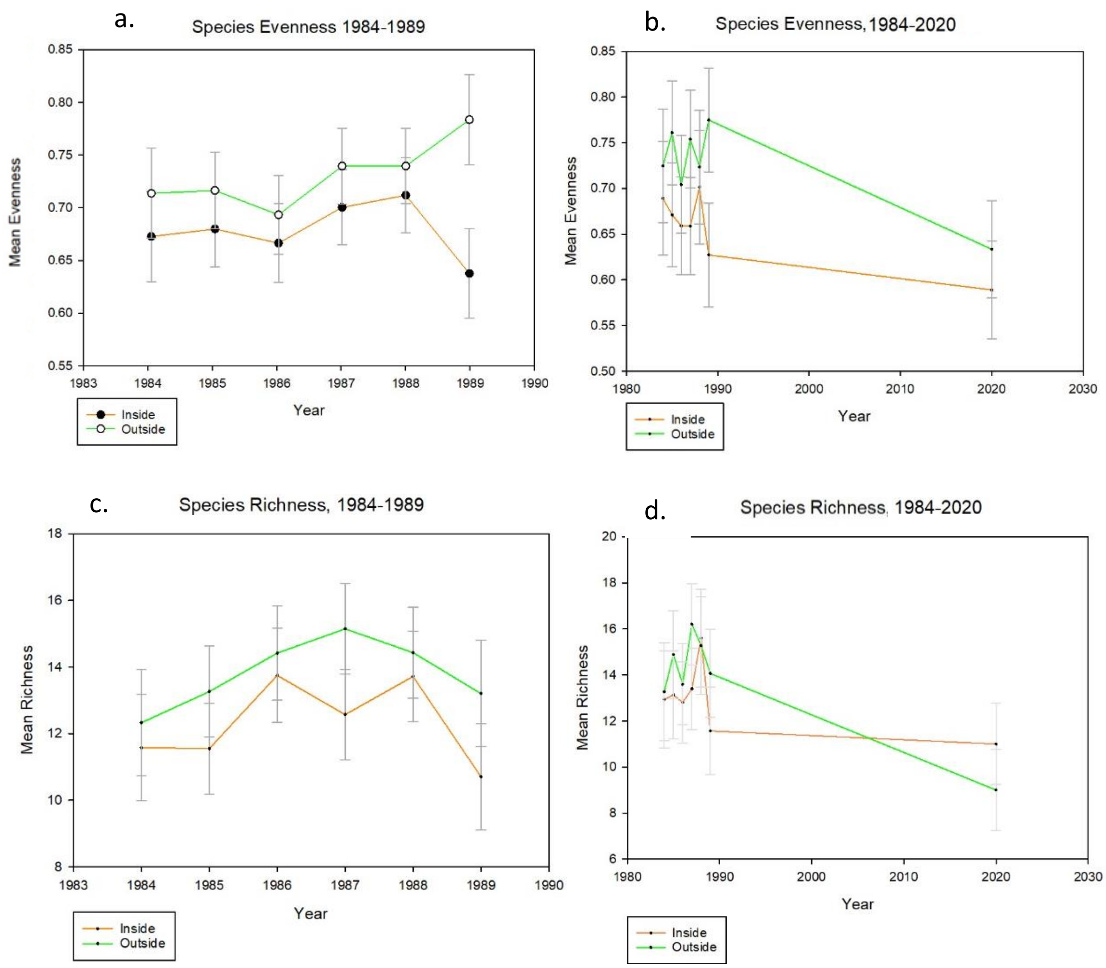

Figure 4: Species evenness and richness for (a, c) exclosures between 1984-1989 and (b, d) 1984- 2020 without Muley Twist and Red Slide.

Community Composition

When pooled across time between 1984 and 1989, there was no significant difference in vegetative community structure inside versus outside of the exclosure (MRPP; A = 0.004, p = 0.177). However, when grouped by time and pooled across treatment, differences were observed (A = 0.229, p = 0.027). We found a similar pattern when Muley Twist and Red slide sites were excluded and 2020 was included (comparing treatment: A = 0.0005, p = 0.374; comparing time: A = 0.052, p = 0.0007).

Our NMS recommended a 3D solution explaining 81.45% of the variance in community composition. The final stress was 13.66. The final solution instability was <0.0001 with 57 iterations.

The first axis explained 35.7% of the variance. The positive end of the axis corresponded with higher *Hilaria jamesii* (r = 0.740) and to a lesser extent *Bouteloua gracilis* (r = 0.395) and *Bromus tectorum* (r = 0.386). Using a cutoff of r = +/- 0.4, cryptobiotic soil cover was weakly corresponded with the positive axis (r = 0.442). The negative end of axis one correlated with the species *Achnatherum hymenoides* (r = -0.688) and to a lesser extent, *Atriplex confirtifolia* (r = -0.479). Axis one was not strongly correlated with any environmental variables (bare ground: r = -0.100). The 84 plots formed distinct groups corresponding to the cluster of sites in ordination space (Figure 5a). Along axis 1, three groups were observed. Hartnet and Lower south Desert on the negative end, Cathedral, Post, and Red Slide in the middle, and Surprise and Muley Twist on the positive end of the axis.

The second axis explained 29.3% of the variance. The one species that corresponded with the positive end of this axis, albeit weakly, was *Chaenactis stevioides* (r = 0.386). The strongest negative correlation of species on axis two were *Gutierrezia sarothrae* (r = -0.692) and at a lesser extent *Sporobolus cryptandrus* (r = -0.493). The environmental variables that corresponded most with axis two were cryptobiotic soil (r = 0.670) at the positive end and bare ground (r = -0.509) at the negative end. Along axis two, sites on the positive end are all latter years of data collection, 1989 and 2020, containing higher cryptobiotic soil. Sites on the negative end represent early years.

The third axis explained 21.8% of the variance. The positive end of axis three correlated with *Sporobolus cryptandrus* (r = 0.548) and *Bromus tectorum* (r = 0.524). The negative end of axis three, although not as strong as the other gradients, correlated most with *Sphaeralcea coccinea* (r = -0.396) and *Hymenopappus filifolius* (r = -0.358).

Successional vectors were drawn to connect plots from the same site by treatment in order of ascending years (Figure 5b-h). By 2020, all sites were moving in a positive direction along axis two, except for Cathedral.

When plots were coded by drought severity, axis two resulted in two groups (Figure 6a). The middle to positive end of axis two contained were sites associated with “severe drought”, which also corresponded with all 1989 and 2020 sites. The remaining earlier years (1984-1988) and drought severity levels were mixed on the negative area of the ordination.

