ECOLOGICAL RESPONSES TO MEADOW RESTORATION

IN THE SAGEBRUSH STEPPE OF MONTANA

by

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TABLE OF CONTENTS

TABLE OF CONTENTS CONTINUED

LIST OF TABLES

LIST OF FIGURES

LIST OF FIGURES CONTINUED

Figure Page

ABSTRACT

In the semi-arid landscapes of the Western United States, water is scarce. In the high-elevation sagebrush steppe of Southwest Montana, mesic meadows are primarily fed by melting snowpack. These meadows, often centered around an ephemeral stream, collect and store water in the soil. Between 60-80% of wildlife rely on meadows for resources when surrounding upland environments are dry. Warming temperatures and declining snowpack are threatening water resources in this region. Stream incision, often induced by historic land use, decreases water storage in surrounding meadows. In 2018 and 2019, rock restoration structures, typically less than 0.3 m tall, were installed in seven impaired catchments in Southwest Montana to mitigate climate change and incision by slowing and spreading water. Catchments spanned an ecological and abiotic gradient that accounts for over 1/3 of the variation within the sagebrush biome. We compared reaches with structures built to reaches without to assess the impact of restoration on average soil moisture, soil drying rate, water storage, water retention time after rainfall events, and sage-grouse chick food resources (plants and arthropods). We installed soil moisture probes at 24 reaches in 4 catchments, recording hourly measurements from June through September in 2020, 2021, and 2022. Using 0.5 m^2 frames, we estimated vegetation canopy cover at the species level once per month in June through September. We set 24 hr pitfall traps once per month from June through August. Vegetation and arthropod sampling occurred in 2021 and 2022 at 54 reaches in 7 catchments. We detected an increase in water residence time at treated reaches of 1.67 days ($p=0.0069$) after controlling for variation between catchments and reaches. We also found evidence for a 27% increase in total canopy cover ($p=0.004$) at treated reaches when compared to control reaches. We found no evidence of a difference in seasonal soil moisture, seasonal drying rate, seasonal water storage, arthropod activity density, or arthropod biomass. Our results indicate that rock restoration structures are effective in increasing soil moisture and vegetation components in mesic meadows in Southwest Montana, potentially providing a tool to restore meadows in semi-arid landscapes and build resilience against climate change.

CHAPTER ONE

INTRODUCTION TO THESIS

Approximately one-third of the Earth's terrestrial landscapes are semi-arid or arid (Guar and Squires, 2018). In these areas, wetlands and meadows provide important sources of water throughout the year, especially when surrounding areas are dry. Mesic sites, defined as areas of intermediate moisture such as meadows or wetlands, make up less than 3% of the land cover yet 60 to 80% of wildlife rely on them for survival and they support over 30% of all plant species (Atamian et al., 2010; Knight et al., 2014). Long-term population maintenance of these species depends on drought resistant mesic resources that offset climactic variability (Donnelly et al., 2018).

In the high-elevation semi-arid sagebrush steppe of Southwest Montana, mesic sites are primarily fed by water from winter snowpack. Spring snowmelt supports the development of ephemeral streams which support adjacent meadows around them. Montane meadows within semi-arid landscapes have higher water tables, higher plant diversity, and unique arthropod communities compared to upland sites and are important drivers of productivity (Debinski et al., 2000; Yu et al., 2006; Stamps et al., 2009, Mathewson et al., 2013). However, meadows are threatened by climate change and agricultural land use practices.

Across the Western U.S., average annual snowpack has declined since the 1950s and Montana snowpack has declined roughly 20% (Cross et al., 2017). These changes have been attributed to climate change. Projected warming over the next 50 years will continue to reduce snowpack in sagebrush ecosystems. Historical observations also show a shift towards earlier snowmelt and an earlier peak in spring runoff (Cross et al., 2017). In the Rocky Mountains, many rivers and ephemeral streams that are fed by snowmelt, are expected to run at maximum capacity 30 to 40 days earlier by 2099 (Stewart et al., 2004; Marshall et al., 2019). This means that in snowmelt-dominated watersheds, a larger percentage of water will leave during winter and early spring resulting in less water availability during summer and early fall (Cross et al., 2017). While species richness and plant diversity tend to be higher in mesic meadows compared to surrounding uplands, they are also more sensitive to environmental change (Harte and Shaw, 1995; Debinski et al., 2000). Due to these factors, climate change is a threat to the health, quality, and extent of mesic meadows in semi-arid environments and the wildlife that rely on them.

Agricultural practices can degrade ephemeral stream systems, leading to a loss of water storage capacity and making them more sensitive to climate change. Throughout most of the sagebrush steppe of the United States, livestock grazing is the most common land use practice (Knick et al., 2003). The replacement of historic bison herds with domesticated cattle has impacted rangelands in many ways. Cattle avoid steep terrain, spend more time near water, and create trails through repeated travel along the same route (Kohl et al., 2013). Meadows, due to their increased plant production and gentler terrain, attract livestock, which makes them susceptible to overgrazing and increased livestock-induced erosion (Randall et al., 2022). Both factors decrease soil stability and lead to stream channelization. Stream bed incision lowers the water table, leading to earlier drying, changing plant communities, and a reduction in resources for wildlife and livestock (Maestas et al., 2018). Management actions are needed to address these landscape-level changes.

Meadow restoration can repair degraded systems and provide resilience against climate change by increasing the landscape's water storage capacity. However, there is little research on meadow restoration in sagebrush steppe environments (Silverman et al., 2019). Low-tech rock structures in high-elevation basins containing montane meadows may be a solution to increase soil moisture storage in light of declining snowpacks. By mimicking stream riffles, rock structures are designed to slow and spread water, capture sediment, raise water tables, reduce erosion, and increase wetland plants and insects (TNC, 2017; Matherne et al., 2018; Silverman et al., 2019; Zeedyk and Clothier, 2014; Sponholtz and Anderson, 2013). This restoration technique is already being implemented in the arid and semi-arid landscapes of the Western United States. Most research related to these structures has been conducted in the arid landscapes of the southwest United States (Martyn et al., 2022). However, there has been no research how this technique may enhance soil moisture and very limited research on how rock restoration structures influence vegetation and other components of wildlife habitat within montane sagebrush systems (Silverman et al., 2019). This gap in understanding about these structures is important to fill as the effectiveness of rock restoration structures may be site specific and influenced by a variety of environmental and abiotic factors such as elevation, slope, stream incision, aspect, and annual precipitation.