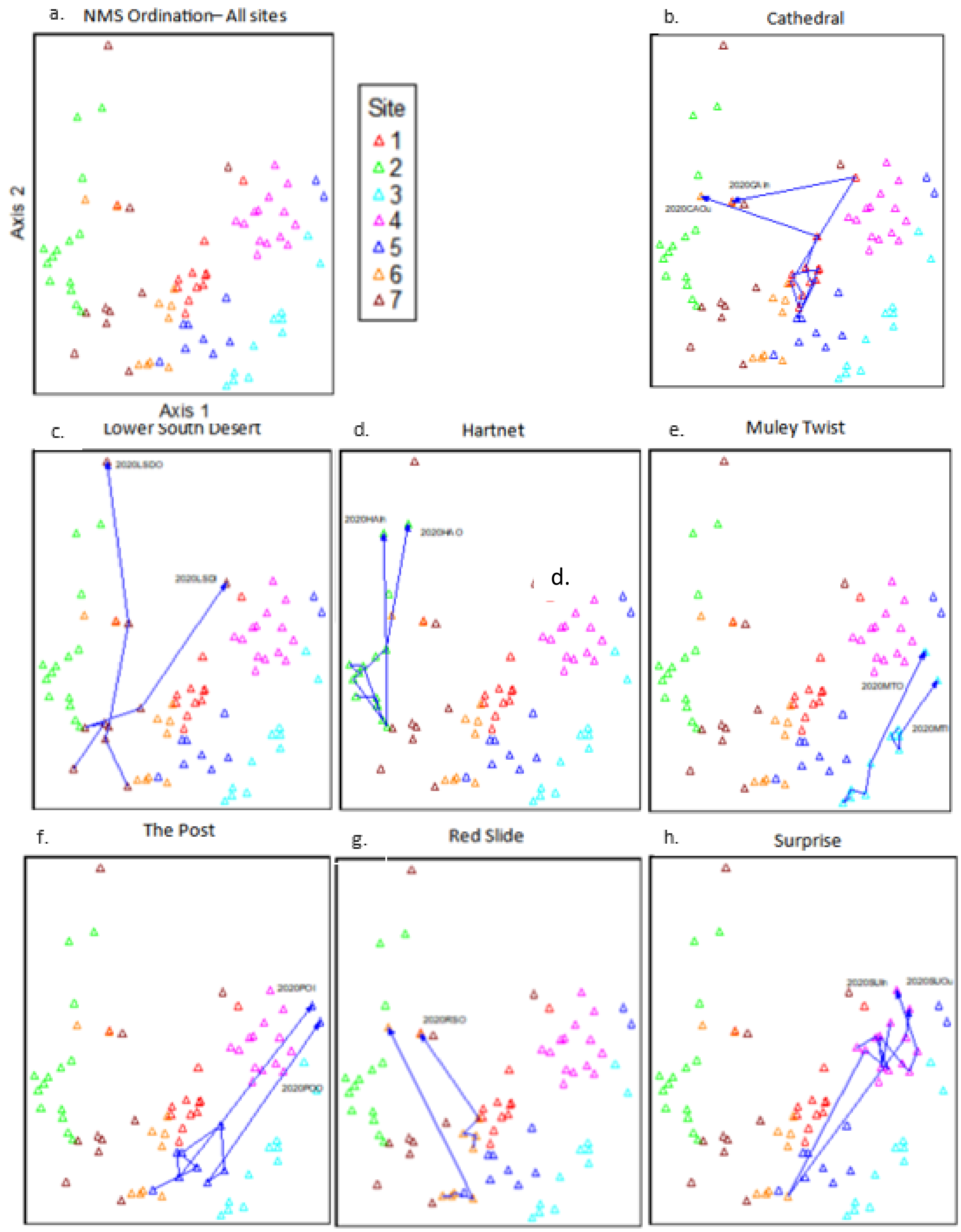

Figure 5: NMS solution of 84 plots, while NMS recommended a 3-D solution only two axes are shown, axis 1 and axis 2. a: NMS ordination of all locations, grouped by site, Capitol Reef National Park, UT. b-h: Successional vectors were drawn to show community response of plots over time for each location; Cathedral, Hartnet and Lower South Desert, Muley Twist, Surprise, Post, Red Slide and Lower South Desert. 1=Cathedral, 2= Hartnet, 3= Muley Twist, 4= Surprise, 5= Post, 6= Red Slide, 7= Lower South Desert.

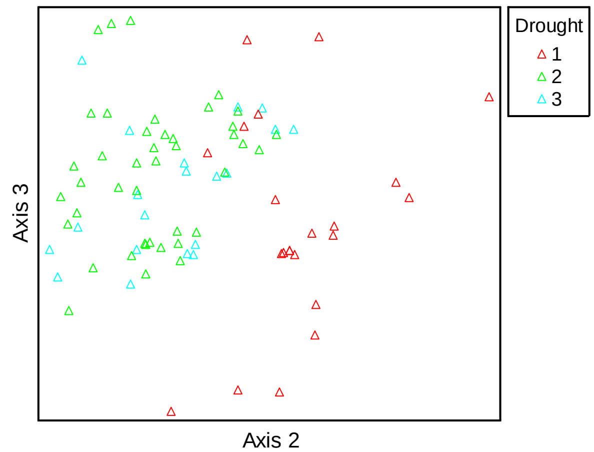

Figure 6: NMS ordination along axis 2 versus axis 3, grouped by drought conditions in Capitol Reef National Park, UT. Numbers were given to drought conditions given by the Palmer Severity Index for the site sample year. Drought conditions were represented by 1= Severe drought (-3cm to -3.99cm), 2 = Mid-range (-1.99cm to +1.99cm), 3 = moderate moist (+2cm to +2.99cm).

Cryptobiotic Soil: Cover

Cryptobiotic cover values for the early, continuous years, 1984-1989 did not show a strong treatment effect ($p = 0.1014$; Table 7) but had a significant time effect ($p = 0.0452$) and time X treatment interaction ($p = 0.0318$). Observing the interaction effect through time until 1989 (Figure 7), the first years of the exclosures, areas accessible to grazing started off with greater cryptobiotic soil than inside the exclosure. Inside diverges from outside over time; however, both showed increases in cover starting in 1987.

Using the same analysis, including the 2020 data but without Muley Twist and Red Slide, inside versus outside did not show a treatment effect ($p = 0.7739$) or a treatment X time interaction ($p = 0.9247$). However, cryptobiotic cover did change significantly over time ($p < 0.0001$). From 1984-1989, inside versus outside responded the same as the previous analysis, despite the deletion of Muley Twist and Red Slide sites. By 2020, cryptobiotic soil cover inside and outside converged to about the same levels, much higher than the earlier years.

Table 7: Linear-Mixed Model tables of cryptobiotic soil crust cover in exclosures at Capitol Reef National Park, UT. Data over the complete time scale, Muley Twist and Red Slide were not included in the analysis.

Cryptobiotic Cover 1984-1989				
Effect	*Df*	*Den Df*	*F Value*	*Pr > F*
Treatment	1	6	3.74	0.1014
Time	5	23	2.72	0.0452
Interaction	5	22	3.02	0.0318

Cryptobiotic Cover 1984-2020				
Effect	*Df*	*Den Df*	*F-value*	*P-value*
Treatment	1	2	0.09	0.7739
Time	6	20	122.23	<0.0001
Interaction	6	19	0.27	0.9427

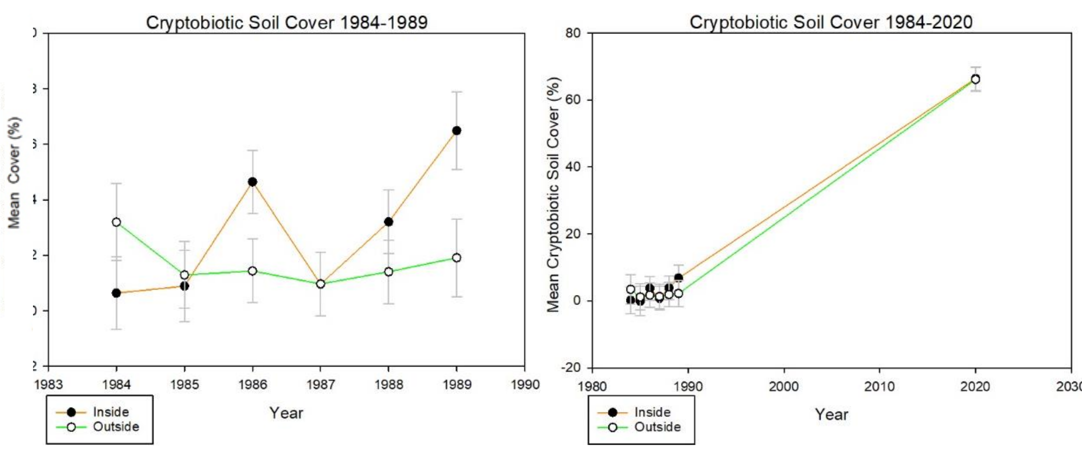

Figure 7: Cryptobiotic soil cover between 1984-1989, inside versus outside and cryptobiotic soil cover over the complete time scale, excluding Muley Twist and Red Slide at Capitol Reef National Park, UT

Soil: Bare Ground

Between 1984-1989, bare ground inside versus outside the exclosure was not significantly different (p = 0.0828; Table 11), although inside was consistently lower than outside the exclosures (Figure 11). The time X treatment interaction effect was also not significantly different (p = 0.4995). The time main effect trended towards significance with bare ground (p = 0.0599).

Using the same question but including the 2020 data and excluding Muley Twist and Red Slide sites, inside versus outside similar results were observed (Table 8). Treatment of inside versus outside was not significantly different (p = 0.0788) nor was the time*treatment interaction effect (p = 0.6561). Time, however, was a significant factor (p = <0.0001), with bare ground decreasing sharply by 2020 (Figure 8).

Table 8: Linear-Mixed Model tables of bare ground cover in exclosures at Capitol Reef National Park, UT. In data including 2020, Muley Twist and Red Slide were not included in the analysis.

Bare Ground Cover 1984-1989				
Effect	*Num DF*	*Den DF*	*F-value*	*p-value*
Treatment	1	6	4.33	0.0828
Time	5	23	2.50	0.0599
Time*Treatment	5	22	0.90	0.4995

Bare Ground Cover, 1984-2020				
Effect	*Num DF*	*Den DF*	*F-value*	*p-value*
Treatment	1	4	5.51	0.0788
Time	6	20	75.25	<.0001
Time*Treatment	6	19	0.70	0.6561

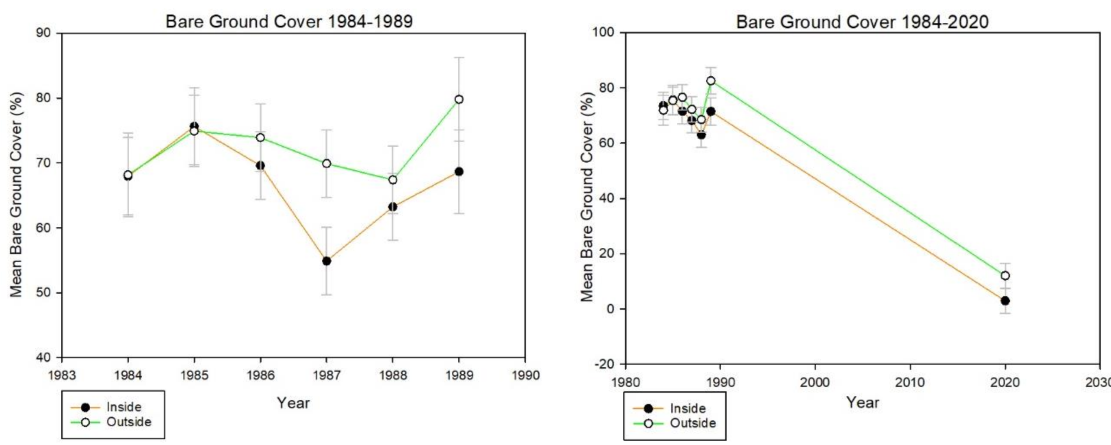

Figure 8: Bare ground cover between 1984-1989, inside verse outside and cryptobiotic soil cover over time, excluding Muley Twist and Red Slide at Capitol Reef National Park, UT.

The subsequent section addresses our second hypothesis, using data only obtained in 2020.

Cryptobiotic Soil: Darkness

Cryptobiotic soil darkness was not significantly different inside compared to outside treatments in 2020 (p = 0.8555; Table 9); however, a trend of slightly less developed cryptobiotic soil was observed outside exclosures compared to inside (Figure 8).

Table 9: ANOVA table for cryptobiotic soil darkness in 2020 at Capitol Reef National Park, UT grazing exclosures. Muley Twist and Red Slide are excluded from the analysis.

Cryptobiotic Darkness 2020					
Source	*Df*	*SS*	*MS*	*F-value*	*P-value*
Model	1	0.008	0.008	0.040	0.8555
Error	8	1.826	0.228		
Total	9	1.834			

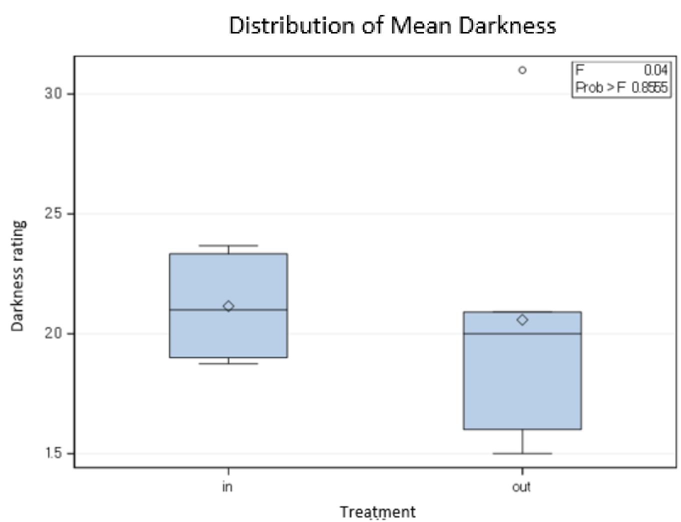

Figure 9: 2020 cryptobiotic soil darkness ratings inside and outside the exclosures, Capitol Reef National Park, UT. Ratings are on a color scale of 1-6 with a rating of 6 as the darkest and most developed soil.

Soil: Stability

In 2020, we found a treatment effect inside versus outside the exclosures (p = 0.0014; Table 10), with more stable soils corresponding with inside the exclosures (Figure 10). Whether the soil aggregate sample was taken under the cover of vegetation or without cover, did not affect soil stability (p = 0.5487), nor did the cover interaction effect (p = 0.7301).

Table 10: ANOVA results for soil stability collected in 2020, Capitol Reef National Park, UT. Muley Twist and Red Slide sites were excluded for this analysis. Soil aggregate stability was rated on a scale 1-6 with 1 being least stable and 6 representing the most stable and/or cryptobiotic soil.