This project was created to investigate the impacts of meadow restoration in sagebrush environments of Southwest Montana. In Chapter Two, I assess whether simple rock restoration structures enhance the retention of soil moisture at both the seasonal scale and the event scale. In Chapter Three, I assess how rock restoration structures may be influencing vegetation and arthropod communities that are used by sage-grouse chicks. In both chapters, I compared areas where rock restoration structures were installed with areas where structures were not installed. In Chapter Four, I discuss the implications for future use of rock restoration structures based on the results from the previous two chapters. My results promote the use of low-tech rock restoration structures to improve mesic meadows in the face of climate change.

CHAPTER TWO

SOIL MOISTURE RESPONSES TO MEADOW RESTORATION IN THE SAGEBRUSH STEPPE

Abstract

In semi-arid environments, high-elevation catchments often contain mesic meadows which act as reservoirs by storing water as soil moisture and releasing it over the summer. Mesic meadows feature higher water tables, increased plant diversity, and increased plant production and provide resources that wildlife and livestock rely on. Climate change, early snowmelt, and historic land use are threatening water resources for people, wildlife, and livestock. Already, across the Intermountain West, spring runoff, derived from winter snowpack, is occurring 1-2 weeks earlier. Since the majority of precipitation comes in the winter in the form of snow, this leads to an increased frequency of late season drought due to soils drying out earlier in the year. Restoration of degraded meadows and maintenance of healthy meadows is one way to support a landscape that is resilient against future change. Ephemeral drainages make up 59% of all streams in the contiguous Unites States, yet there has been little research on how to restore these systems, especially in high-elevation landscapes. In Southwest Montana, over 200 low-tech rock structures, designed to capture sediment, slow runoff, and disperse water flow within a drainage, were installed at four high-elevation catchments (ranging from 2,000 to 2,250 m in elevation) in 2018 and 2019. Rock structures are less than 0.3 m tall and act like dams in the stream bed. The catchments span an ecological and abiotic gradient that accounts for over 1/3 of the variation within the sagebrush biome. In this study, we investigate the impact of rock restoration structures on seasonal soil moisture, seasonal drying rates, soil water retention, and water residence time after rainfall events while accounting for variation between catchments. In the summers of 2020 through 2022, we collected hourly volumetric soil moisture measurements at 24 locations (12 treatment and 12 control) within 4 geographically separate catchments. We found that, after rainfall events, treated reaches held water an average of 1.67 days longer than control reaches ($p=0.0069$). However, we found no difference in average soil moisture, seasonal drying rate, or total seasonal water retention between treatment and control reaches across all catchments and years. The longer water residence time following a precipitation event indicates that low-tech restoration structures provide a promising option for improving montane meadows, increasing water storage across landscapes, and building resilience against a changing climate. Restoration may be most successful in areas with increased summer precipitation, moderate slopes, and loamy soils.

Introduction

Across semi-arid landscapes of the Western United States, water is a limiting factor and storage of this scarce resource is critical to sustain ecosystem functioning. In the high-elevation regions of southwest Montana, and other areas in the Intermountain West, the majority of annual precipitation (62-65%) comes as winter snowfall (Serreze et al., 1999; Cross et al., 2017). Snowbanks are therefore an important reservoir of water in semi-arid sagebrush ecosystems. Water is also stored in soils, as soil moisture, which is concentrated in mesic sites. Mesic sites, or areas of intermediate moisture such as temporary wetlands, streambanks, and wet meadows, are characterized by higher water tables and increased plant diversity and production (Debinski et al., 2000; Yu et al., 2006; Stamps et al., 2009; Mathewson et al., 2013). Despite occupying less than 3% of the landscape, mesic sites are important drivers of productivity within sagebrush ecosystems (Atamian et al., 2010; Silverman et al., 2019; Valayamkunnath et al., 2019). Sixty to eighty percent of wildlife species rely on mesic sites for resources, especially in late summer when surrounding areas have dried (Atamian et al., 2010).

While the value of mesic sites within sagebrush environments is well documented, less is understood about how water moves through high elevation meadows and drainages (McNamara et al., 2005). Soil moisture, in particular, is not well constrained in hydrologic modeling. Soil moisture makes up 13.7% of available surface fresh water – second only to lakes at 75.2% and much greater than rivers at 1.8% (Gleick, 1993). While this accounts for only 0.05% of the total global hydrological cycle, its impact and role is significant (Robinson et al., 2008). Soil moisture affects weather conditions, plant growth, groundwater storage, soil erosion, and flood conditions (Mukhlisin et al., 2021; Pandey et al., 2021). Soil moisture is also the main regulator of runoff from rainfall and snowmelt, a fundamental resource for vegetation, and a driver of structure and diversity in ecosystems (Robinson et al., 2008; O'Donnell and Manier, 2022). However, soil moisture can be difficult to predict as it is influenced by many variables including soil texture, organic matter content, topography, vegetation, and surrounding environmental conditions (Salve and Allen-Diaz, 2001; Gaur and Mohanty, 2013; Knowles et al., 2015; Zapata-Rios et al., 2016; Staudinger et al., 2017). In addition, soil moisture varies both temporally and spatially. Many factors influence soil moisture response to rainfall events including aspect, slope, initial soil water content, and relative elevation (Castillo et al., 2003; Penna et al., 2011; Zhu et al., 2014). Less is known about soil water storage within high-elevation catchments compared to lower elevations areas (Staudinger et al., 2017). However, understanding small scale patterns of soil moisture can help improve the management of water resources.