Soil Stability Results 2020					
Source	*Df*	*SS*	*MS*	*F-value*	*P-value*
Treatment	1	5.9905	5.9905	14.8700	0.0014
Cover	1	0.1513	0.1513	0.3800	0.5487
Trt*Cover	1	0.0497	0.0497	0.1200	0.7301

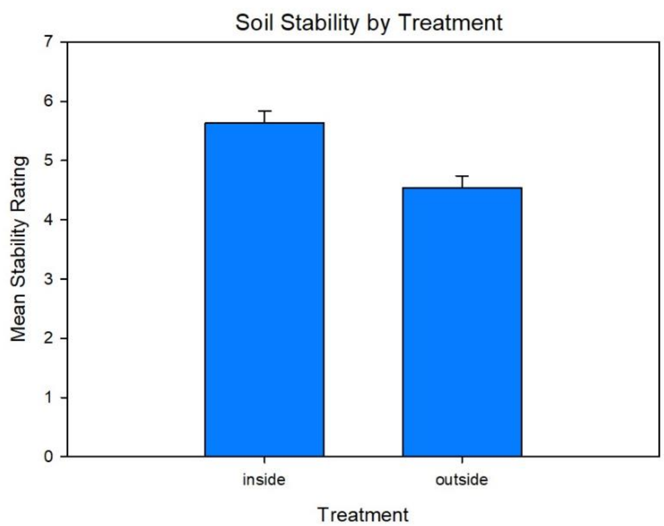

Figure 10: Mean soil stability ratings in 2020 inside versus outside within Capitol Reef National Park, UT. Muley Twist and Red Slide were not included in this analysis.

Soil: Nutrients

Soil nutrients, including organic matter (OM), estimated nitrogen release (ENR), phosphorous (P), sodium phosphate (NaP), potassium (K), magnesium (Mg), calcium (Ca) and nitrate, along with pH and cation-exchange capacity (CEC), did not differ between inside or outside the exclosures in the PCA ($F = 0.42336$, $p = 0.67522$). There was also no pattern present in soil nutrients among site locations present in the ordination. All locations, both inside and outside exclosures, were characterized by typical basic soils with low overall nutrients.

Ecological Sites

Using soil stability data and cover percentages for cryptobiotic soil, bare ground, tree, shrub, grass, forb, and invasive species in 2020 only, each treatment was compared to its assigned Ecological Site Description Reference Site (USDA-NRCS, 2006; Table 11). For soil stability, both under vegetation cover and under no cover, inside sites were all within or above their reference state. Outside sites contained two locations under vegetation cover that were below their reference site: Lower South Desert and Muley Twist. Under no vegetation cover, only Muley Twist was below its reference site. All locations under both treatments had cryptobiotic soil cover within or above their given reference value. Only one site had greater than expected bare ground cover in both treatments, Muley Twist. All treatments were within range for tree cover, except for Muley Twist in both treatments. Shrub cover each had two treatments within reference range. Cathedral and Hartnet, both inside and outside were within the lower range of their ecological site. Grass cover in both treatments only had one site within range, Lower South Desert and one above the reference state range, Muley Twist. All other sites were below their reference state. Forb cover was below in two sites for each treatment, Lower South Desert and Red Slide. For cover also had one site within each treatment above reference range, Post in both inside and outside. All other treatments were within range. All sites in all treatments contained at least a trace of invasive plant species.

Table 11: Comparisons of each site and treatment to its given Ecological Reference State in Capitol Reef National Park, UT. Trt = treatment (inside or outside), Stability: Cover = soil aggregate stability rating under vegetation cover, Stability: NC = soil aggregate stability rating with no vegetation cover, Crypto % = cryptobiotic soil percent cover, BG % = bare ground percent cover, Tree = tree percent cover, Shrub = shrub percent cover, Grass % = grass percent cover, Invasive % = invasive species percent cover, O = observed value, ES = ecological site reference state value. Color coding represents whether the observed value was above (green), within (yellow), or below (red) its reference state value.

		Stability:Cover		Stability:NC		Crypto %		BG %		Tree %		Shrub %		Grass %		Forb %		Invasive %
Site Name	Trt	O	ES	O	ES	O	ES	O	ES	O	ES	O	ES	O	ES	O	ES	O
Cathedral	In	6	4-5	6	4	82	0-10	10	40-65	0	0	2	0-6	5	18-28	1	0-6	0.03
Hartnet	In	6	4-5	5	4	86	0-10	8	40-65	0	0	2	0-6	3	18-28	1	0-6	0.20
LSD	In	5	5	5	2-3	27	0-60	6	30-60	0	0	1	20-60	17	5-20	1	5-10	0.01
Surprise	In	6	4-5	5	3-4	76	0-2	12	10-50	0	0-3	2	5-15	7	10-25	3	0-5	0.70
Muley Twist	In	4	4-5	5	4	0	0-52	71	16-30	0	6-16	1	6-16	22	0-5	0	0-2	4.10
The Post	In	5	4-5	6	3-4	61	0-2	10	10-50	0	0-3	0	5-15	3	10-25	13	0-5	3.10
Red Slide	In	5	4-5	5	3-4	68	5-15	9	20-30	0	0	2	10-25	5	15-30	1	2-10	0.03
Red Slide	Out	6	4-5	4	3-4	39	5-15	27	20-30	0	0	2	10-25	7	15-30	1	2-10	0.15
Cathedral	Out	4	4-5	4	4	58	0-10	32	40-65	0	0	2	0-6	7	18-28	1	0-6	0.15
Hartnet	Out	5	4-5	6	4	76	0-10	11	40-65	0	0	4	0-6	1	18-28	3	0-6	0.30
LSD	Out	3	5	4	2-3	58	0-60	23	30-60	0	0	0	20-60	17	5-20	0	5-10	0.04
Surprise	Out	4	4-5	5	3-4	71	0-2	19	10-50	0	0-3	1	5-15	7	10-25	1	0-5	1.53
Muley Twist	Out	2	4-5	2	4	2	0-52	72	16-30	0	6-16	4	6-16	21	0-5	0	0-2	3.60
The Post	Out	5	4-5	6	3-4	68	0-2	11	10-50	0	0-3	0	5-15	6	10-25	6	0-5	1.20

Discussion

Inside Versus Outside Over Time

There were a few observable differences when comparing inside versus outside the exclosures, which suggests that grazing does have some impact on the landscape, even after long-term rest from grazing. However, most treatment differences occurred over the earlier years when grazing was active at all locations. By 2020, high cover of cryptobiotic soil is evident in many locations; however, climate seems to have an overarching effect on the land that makes our other measured factors (percent ground cover, vegetation, soil stability, cryptobiotic soil attributes) respond similar to their corresponding treatment. Time had a greater impact on soil and vegetation differences than grazing treatment. For example, with time, cryptobiotic soil cover significantly increased while bare ground decreased. There are also clearly two main time periods in terms of community composition (i.e., 1989 and 2020 versus pre-1989), as shown by axis two in the NMS ordination. Lack of significance between treatments in richness, cryptobiotic soil cover, and bare ground cover over the entire dataset, 1984-2020, may be due to grazing histories.