Threats to Soil Water

Climate change, early snowmelt, and drought threaten the sustainability of water resources for people, wildlife, and livestock. Climate warming is reducing annual snowpack and accelerating spring runoff (Seager and Vecchi, 2010). Across the Western U.S., average annual snowpack has declined since the 1950s and in Montana, snowpack has decreased roughly 20% in the last 80 years (Cross et al., 2017). Snowpack in Montana is at higher risk than other regions since a larger portion is stored at mid and low elevations. Projected warming over the next 50 years will continue to reduce snowpack in sagebrush ecosystems and increase the percent of annual precipitation that comes as rain (Whitlock et al., 2017). Historical observations also show a shift towards earlier snowmelt and an earlier peak in spring runoff (Regonda et al., 2005; Pederson et al., 2011; Cross et al., 2017). In the Rocky Mountains, many rivers and ephemeral streams that are fed by snowmelt, are expected to run at maximum capacity 30 to 40 days earlier by 2099 (Stewart et al., 2004). This means that in snowmelt-dominated watersheds, a larger percentage of water will leave during winter and early spring resulting in less water availability during summer and early fall (Cross et al., 2017). Experimental trials that accelerated snowmelt by 3-6 days resulted in soil moisture depletion in the top 15 inches of the soil profile that lasted for two months (Blankinship et al., 2014).

Land use practices can amplify the negative impacts of climate change. Past and current livestock grazing activities have impacted meadow environments through overgrazing, trailing of livestock through meadows, and trenching for irrigation (Hansen and Wyckoff, 1991; Runge et al., 2019). Meadows and drainages in semi-arid environments are particularly susceptible to enhanced erosion and channel incision by livestock trailing due to the presence of water and palatable forage (Knight et al., 2014; Maestas et al., 2018; Pan et al., 2020; Randall et al., 2022). As incision occurs, and sediment leaves the system, water tables drop resulting in decreasing soil moisture and water storage capacity and a loss of mesic vegetation (Schwinning and Ehleringer, 2001; Wright and Chambers, 2002; Darrouzet-Nardi et al., 2006; Loheide and Booth, 2011). This loss of water storage capacity makes landscapes less resilient against fluctuating precipitation regimes.

Meadow Conservation Structures

Meadow restoration provides an opportunity to improve water storage in the face of climate change. Low-tech strategies are especially desirable due to their versatility, lower associated costs, and ease of implementation (Silverman et al., 2019). Within perennial stream systems, beaver dam analogues are a common technique that slows water, captures sediment, and increases wetland connectivity, thereby enhancing mesic sites around the stream (Scamardo and Wohl, 2020; Salazar, 2022). Studies have shown that beaver dam analogues have the ability to raise the water table, extend the growing season, and increase vegetation productivity (Silverman et al., 2019; Orr et al., 2020; Munir and Westbrook, 2021). However, less is known about how to restore and maintain meadow sites in systems surrounding the ephemeral drainages which make up 59% of all streams in the contiguous United States (Levick, 2008).

The installation of low-tech rock restoration structures, designed to capture sediment and disperse water flow within an ephemeral drainage, can accelerate the recovery of incised channels and increase the extent of mesic sites within sagebrush ecosystems (Zeedyk and

Clothier, 2014; Maestas et al., 2018; Silverman et al., 2019). Slowing and spreading water through the construction of these dam-like structures has the potential to reduce the impacts of climatic variability by retaining more moisture at mesic sites (Donnelly et al., 2016). However, while there is anecdotal evidence of their success, there is very little scientific data supporting these claims. The limited research conducted has focused on the success of reducing erosion, increasing sediment deposition, and increasing vegetation productivity (Norman et al., 2017; Matherne et al., 2018; Martyn et al., 2022; Silverman et al., 2019; Wilson and Norman, 2022). Despite an increase in implementation across the entire Western United States, there has been no research on the effect of rock structures on soil moisture in high-elevation sagebrush steppe ecosystems which have a cold climate and short growing seasons. In addition, past research has focused on individual restoration sites which limits the application of findings to broader landscapes.

In 2018 and 2019, over 300 small rock structures were constructed in four mesic catchments in the high-elevation sagebrush environments of Southwest Montana. The goal of the rock structures is to convert the water stored in high-elevation snowpack into high-elevation soil moisture, retain snowmelt runoff in upper headwater basins, prolong water availability, and increase plant production and late summer greenness. Our research compares areas with and without restoration structures within impaired mesic meadows in sagebrush communities that represent and ecological and abiotic gradient that accounts for over 1/3 of the variation within the sagebrush biome. We installed long-term soil moisture probes within our study catchments to investigate whether this restoration technique is increasing average soil moisture, slowing the seasonal drying rate, increasing water retention time after rainfall events, and increasing overall water storage in years 2-4 post-installation. Since there has been no research on simple techniques to enhance soil moisture in montane sagebrush systems, this research fills an important void in our knowledge of meadow restoration and maintenance.

Methods

Study Area

Our study was conducted within the Centennial and Medicine Lodge valleys of Beaverhead County, Montana (Figure 2.1). Beaverhead County is the largest cattle producing county in the state and both the Centennial and Medicine Lodge valleys are used for summer grazing. There were 4 study catchments that each contained an intermittent drainage channel where rock restoration structures were installed in 2018 or 2019. Catchments were located at Little Basin Tributary 24 (LB24), Little Basin Tributary 7 (LB7), Keystone Tributary (Key), and Teepee Creek (TP) which all occur within the Jefferson River Basin Watershed. The climate of the region consists of short, mild summers and long, cold winters with an average annual temperature of 0 to 2.2°C (PRISM, 2020). All catchments are within sagebrush (Artemisia tridentate spp.) dominated plant communities with a relatively short growing season from June through September. However, catchments span ranges of elevation (2,000-2,250 m), aspect (north or south facing), slope (1-10%), soil type (clay and sandy loams), and drainage incision (5.5 to 14.6 width to depth ratio) (Tables 2-1, 2-2, and 2-3). Precipitation varies annually and between catchments but ranges from 20-71 cm (12 to 28 in) with May and June being the wettest months (USFWS, 2009; PRISM, 2020). Drainages typically flow in May and June, but timing, amount, and duration vary annually. All catchments occur in areas used as summer grazing for cattle, but frequency and intensity of grazing differ.

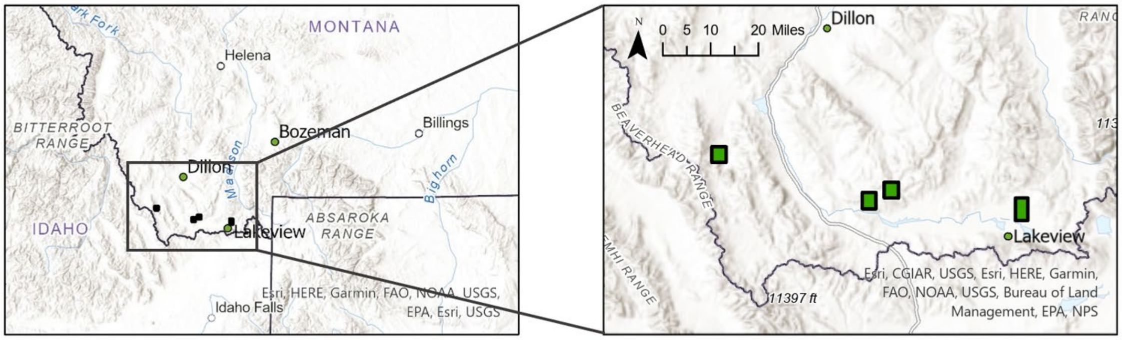

Figure 2-1: Map of four study locations in Southwest Montana where rock restoration structures were installed in intermittent drainages. Drainages were divided into treated and control reaches (n=24) to compare soil moisture from 2020 to 2022. 1. Keystone Tributary (KEY); 2. Teepee Creek (TP); 3. Little Basin Tributary 24 (LB24); 4. Little Basin Tributary 7 (LB7).

Table 2-1: Characteristics of study catchments in Southwest Montana which contained control and treated reaches. Average annual precipitation values obtained from PRISM (2020). June to August precipitation values obtained from rain gauges placed in each catchment.

Catchment	Elevation	Average Annual Precip.	Average Slope of Drainage	Average Aspect of Drainage	Average Gully Width to Depth Ratio	June to August Precip. 2021	June to August Precip. 2022
TP	2000 m	49 cm	1%	172°	14.6	5 cm	9 cm
KEY	2100 m	47 cm	5%	15°	7.8	10 cm	11 cm
LB7	2250 m	58 cm	7%	326°	10.4	11 cm	15 cm
LB24	2200 m	45 cm	10%	18°	5.5	9 cm	10 cm

Table 2-2: Average soil characteristics at each catchment from 6 soil pits, one at each reach. Characteristics listed are from a depth of 15 cm which corresponds to the depth the soil moisture probe was placed. Average annual precipitation values obtained from PRISM (2010).

Catchment	Number of Horizons	Soil Texture at 15 cm	Percent Rock	Rock Size	Percent Roots	Matrix Color Dry	Matrix Color Wet
TP	3	Sandy loam	1%	3 mm	2%	10 yr 3/2	10 yr 2/2
KEY	2	Clay Loam	4%	2 mm	5%	10 yr 3/1	10 yr 3/2
LB7	2	Loam	15%	25 mm	3%	10 yr 2/2	10 yr 2/1
LB24	2	Silty clay/ clay loam	40%	5 mm/ 75 mm	3%	10 yr 4/3	10 yr 3/3

Table 2-3: Average soil structure at each catchment from 6 soil pits, one at each reach. Characteristics listed are from a depth of 15 cm which corresponds to the depth the soil moisture probe was placed.

Catchment	Type	Size	Grade
TP	Subangular/blocky	Fine	1
KEY	Subangular/blocky	Medium	3
LB7	Granular	Very fine	1
LB24	Angular/blocky	Medium	1

Treatment Design

At each of the four catchments, a restoration professional selected locations for the installation of water slowing structures. The structures, termed one-rock dams, are small structures made of a single layer of rocks laid by hand on the stream bed and are typically less than one foot tall (Zeedyk and Clothier, 2014). They are intended to mimic stream riffles and designed to gradually increase streambed elevation by accumulating roughness and capturing sediment (Figure 2) (Zeedyk and Clothier, 2014; Sponholtz and Anderson, 2013). Each drainage channel was divided into at least six reaches. A reach was defined as three or more water slowing structures or potential locations for water slowing structures. Reaches were considered separate experimental units when they were separated by 20 times the channel/gully width (Sutton, 2022). The uppermost reach in each catchment was randomly assigned to be treated (structures built) or to be a control (no structures built) before structures were installed. Reach treatment was then assigned in an alternating order going downstream from the uppermost reach. This created pairs of treated and control reaches that could be compared.

A total of 24 reaches (12 treatment and 12 control) were sampled during the summers of 2020, 2021 and 2022 to collect data 2-, 3-, and 4-years post-treatment. Reaches were evenly divided between all four catchments. Six reaches at LB7 (3 treatment and 3 control) were sampled during the summer of 2019 as a pilot study. Sampling occurred from late May through September, but timing varied by catchment and year due to road access and timing of snowmelt. Since the rock structures are designed to work in concert with each other, sampling occurred at the lowermost structure or location within a reach.