Our first hypothesis suggested that through time, inside treatments will have higher species diversity, differing community structures, higher cryptobiotic soil cover and lower bare ground cover than outside. These differences were more apparent when grazing was active in all locations rather than when 2020 data was included (Muley Twist and Red slide were not included due to removal of exclosure fence), where many sites were rested from grazing for some time. Grazing intensity decreased over time, so while recovery may be happing, management of livestock may also have lessened any effects over the entire timespan.

Between 1984 and 1989, we observed differences in treatment within species diversity (Shannon's and Simpson's) evenness and richness, however outside had overall higher species richness than inside and both treatments decreasing starting in 1986. During this period, there was no treatment effect in cryptobiotic soil cover and bare ground cover. These two covers behaved opposite to each other and inside remained higher in cryptobiotic soil and lower bare ground cover as time pass. Species composition also did not show a significant treatment effect. Time, however, is a significant factor between 1984 and 1989, where cryptobiotic soil, bare ground, and species composition did show an effect.

Between 1984-1989 and the addition of 2020 data (without Muley Twist and Red Slide), only a treatment effect in species evenness remained. However, time reflected different results. Diversity (Shannon and Simpson's), richness, species composition, cryptobiotic soil, and bare soil all shown significant changes. Evenness and diversity decreased in all treatments over time. Although diversity was not significant between treatments, by 2020 diversity inside was higher than outside. There was a large increase in cryptobiotic soil cover between historical years and 2020 in both treatments. This large difference in percent coverage may be due to knowledge and categorization of cryptobiotic soil and bare ground between recent versus early years. It is possible that historically, cryptobiotic cover may not have been noted in its earlier stages (ratings 1-2) and may have been recorded as bare ground. However, as Young (et al., 2016) suggests, cryptobiotic soil's biggest threat is physical disturbance. This and the lack of overall treatment effects may be due to decreased grazing AUM's and intensities over the various areas of the park. By the 1990's most allotments, besides Hartnet and Sandy 3, were rested from grazing (NPS, 2018). Therefore, Cathedral, Post, and Red Slide have had both treatments absent of livestock grazing for about 30 years.

Community Patterns

To better visualize these interactions between species diversity, composition, cryptobiotic soil cover and bare ground cover over treatment and time, the following section interprets the axes and community composition of our plots.

Ordination axis one primarily represents a geography and grazing history gradient in vegetation community structure. On the positive end, the plants were *Hilaria jamesii* and *Bouteloua gracilis*. *Hilaria jamesii* and *Bouteloua gracilis* are characterized as warm season grasses (Hewins et al., 2015; Massatti and Knowles, 2020). The negative end is dominated by *Achnatherum hymenoides*, a cool season grass (Hewins et al., 2015). In

Capitol Reef National Park, grazing primarily occurs in late winter or early spring and, on average, concludes in May (NPS, 2008). Warm season grasses such as *Hilaria jamesii* tend to not be as impacted by winter grazing, as they generally emerge in the summer season (Humphrey and Schupp, 1999) after grazing has already been removed for the season. The positive end represents plots that are more likely grazed and occupied by plants that are adapted to avoid or tolerate grazing.

Along axis one, our plots are broken up into roughly three groupings based on their geography and grazing histories within the park. On the negative end of the axis are Hartnet and Lower South Desert exclosures. These exclosure are in the northern section of the park and have been rested from grazing since 2018. In the middle of the axis are Cathedral, Red Slide and Post. Although these are in both the northern and southern sections of the park, they have all been rested from grazing for about 30 years. The positive axis, corresponding with the most warm-season grasses, are Surprise and Muley Twist. These exclosures are both located in the southern areas of the park.

The second axis corresponds to both a drought and time gradient. These two factors are likely conflated, and we were unable to distinguish which has a greater influence on community structure. Species composition in all sites was distinctly different along this axis, with plots from 1989 and 2020 occurring at the positive end. These years correspond with “severe drought” (Figure 8a, 8b). In all other years sampled, Palmer drought severity was “mid-range” or “moderately moist” and were not configured in any pattern. Verwijmeren et al. (2014) found that aspect has a bigger impact on vegetation cover than grazing, with a decline of 17.2% from north to south facing slopes.

As south facing slopes are drier and warmer (Bennie et al., 2006, Gong et al., 2008), this may be analogous to the response from drier conditions we observe with drought.

Axis two also represents a gradient in ground cover among the sites, bare ground cover at the negative end and cryptobiotic soil cover on the positive. These ground cover attributes also correlate with time and drought as 1989 and 2020 represented the years with the greatest cryptobiotic soil cover overall (Figure 9). These trends show that cryptobiotic soil, regardless of treatment, increased over time. As all but two sites have had rest from grazing for at least three years, this corroborates Miller et al. (2017) and Warren et al. (2019), where cryptobiotic soil regenerated between 1989-2020.

The third axis represents a species abundance gradient. At the negative end of the gradient, plots associated with the lowest species abundance were observed. The sites associated with the least abundance of species were both in 1989, Lower South Desert, both inside and outside and Cathedral outside. In 1989, grazing was active in the allotments where these exclosures occurred, both located in the northern region of the park. This finding may suggest heavy grazing in this area of the allotment during this year.

Hartnet and Lower South Desert are both characterized by communities higher in *Achnatherum hymenoides, Sporobolus cryptandrus,* and *Atriplex confirtifolia.* Over time, their compositions responded similarly with greater bare ground in the earlier years and gradually were characterized by higher cryptobiotic soil over time. Their species abundance remained similar through time; however, Lower South Desert overall had lower abundance of the same species that occurred in the Hartnet plots. These northern localities that have been rested from winter grazing since 2018 share more similar compositions than others, despite not sharing the same ecological site. The NMS ordination shows Hartnet and Lower South Desert share more similarities in composition than Hartnet and Cathedral do, which share the same ecological site (Desert Alkali Sandy Loam; USDA-NRCS, 2006).

The southern localities, Surprise and Muley Twist, both currently grazed, shared similar compositions. They are dominated by warm-season grasses, *Hilaria jamesii, and Bouteloua gracilis* and the invasive, noxious weed *Bromus tectorum.* These localities have similar abundances and behave similarly over time/drought, increasing in cryptobiotic soil cover. These two sites share the same ecological site (Semidesert Gravelly Loam (Shadscale)), grazing history and overall composition. This similarity in composition also supports that composition and responses of rangeland may respond similar under similar grazing histories (Belote et. al., 2009; Condon et., al., 2018).

Cathedral, Red Slide, and Post localities share a mix of *Achnatherum hymenoides* and *Hilaria jamesii* associated communities. They remain similar in moderate to low cryptobiotic soil crust, species abundance, and similar composition and responses through time. Although these localities are mixed between north and south localities and ecological sites, these locations have all been excluded from grazing in the late 1980's or early 1990's, further supporting grazing histories influence on composition over time and response to climate drivers, such as droughts.

Interestingly, shown in our data and in the ordination, drought seems to be a bigger overall influence over, richness, species diversity, and composition, rather than cryptobiotic soil influences. This may explain some of our large changes over time in these factors. Belote et al. (2009) found when observing only rangeland vegetation communities at Surprise, Hartnet, and Cathedral sites (1984-1993), that grazing can alter the way rangeland communities respond to climate by examining shifts in vegetation community. Like this study, we observed changes in vegetation and cover during the drought year of 1989. Belote et al. (2009) results were consistent with Loesser et al. (2007) in Arizona, where they found grazing and climate influenced shifts in community composition. These two studies suggest grazed plots are less resilient to climatic variability and have greater increases in exotic annual species.