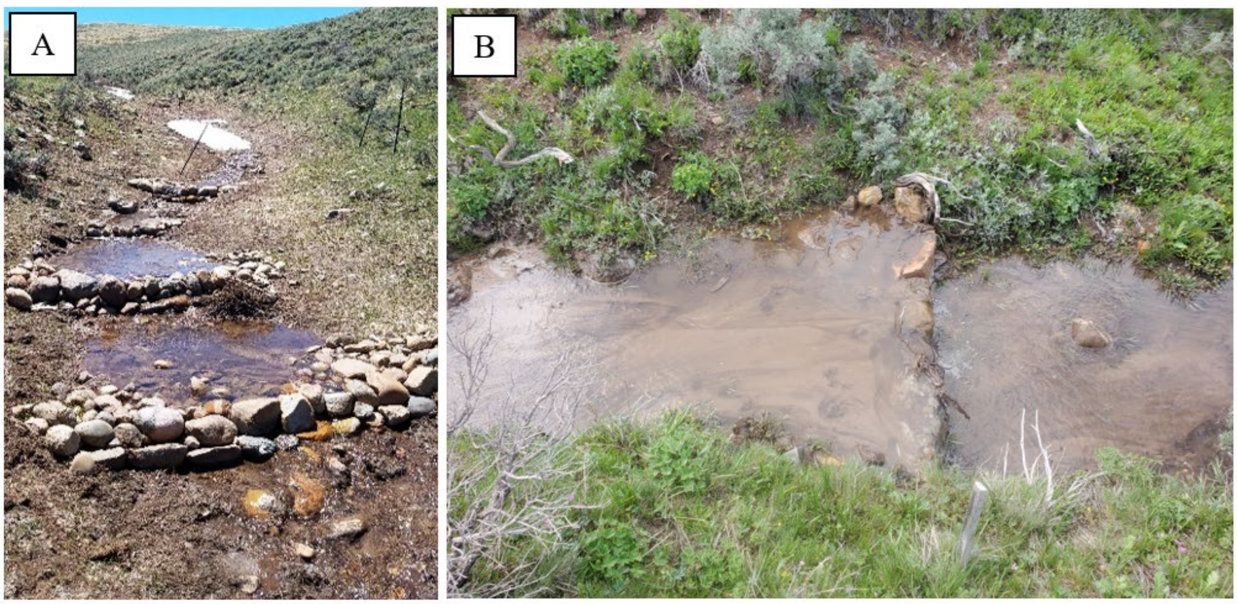

Figure 2-2. Examples of low-tech rock structures constructed in Southwest Montana. (A) Water from snowmelt pooling behind restoration structures in June 2021 at Little Basin Tributary 24. (B) Sediment collection upstream of restoration structures in June 2022 at Little Basin Tributary 24. Photograph credits: Laura Robison.

Soil Moisture and Weather Monitoring

At each reach, we installed a HOBO H21-USB Microstation, with a set of three volumetric soil moisture probes (HOBO 10HS moisture sensors), 2 m upstream from the most downstream structure in the reach on river left. We placed one probe at 0 m, 1 m, and 3 m from the centerline of the gully. The probes were buried at 15 cm depth since the majority of grasses and forbs reach their maximum root density at less than 20 cm (Sun et al., 1997). In addition, soil moisture at this depth is affected by precipitation and evaporation more than soil moisture at deeper depths (Mukhlisin et al., 2021). Data loggers were programmed to record a measurement every hour. At each catchment, we mounted a Davis tipping bucket (0.2 mm accuracy) rain gauge to one of the Microstations to record rainfall events and total precipitation. Microstations were installed in late May and early June shortly after snowmelt and were removed in September prior to freeze-up (Table 2-4). In addition to soil moisture probes being placed for different lengths of time based on seasonal constraints, some probes were disturbed by cattle and rodents. While we tried to re-install the probes as soon as possible, this inevitably resulted in missing soil moisture measurements at some reaches.

Table 2-4: Dates of installation and removal of soil moisture probes during each year at each of the four catchments. In 2019, soil moisture probes were only installed at LB7. Dates varied due to road conditions and snowmelt timing.

Catchment	2019		2020		2021		2022	
	Start	End	Start	End	Start	End	Start	End
TP	N/A	N/A	June 2	Oct 7	May 16	Sept 15	May 23	Sept 7
KEY	N/A	N/A	June 2	Oct 11	May 24	Sept 1	May 20	Sept 6
LB7	July 2	Oct 3	June 2	Oct 7	May 30	Sept 7	May 25	Sept 12
LB24	N/A	N/A	June 2	Sept 24	June 8	Sept 8	June 11	Sept 13

Catchment Characterization

Since our catchments span a geographic and topographic gradient, we collected catchment data on physical factors that are known to influence soil moisture: 1) elevation, 2) average annual precipitation, 3) average slope of drainage, and 4) average gully width to depth ratio. At each reach, we dug a 30 cm pit 0.5 m downstream of the 1 m probe location for a total of 6 soil pits per catchment. We identified the soil horizons present and for each horizon recorded depth, soil structure, percent rock fragments, rock fragment shape and size, percent roots, texture, and matrix color using field characterization methods set by the USDA (Schoeneberger, 2012).

Seasonal Soil Moisture Analysis

Hourly volumetric soil moisture was averaged on the daily time scale. We only included data from June through August in our analysis, as these are the months when soil moisture is most important for plant growth (Mueggler, 1972). These are also the months where we had data across all catchments and years as installation and removal dates varied from catchment to catchment and year to year. Soil moisture changes caused by rainfall events result in a rapid increase, typically reaching their peak in less than 3 days. They then gradually descend, taking longer than 3 days to decline to pre-event levels. Due to this, measurements from every third day were used for analysis to maintain some sampling independence. We modeled average daily soil moisture as a function of treatment, month, and year using linear mixed models. We included random intercepts for catchment, pair, reach, and distance from gully center to account for repeated sampling and the nested nature of our experimental design. We modeled all combinations of our fixed covariates and compared these models based on Akaike's information criterion (AIC). We eliminated models based on a ΔAIC cutoff of 2 (Arnold, 2010). We completed a separate analysis for each catchment in addition to an overall analysis. For these, we removed catchment from the random effects.

Seasonal Drying Rate Analysis

For each soil moisture probe in each year, we subset the daily soil moisture from June 1st to August 15th as this period is when the primary drying takes place in high-elevation meadow systems and typically contained both the highest and lowest soil moistures of the season. We then calculated a linear line of best fit and identified the slope of this line. We modeled the slopes as a function of treatment and year using linear mixed models. We included random intercepts for catchment, pair, reach, and distance from gully center to account for repeated sampling and the nested nature of our experimental design.

Seasonal Water Storage Analysis

We compared the area under the volumetric soil moisture curve for paired reaches and calculated the difference in total water retention over the entire season (Figure 2-3). We modeled the difference in total water retention as a function of year and distance from gully center using linear mixed models. We included a random intercept for catchment to account for repeated sampling.

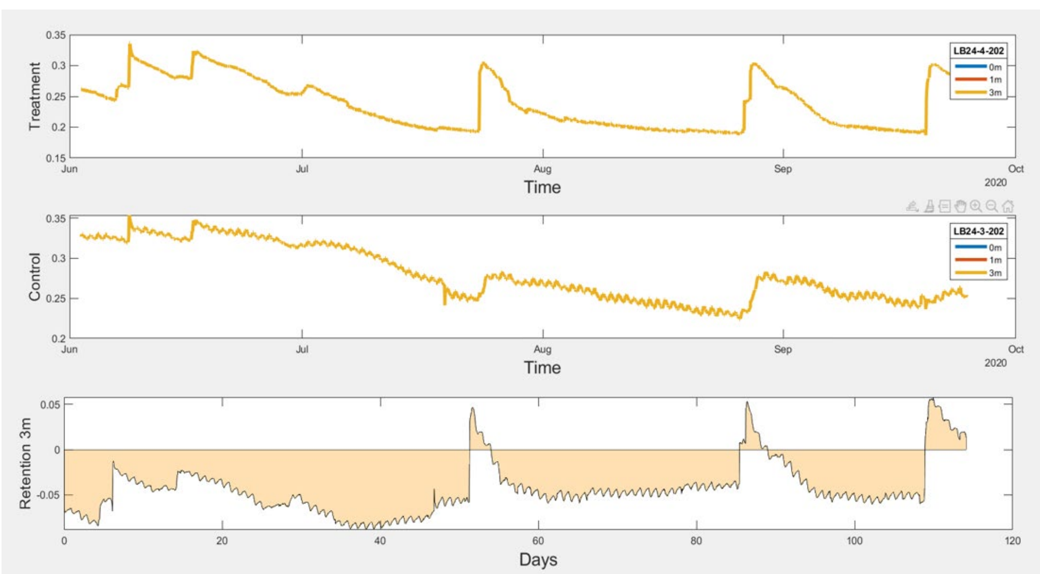

Figure 2-3: One example of the difference in seasonal water storage between paired treatment and control reaches. The top two volumetric soil moisture graphs show data from 3 m probes at two reaches at LB24 in 2020. The treatment reach is where restoration structures were installed. The bottom graph shows the control soil moisture subtracted from the treatment soil moisture. Negative values indicate the control had higher soil moisture and positive values indicate the treatment reach had higher soil moisture.

Soil Moisture Responses to Summer Precipitation Analysis

Rainfall events were identified by looking for sudden increases in soil moisture that occurred at more than one reach at a catchment. In 2021 and 2022, increases in soil moisture resulted from precipitation events greater than 8 mm. For each rainfall event, we recorded whether the soil moisture at each probe increased, decreased, or stabilized. We modeled these responses to a rainfall event as a function of treatment using an ordinal logit regression that had event, catchment, location in the drainage, and distance from gully center as random effect. An ordinal logit regression was used since the categories (increased, decreased, or stabilized) have a clear order. Since we were interested in how long water was held on the landscape, we identified all occasions where both reaches in a pair experienced an increase in soil moisture. For these instances, we measured the time from the initial increase in soil moisture to when the soil moisture returned to pre-event levels. We modeled response to rainfall events as a function of treatment, catchment, and distance from gully center using linear mixed models. We included random intercepts for rainfall event and pair to account for repeated sampling, the nested nature of our experimental design. We modeled combinations of our fixed covariates and compared these models based on Akaike's information criterion (AIC). We eliminated models based on a ΔAIC cutoff of 2 (Arnold, 2010).

Results

In 2021, the Jefferson River Basin watershed received 94% of normal annual precipitation in 2020, 79% in 2021, and 99% in 2022 (USDA, 2022). Across all years, TP and KEY had the earliest melt-off occurring in April. At LB7, melt-off occurred in early May while at LB24, melt off did not occur until early June.

Catchment Characterization

Three of the four catchments were north-facing and only TP had a predominantly southerly aspect (Table 2-1). Soil characteristics varied between basins; notably TP contains the highest soil sand content, LB7 has more well-developed loamy soils, and LB24 has the most rocks present within the sampled soil profile (Table 2-2 and 2-3; Figure 2-4). The degree of channel incision varied as well with LB24 having the lowest width to gully depth ratio (5.5) and TP having the highest (14.6). Average soil moisture also varied between sites. Across all years from June through August, average soil moisture at TP was 0.1960 m^3/m^3, KEY was 0.2799 m^3/m^3, LB7 was 0.3215 m^3/m^3, and LB24 was 0.2399 m^3/m^3.

Figure 2-4: Representative soil pits at each catchment dug 30 cm deep 0.5 m upstream of lowermost structure or location in a reach. (A) Teepee Creek (TP); (B) Keystone Tributary (KEY); (C) Little Basin Tributary 7 (LB7); (D) Little Basin Tributary 24 (LB24)

Seasonal Soil Moisture

We did not detect differences in average soil moisture levels between treatment (0.32 m^3/m^3, SE=0.01 m^3/m^3) and control (0.33 m^3/m^3, SE=0.04 m^3/m^3) reaches after accounting for variations between catchment, reach pair, reach, and distance ($p=0.1749$) (Table A.1). Similarly, we did not detect differences in average soil moisture levels between treatment and control reaches at any given catchment after accounting for variations in reach pair, reach, and distance ($p>0.10$). As expected, soil moisture at all catchments and treatments, decreased from June to August ($p<0.001$). Soil moisture differed by year as well, with soil moisture the lowest in 2020 (0.23 m^3/m^3, SE=0.14 m^3/m^3) and the highest in 2019 (0.33 m^3/m^3, SE=0.04 m^3/m^3) ($p<0.001$).

Seasonal Drying Rate

There was no difference in the drying rate of treatment and control reaches ($p=0.1736$). The average drying rate for control reaches was 0.0023 m^3/m^3 (SE=0.0002 m^3/m^3) per day across all years and sites while the drying rate for treatment reaches was 0.0026 m^3/m^3 (SE=0.0003 m^3/m^3) per day (Table A.2). Soil at 3 m from the gully center dried on average 0.0005 m^3/m^3 per day slower than soil at 0 m and 1 m from the gully center ($p=0.0375$).

Seasonal Water Storage

The difference in total water retention between treatment (37.2 m^3/m^3, SE= 4.8) and control (39.6 m^3/m^3, SE= 6.0) reaches (n=41) was not different from zero ($p=0.204$). We found no difference in total water retention based on distance from gully center or year ($p>0.10$).

Soil Moisture Responses to Summer Precipitation

We detected a total of 34 rainfall events across all catchments and years (Table 2-5). For treated reaches, the odds of experiencing a soil moisture increase or stabilization, is 2.98 times greater (SE=1.38) than that of control reaches ($p=0.008$). This resulted in a 97% chance of an increase for treated reaches and an 89% chance of increase for untreated reaches.