Corroborating these studies, we did generally see higher invasive species outside than inside the exclosures with the currently grazed sites having the highest abundance. Although we did not break up vegetation functional groups over time as Belote et al. (2009), we found that by 2020 most plots, despite treatment, decreased in abundance over all functional groups (trees, shrubs, grasses and forbs). This more closely reflected results of Condon et al. (2020) where differences observed in vegetation cover were associated with differences in plant communities and not the presence or absence of grazing. Differences in findings from Belote et al. (2009) and Loesser et al. (2007) may be due to longer records of drought. Belote et al. (2009) study observed vegetation shifts with only one year of drought between a 9-year period while our study spans over 31 years. Most years between 2000-2021 in Wayne County, where Capitol Reef is located, were plagued by drought and the latter years being the most extreme drought conditions. (https://www.drought.gov/states/utah/county/wayne). Our data suggest that consistent drought over a long period affects all plant functional groups, despite grazing history and thus may explain our lack of treatment difference over time.

However, cryptobiotic soil did not follow the same trend as vegetation under long drought conditions. Studies show cryptobiotic soil crust's desiccation tolerance is one of many life history traits that help it to colonize severe environments (Bowker, 2007; Oliver et al., 1993). Our data supports these findings with cryptobiotic soil cover increasing over time, despite environmental conditions, when physical disturbance decreased. In Australia, Reed (et al., 2011) found cryptobiotic soil recovery observed within 20 years. Inverse to cryptobiotic soil cover, bare ground decreased significantly over time showing a pattern of areas inside exclosures consistently being higher in cryptobiotic soil cover and lower bare ground cover. This may be due to less grazing and trampling to create bare ground spots or bare ground being colonized by the cryptobiotic soil. Although not significant, inside versus outside showed an effect fairly quickly in bare ground between the treatments, seeing divergence starting two years after the exclosures were constructed. Increased cryptobiotic cover across all sites and treatments may be seen because the majority the sites have been recovering, both inside and outside, since the 1990's as grazing has been removed over time in the park in various allotments.

As some other studies suggest, cryptobiotic soil is associated with higher plant species diversity and richness (Belnap, 2002; Harper and Belnap, 2001; Rosentreter and Root, 2019; Stohlgren et al., n.d.). While this may still hold true, our data show that cryptobiotic soil and vegetation do not follow the same trends, neither inside nor outside the exclosure with increase in cryptobiotic soil cover and decrease vegetation. Cryptobiotic soil may be more tolerant than vegetation or not able to respond as quickly.

Inside Versus Outside in 2020

Our second hypothesis suggests in 2020, inside exclosures would show higher cryptobiotic soil darkness ratings, differences in morphology, increased soil stability and increased soil nutrients. Our data only partially supported this hypothesis with significant treatment differences only in soil stability.

The darkness and thus amount of cyanobacteria in the cryptobiotic soil was generally greater inside than outside the exclosures. This was shown by darker rated cryptobiotic soil inside, supporting the same pattern of more cover and more developed cryptobiotic soil versus outside, albeit not significantly different between treatments. Greater cyanobacteria within the crust increases the soil's resistance to wind and water erosion by strongly binding together soil particles (Rosentreter et al.,2007).

Morphology followed the trend of greater pinnacled and rolling inside exclosures but was also not significantly different. This is not surprising as soil morphology generally reflects the local climate, not necessarily disturbances (Rosentreter et al., 2007). Although smooth morphology is also corresponded with highly disturbed deserts (Rosentreter et al., 2007). These data may provide a baseline of current conditions and would be an interesting factor to observe if morphological changes occur over time with changing climate.

Although there was no difference in stability under vegetation compared to the canopy interspaces, inside locations had significantly greater soil stability averages than outside in 2020. The biggest soil stability differences between treatments were in Muley Twist (2.2 rating difference), Cathedral (1.8 rating difference) and Lower South Desert (1.3 difference). Interestingly, all three of these sites have different grazing histories;

Cathedral rested from grazing for about 30 years, Lower South Desert about three years, and Muley Twist is currently grazed. However, soil stability overall was unexpectedly high in all treatments, especially under extreme drought conditions. This differs from that of Washington-Allen et al. (2010) who suggested that threats of soil erosion on grazed lands increases in periods of reduced precipitation.

As soil stability is an essential element in landscape stability (Jimenez Aguilar et al., 2009), it is extremely important to understand its relationship with other elements. When performing the soil stability test, cryptobiotic soil is automatically rated as “6”, the highest soil stability rating (Toevs et al., 2011). Sites with the greatest cryptobiotic soil cover had higher overall stability, similar to findings in other studies of cryptobiotic soil crust and soil health (Belnap & Gillette, 1998; Belnap and Lang, 2003; Bowker, 2007; Bowker et al., 2008; Kirdron and Yair, 1997; Mazor et al., 1996). Although there was not a significant difference between treatments in cryptobiotic soil cover, starting the year after exclosures were built, there was a noticeable trend that inside consistently had higher cryptobiotic soil cover. Inverse to cryptobiotic soil, bare ground cover, although not significant, was higher outside than inside, consistently over all years.

Comparison to Ecological Site Descriptions

Our third hypothesis aimed to determine if sites currently have restored enough to resemble their ecological reference state using 2020 data. We hypothesized that inside treatments would show closer resemblance to their ecological reference state than outside treatments. In many individual observed factors, our hypothesis was correct, but no location or treatment was fully within its goal reference state. This is because at every site and every treatment there was presence of invasive species, despite not being dominate in most locations. With the presence of any invasive species, the ecological site downgrades to invasive state or potential state, per each ecological site descriptions (USDA-NRCS, 2006). There is also no site, inside or outside, that is within its ecological reference site for all attributes which means recovery, especially of vegetation, either needs more time or active restoration techniques.

Vegetation in all treatments across the park had cover of many groups of vegetation that were less than their reference ecological state, mostly in shrub and grass cover. Grasses had 5 inside and 5 outside treatments all below reference percent values. Forbs had 4 of each treatment below reference values. For forb cover, besides Post, percent cover was on the lower end of the reference scale. Despite treatment, cryptobiotic soil cover is higher or within its reference state at all site locations. Contrary to cryptobiotic cover, bare ground is lower in two of each treatment than the reference state. The highest bare ground cover, well above its reference state inside and outside was Muley Twist.

Inside treatments had higher ratings of cryptobiotic soil in all inside treatments, except for Lower South Desert, Red Slide and Muley Twist. However, Muley Twist and Red Slide both did not have fences associated with their sites. Soil stability in the interspaces of vegetation cover showed 6 inside plots and 5 outside above references conditions. In sites currently grazed, there are larger differences in inside versus outside in reference site characteristics, although in many cases, both still meet reference site conditions when comparing soil stability and cryptobiotic soil. This may be due to a combination of many sites having the ability, both inside and outside, to passively recover due to removed grazing and similar responses from extreme drought. These comparisons, however, do not give the entire picture. Dead vegetation was not counted in this study and with 2020 being a drought year, there may have been more dead vegetation. Like Belote et. al. (2009) found, exotic annuals, such as *Bromus tectorum*, sharply decreased during drought years. The year 2020 was the most extreme drought conditions in the past six years, which may skew the perception of recovery. For example, Muley Twist, which has had a fence since at least the 1990's, still showed some treatment differences, such as in soil stability. Average inside stability was 4 while outside was rated as a 2. However, despite this difference, cryptobiotic soil was low on the reference site scale, albeit, still within range at 0 and 2 percent, respectively. Bare ground was much higher, but that may to attributed to the amount of dead *Bromus tectorum.* Despite being attached to its stalk, it was not considered litter nor was it accounted for in the in-cover estimates. However, visually, the entire site was dominated with dead stalks.