Table 2-5: List of rainfall events at each catchment by start date from 2020 to 2022. Rainfall events were identified by looking for increases in soil moisture that occurred at more than one reach within a catchment.

Catchment	2020				2021				2022			
TP	6/16	7/23			5/22	7/2			5/28	7/3	7/15	8/13
KEY	6/16	7/29			8/6				5/29	6/12	7/3	8/11
LB7	6/16	7/29	8/26		8/19				7/14	8/11	8/21	
LB24	6/16	6/28	7/23	8/26	7/1	7/20	8/1	8/19	6/12	7/14	7/29	8/11

Across all catchments, we estimate that treated reaches hold moisture an average of 1.67 (SE=0.61) days longer than control reaches (p-value=0.0067) (Figure 2-5; Table A.3). There was no difference between how long water was held at different distances from the gully center indicating that the treatment had the same effect up to 3 m from the gully center ($p>0.10$). There was also no difference in water residence time between catchments ($p>0.10$). Model selection using AIC scores supported the model with no treatment by catchment interaction so the additive model was used (ΔAIC=4.2).

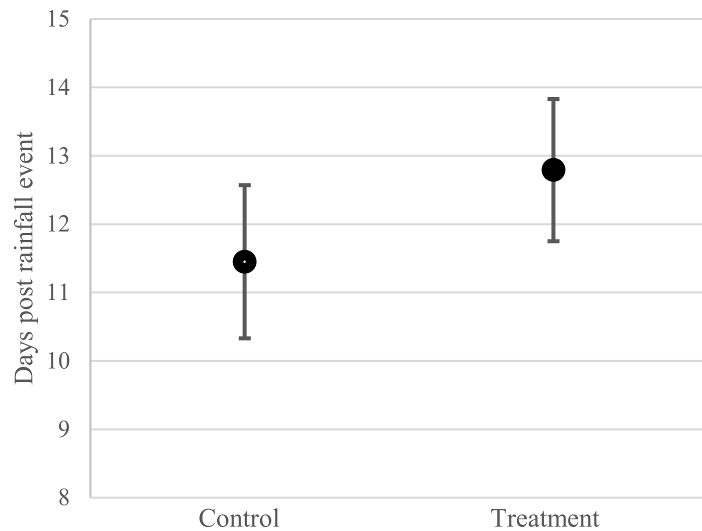

Figure 2-5: Number of days treatment and control reaches experienced an increase in soil moisture after a rainfall event across all four study catchments (means +/- SE). Treatment reaches are where rock restoration structures were installed and held soil water 1.67 days longer than control reaches (p=0.0069). Differences were tested using linear mixed effects models.

Discussion

The restoration structures investigated in this study were built with the intent to increase soil moisture in mesic meadow habitats by capturing snowmelt runoff. However, we found no increase in average soil moisture across the summer season at reaches where restoration structures were installed compared to control reaches. In addition, treated and control reaches had similar seasonal drying rates and seasonal water storage. Since, observationally, we did see an increase in sedimentation and pooling immediately upstream of restoration structures, more time may be needed to detect measurable differences. Rebuilding the bed of an incised drainage can increase soil water storage (Scamardo and Wohl, 2020). However, sedimentation is a slow process. Depending on the degree of incision, it may take several years for an impactful amount of deposition to occur in ephemeral systems which have less water flow and more sporadic sediment movement (Billi et al., 2018; Jaeger et al., 2017). Research at a perennial stream site located between Little Basin 24 and Teepee Creek where beaver dam analogs were installed, showed an increase in soil moisture at a 30cm depth in the 2 years post-treatment (Whitehead, 2020). Our differing results could indicate that the lack of overall soil moisture response to treatment was due to the limited amount of water that flows in ephemeral drainages compared to the perennial streams where beaver dam analogues are implemented.

Our data show that between 2 and 4 years post-installation, low-tech rock restorations structures increased the time that water was held on the landscape after rainfall events in four ephemeral catchments in Southwest Montana. While this project was designed with the intention of capturing snowmelt, our data indicate that rock structures are effective at capturing rainfall which may be increasingly important in the future climate. Climate projections predict that, while Southwest Montana may receive similar or higher annual precipitation, this precipitation is more likely to come as rain rather than snow (Whitlock et al., 2017). More frequent or extreme rainfall events necessitate a landscape that is able to capture and store water in ways other than snowbanks. Water retention in soils can potentially provide this resilience but only if high-elevation drainages are intact and properly functioning. This requires floodplain connectivity and meadows that have higher water storage capacity. While heightened soil moisture for an additional 1.67 days is relatively short, when applied across multiple rainfall events (like the average of 3 rainfall events per season that we observed), the amount of time becomes more substantial. In 2021, a drought year, the restored reaches we observed may have only experienced 3 extra days of moisture. However, in a normal year such as 2022, the restored reaches may have experienced 7 extra days. In montane environments, with short growing seasons, even a week of extra water can have a meaningful impact on vegetation growth and production availability (Mueggler, 1972; Dolezal et al., 2020). However, this also indicates that effects may be limited in areas with low summer rainfall.

The catchments in this study differed in average soil moisture but they did not differ in treatment effect. Since all sites experienced a similar increase in water retention after rainfall events, this indicates that rock structures are potentially suited for most ephemeral drainages that have similar characteristics to those in this study. However, due to the limited number of reaches in each catchment, we may have not had the statistical power to detect small differences between catchments. Some of the data indicates that Little Basin 7 may have had the greatest treatment success. Characteristics associated with catchment that may increase the treatment effect include increased water flow from a spring, higher precipitation, moderate slopes, and loamy soils. On the contrary, Little Basin 24 may have had the least treatment success despite holding snow the latest out of all four catchments. Characteristics associated with this catchment that may decrease treatment effect include a steeper slope, greater incision, and rockier soil. Further investigation into our data may reveal insights into mechanisms that are driving soil moisture in high-elevation sagebrush steppe systems. By tying in reach specific data regarding soil characteristics, topography, underlying geology, monthly precipitation, elevation, and average temperatures, we hope to better understand what is driving the differences between soil moisture in montane catchments regardless of restoration status.