As for the exclosure location itself, vegetation functional groups and composition inside and outside the exclosure boundaries of Muley Twist did not match the rest of the landscape, which was dominated by a pinyon/juniper community. Instead, it was dominated by *Bromus tectorum*. Here, vegetation structure has completely shifted from its ecological reference state, yet just looking at the percent cover of grass, it looks within reference. It is important to take into consideration what species are dominate in ESD's in years without drought, we may have seen live *Bromus tectorum* which would show a full shift in vegetation community, but this was not evident by the data collected.

The Hartnet exclosure shares a similar story, except with the annual species *Vulpia octoflora.* Despite this species being native, it shows collecting data specifically in a drought year may not be a true representation of the landscape's response to grazing over time as it has shown to drastically change landscape factors within the frame of the drought.

With so many variables in location, soil composition, vegetation, precipitation, and elevation, comparing multiple exclosures under multiple grazing histories and intensities proves difficult. Climate changes may have a greater effect on landscape changes, especially vegetation. With the combination of drought stress and grazing pressure on shaping plant-plant interactions still not fully understood (Verwijmeren et al., 2014), sustainable management of arid and semi-arid regions heavily depends on how land managers understand these ecological processes (Popp et al., 2009). Continued efforts to improve prediction of future trends of both abiotic and biotic factors under climate change are imperative.

With this in mind, we do see recovery over time, but restoration decisions should be taken on a site-by-site basis, as Popp et al. (2009) suggested. Depending on the location and severity of impact, some locations may be able to recover under passive conditions. In locations where vegetation structure severely deviates from its ecological reference site or invasive species are increasing, recovery may not be attainable without intervention.

Continual landscape monitoring is ideal in Capitol Reef to record long term recovery of the rangeland. This is especially important as exclosures can be useful in monitoring rehabilitation, but they are rarely maintained over long time scales (Bowker, 2007). These exclosures can provide a unique opportunity to observe landscape changes and trajectories after the current drought conditions and how those differ from treatment.

In addition, with recovery times of cryptobiotic soil varying greatly, depending on preceding sampling (Belnap et al., 2006; 2008), it would be of immense interest to the scientific community to observe how it responds in the upcoming years.

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CHAPTER III

SUMMARY OF GRAZING AND DROUGHT IMPACT ON RANGELAND AND FUTURE DIRECTIONS

There were a few observable differences when comparing inside versus outside the exclosures, which suggests that grazing does have some impact on the landscape, even after long-term rest from grazing. However, most treatment differences occurred over the earlier years when grazing was active at all locations. By 2020, high cover of cryptobiotic soil is evident in many locations; however, climate seems to have an overarching effect on the land that makes our other measured factors (percent ground cover, vegetation, soil stability, cryptobiotic soil attributes) respond similar to their corresponding treatment. Time had a greater impact on soil and vegetation differences than grazing treatment. For example, with time, cryptobiotic soil cover significantly increased while bare ground decreased. There are also clearly two main time periods in terms of community composition (i.e., 1989 and 2020 versus pre-1989), as shown by axis two in the NMS ordination. Lack of significance between treatments in richness, cryptobiotic soil cover, and bare ground cover over the entire dataset, 1984-2020, may be due to grazing histories.

Hartnet and Lower South Desert are both characterized by communities higher in *Achnatherum hymenoides, Sporobolus cryptandrus,* and *Atriplex confirtifolia.* Over time, their compositions responded similarly with greater bare ground in the earlier years and gradually were characterized by higher cryptobiotic soil over time. Their species abundance remained similar through time; however, Lower South Desert overall had lower abundance of the same species that occurred in the Hartnet plots. These northern localities that have been rested from winter grazing since 2018 share more similar compositions than others, despite not sharing the same ecological site. The NMS ordination shows Hartnet and Lower South Desert share more similarities in composition than Hartnet and Cathedral do, which share the same ecological site (Desert Alkali Sandy Loam; USDA-NRCS).

The southern localities, Surprise and Muley Twist, both currently grazed, shared similar compositions. They are dominated by warm-season grasses, *Hilaria jamesii, and Bouteloua gracilis* and the invasive, noxious weed *Bromus tectorum.* These localities have similar abundances and behave similarly over time/drought, increasing in cryptobiotic soil cover. These two sites share the same ecological site (Semidesert Gravelly Loam (Shadscale), grazing history and overall composition. This similarity in composition also supports that composition and responses of rangeland may respond similar under similar grazing histories (Belote et. al., 2009; Condon et. al., 2018).

Grouped together in the ordination along axis one is also Cathedral, Red Slide, and Post. These localities share a mix of *Achnatherum hymenoides* and *Hilaria jamesii* associated communities. They remain similar in moderate to low cryptobiotic soil crust, species abundance, and similar composition and responses through time. Although these localities are mixed between north and south localities and ecological sites, these locations have all been excluded from grazing in the late 1980’s or early 1990’s, further supporting grazing histories influence on composition over time and response to climate drivers, such as droughts.

Although our data supported differences between both treatments and time, we aimed to determine if sites currently have restored enough to resemble their ecological reference state using 2020 data. We hypothesized that inside treatments would show closer resemblance to their ecological reference state than outside treatments. In many observed factors, our hypothesis was correct, but no location or treatment was fully within its goal reference state. To our surprise, all sites are not in their ecological reference state due to the presence of invasive species, despite not being dominate in most locations. With the presence of any invasive species, the ecological site downgrades to invasive state or potential state, per each ecological site description. There is also no site, inside or outside, that is within its ecological reference site for all attributes which means recovery, especially of vegetation, either needs more time or active restoration techniques.

Vegetation in all treatments across the park, had many cover percentages less than their reference ecological state, mostly in shrub and grass cover. Besides The Post, forb cover was on the lower end of the reference scale. However, further supporting how cryptobiotic soil is responding to current conditions differently. In all treatments, cryptobiotic soil cover is higher or within its reference state. Contrary to cryptobiotic cover, bare ground is lower is most than the reference state.

Inside treatments had higher ratings than outside in many attributes such as soil stability and cryptobiotic cover, with less bare ground cover (Table 11). In sites currently grazed, there are larger differences in inside versus outside in reference site characteristics, although in many cases, both still meet reference site conditions. This may be due to a combination of many sites having the ability, both inside and outside, to passively recover due to removed grazing and similar responses from extreme drought. These comparisons, however, do not give the entire picture. Much more must be taken into consideration. Dead vegetation was not counted in this study and with 2020 being a drought year, there may have been more dead vegetation and annuals that did not germinate. Like Belote et. al. (2009) found, exotic annuals, such as *Bromus tectorum,* sharply decreased during drought years. The year 2020 was the most extreme drought conditions in the past six years, which may skew the perception of recovery.

Belote et al. (2009) found that when observing only rangeland vegetation communities at Surprise, Hartnet, and Cathedral sites (1984-1993) that grazing can alter the way rangeland communities respond to climate. This was consistent with that of Loesser et al. (2007) in Arizona where they found grazing and climate influenced shifts in community composition. These two studies suggest grazed plots are less resilient to climatic variability and have greater increases in exotic annual species. Similar with these studies, over our sites, we did generally see higher invasive species outside than inside the exclosures (Table 11) with the currently grazed sites having the most abundance.

Although, we did not break up vegetation functional groups over time as Belote et al. (2009), we found by 2020 most plots, despite treatment, decreased in abundance over all functional groups. This more closely reflected Condon et al. (2020) results where differences observed in vegetation cover were associated with differences in plant communities and not the presence or absence of grazing. Differences in findings from Belote et al. (2009) and Loesser et al. (2007) may be due to the longer-term study with longer drought histories. Both Belote et al. (2009) and Loesser et al. (2007) studies occurred over a nine- and eight-year period, respectively. Belote et al. (2009) used a three of the same exclosures as our study (Surprise, Hartnet and Cathedral), and they observed Hartnet exclosure when its allotment was being actively grazed. Our study included an additional four exclosure locations (Lower South Desert, Muley Twist, The Post and Red Slide), each with different grazing histories, intensities, and recovery periods. Shifts due to drought conditions in Belote et al. (2009) were concluded based only on a singular drought year, 1989, with all the previous year's being mid-range (-1.99cm to +1.99cm) to moderate moist (+2cm to +2.99cm; Figure 8a-b). A consistent drought over a long period of time followed by even more extreme drought, may skew the perception of data and treatments comparisons to their ecological reference site, at least in terms of vegetation. Our data suggests that consistent drought over a long period affects all functional groups, despite treatment, grazing history, or intensity. Ideally, data would have been gathered during the 31-year gap to support Belote et al. (2009). However, gathering future data with continual extreme drought conditions, would provide additional information on the rangeland's overall response to drought conditions.

Most years between 2000-2021 in Wayne county-where Capitol Reef is located-were plagued by drought with late 2020 and 2021 being the most extreme drought condition (https://www.drought.gov/states/utah/county/wayne). Excepting 2001-2002, 2011-2012, 2019-2020, all years between 2000 and 2020, are characterized as "abnormally dry or an even stronger drought rating (moderate drought, severe drought, and extreme drought). These persistent drought conditions may explain the discrepancies between our data and Belote et al. (2009). Our data suggest that consistent drought over a long period affects all functional groups, despite treatment. The true resilience of the rangeland may not be observable until after the extreme drought has passed and may reflect resilience of drought conditions, rather than grazing impacts.

With so many variables in location, soil composition, vegetation, precipitation, and elevation, comparing multiple exclosures under multiple grazing histories and intensities proves difficult. Although our data suggests that removing cattle did benefit the rangeland overall and differences inside versus outside exclosures shows notable e differences. Climate changes may have a greater effect on landscape changes, especially vegetation. With the combination of drought stress and grazing pressure on shaping plant-plant interactions still not fully understood (Verwijmeren et al., 2014), sustainable management of arid and semi-arid regions heavily depends on how land managers understand these ecological processes (Popp et al., 2009). Continued efforts to improve prediction of future trends of both abiotic and biotic factors under climate change are imperative.

With this in mind, we do see recovery over time, but restoration decisions should be taken on a site-by-site basis. Depending on the location and severity of impact, some locations may be able to recover under passive conditions. In locations where vegetation structure severely deviates from its ecological reference site or invasive species are increasing, recovery may not be attainable without active intervention.

Loss of cryptobiotic soil crust may be associated with crossing degradation thresholds as they are ecosystem engineers in high abiotic stress systems (Bowker, 2007). This threshold knowledge may benefit land managers at Capitol Reef who aim to restore plant communities as the park shows large increases in cryptobiotic soil recovery over time with well-developed cryptobiotic soil cover. With loss of cryptobiotic soil important ecosystem engineers, managers should aim to continue positive cryptobiotic soil trends in the park.

With long-term post-disturbance and recovery data rare (Anderson et al., 2008; Bennett et al., 2009; Reich and Lake, 2015), this study provides a unique look into the relationship of both biotic and abiotic factors across a semi-arid rangeland. Additionally unique, site locations across a variety grazing histories; currently grazed, rested for three years and rested for about 30 years. Another study observing passive restoration of vegetation and cryptobiotic soil from grazing (Condon et al., 2020) points out that across the Great Basin, composition, and abundance of biocrusts vary with plant communities. Therefore, restoration goals should be focused on the specific plant community.

In a previous study of cryptobiotic soil cover under a controlled warming environment (Maestre et al., 2013), they found four years after the experiment began, there was reduction of lichens and mosses in areas with well-developed cryptobiotic soil, contrary to our results. As there is currently an incomplete understanding of how cryptobiotic soil crusts will respond to climate change (Young et al., 2016), we aim to point out the importance of understand cryptobiotic soil responses and its association with its plant community under these more frequent drought conditions.

Further work beyond the scope of this study would include a more in-depth analysis of how observed factors interact with each other. Like Belote et al. (2009), future monitoring should observe changes in different vegetation functional groups over time between treatments. This functional group delineation will provide more definitive evidence of long-term drought influences over plant functional group communities between treatments.

More detailed cryptobiotic soil information, such as lichen or moss species presence and morphological group (crustose/squamulose/foliose/fructose lichens, short/tall mosses) can provide us more insight on soil stability, seedling establishment, hydrology, and carbon fixation (Rosentreter et al., 2007). Recording chlorolichens and cyanolichens separate will also provide greater information on nitrogen contributions (Rosentreter et al., 2007).

Additionally, with soil core samples taken, we would like to observe soil bacterial and fungal diversity among sites and treatments. Obtaining this information will help us understand grazing effects on bacterial and microbial communities and potentially their interactions with soil nutrients, cryptobiotic soil and vegetation. This will give us a much larger understanding of ecosystem processes over time and under changing climates.

To confirm ecological sites more confidently, ideally, a soil pit would be dug to match soil properties with that of the assigned ecological site. Confirming the ecological site in the field, instead of through the various sources used, soil properties, vegetation and geologic position can be more confidently matched. If the site deviated from the assigned soil type and ecological site, adjustments are able to be made more efficient and accurately.

Future goals in Capitol Reef National Park should focus on continual monitoring of these exclosure sites. Ideally, with more detailed observations of cryptobiotic soil and vegetation characteristics than we observed in this study. This may include detailed community structure estimates of lichens, mosses and liverworts. Additionally, valuable information would be trajectories of plant communities at each specific location in future years. How both cryptobiotic soil and vegetation interact with soil stability will give land managers insight on how long stability takes to recover and how it is also affected by climate change. With recovery times of cryptobiotic soil varying greatly, depending on preceding sampling (Belnap et al., 2006, 2008), it would be of great interest to the scientific community to observe how it responds in the upcoming years under the current and future climate conditions. The historical data in combination with current and future data will provide a much greater knowledge of interactions among grazing impacts, biotic and abiotic factors, and climate. Further understanding of these interactions will better equip land managers in arid/semi-arid range lands to make management and restoration decisions amidst a rapidly changing climate.

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