Soil moisture is the primary driver that differentiates meadows from surrounding uplands (O'Donnell and Manier, 2022). As such, increasing soil moisture is the first step towards successful meadow restoration. However, the increases in water retention time after rainfall events that we observed were not enough to result in an increase in seasonal soil moisture. One possibility is that after the moisture pulse from a rainfall event, the water dispersed throughout the catchment, diminishing the effect seen at the individual reach. Another possibility is that there were not enough rainfall events each year to significantly impact the broader soil moisture regime. A larger dataset with soil moisture probes placed across a larger area would be needed to investigate these hypotheses.

Sagebrush dominated ecosystems exist across much of the western United States and encompass a broad ecological and abiotic gradient. Our study sites represent about a third of the moisture and temperature variability in sagebrush systems. Given the similar responses to treatment across all catchments in the study, it appears that rock restoration structures could help enhance soil moisture across a wide range of environmental conditions. Our research shows that low-tech restoration with rock structures can be one mechanism in helping restore and maintain montane meadows thereby building resilience against a changing climate through improved water storage.

In addition, ephemeral and intermittent streams are prevalent across the semi-arid west and rock structures have the potential to be implemented across a vast geographic region (Levick, 2008). However, impactful change in ephemeral stream systems may take more time than in perennial systems that have more water. Since the magnitude of impact on the entire meadow ecosystem is unknown, rock structures could also be used in tandem with other techniques such as snow fences or more intensive stream restoration to promote a more rapid recovery from a degraded state (David, 2013; Silverman et al., 2019). In the future, longer-term monitoring of the rock restoration structures would be worthwhile to understand their full effect, especially when implemented in semi-arid montane environments.

CHAPTER THREE

MONTANE MEADOW RESORATION IMPROVES COMPONENTS OF SAGE-GOUSE HABITAT

Abstract

Across semi-arid landscapes, water is a limiting factor that is essential for life. As a result, 60-80% of wildlife species rely on high-moisture (mesic) sites to survive. In the montane sagebrush steppe of Southwest Montana, a topographically diverse and high-elevation landscape, water from spring snowmelt and summer rainfall is often stored as soil moisture in mesic meadows. As a result, plant communities in montane meadows differ from the surrounding uplands and are important drivers of wildlife habitat productivity within sagebrush ecosystems. The greater sage-grouse (*Centrocercus urophasianus*), a sagebrush obligate species, rely on montane meadows for forage (forbs and arthropods) during late-summer when the surrounding uplands dry. Climate change is reducing snowpack and advancing the timing of spring runoff. Changes in seasonal water availability and degradation of streams due to livestock-induced erosion are threatening montane meadows and the resources they provide to wildlife. Ecological restoration of montane meadows provides a unique opportunity to improve the resiliency of impaired sage-grouse habitat in the face of climate change and land use change. However, limited research exists on restoration effectiveness in montane landscapes. In 2018 and 2019, over 300 ecological restoration structures (i.e. rock dams) were installed in impaired montane meadows systems across Southwest Montana, an area critical for sage-grouse conservation. These catchments span an ecological and abiotic gradient that accounts for over 1/3 of the variation within the sagebrush biome. The rock structures constructed are less than 0.3 m tall and act like flow-over dams in the stream bed, slowing and spreading the water immediately upstream at localized-scales. In 2021 and 2022, we investigate 1) if, where, and how much this restoration technique is increasing plant canopy cover and/or arthropod availability of sage-grouse chick food resources at 54 reaches in 7 catchments, and 2) how these findings can inform future management of mesic habitats. We estimated total vegetation canopy cover and live canopy cover based on field-derived measurements (0.5 m^2 scale using frames) of total plant species level at six locations in each reach once per month in June through September (n=1,296). We additionally placed 24 hr pitfall traps at three locations in each reach once per month from June through August (n=486). In comparison to control reaches, we found a 27% increase in the total canopy cover (p=0.004) and a 30% increase in the live canopy cover (p=0.003) across both years within treated reaches. We found no difference in arthropod activity density or dry biomass between treated and control reaches. Our results indicate that meadow restoration using low-tech rock structures in ephemeral drainages is an effective way to increase vegetation associated with sage-grouse food resources and can potentially increase the resiliency of sagebrush steppe landscapes in the face of climate change.

Introduction

Across semi-arid landscapes, water is a limiting factor that is essential for life (Schwinning and Ehleringer, 2001). In the sagebrush steppe of the American West, as snow melts in the spring and rain falls in the summer, water collects and is stored as soil moisture in mesic meadows that are fed by seasonal streams (Knight et al., 2014). Mesic areas are critical to retaining soil moisture late into the summer and are important drivers of productivity within sagebrush ecosystems (Silverman et al., 2019; Valayamkunnath et al., 2019). Mesic meadows feature higher water tables, increased plant diversity, higher forb cover, and unique arthropod communities when compared to adjacent arid upland catchments (Debinski et al., 2000; Yu et al., 2006; Stamps et al., 2009; Mathewson et al., 2013). When degraded, from past disturbance (e.g. irrigation, farming, grazing, livestock-induced erosion), decreasing soil moisture and dropping water tables can allow for invasion of more arid dwelling sagebrush and a subsequent loss of water availability and mesic vegetation (Schwinning and Ehleringer, 2001; Wright and Chambers, 2002; Darrouzet-Nardi et al., 2006).

Climate warming is threatening, already scarce, mesic meadows of the sagebrush steppe ecosystem. Increases in average annual temperature is reducing annual snowpack and accelerating spring runoff (Seager and Vecchi, 2010). Snowpack in Montana has declined 20% in the last 80 years and is at higher risk than other regions in the west since a larger portion is stored at mid to low elevations (Cross et al., 2017; Whitlock et al., 2017). Already, since 1950, spring runoff in Montana and in other western states has shifted 1-2 weeks earlier in basins below 2400 m (8000 ft) (Regonda et al., 2005; Pederson et al., 2011). This causes meadows to dry earlier in the year and increases the frequency of late-season drought. In addition, mesic meadows are more sensitive to environmental conditions when compared to other meadow types as well as upland sagebrush communities (Debinski et al., 2000). Forbs are more abundant in meadows and are more vulnerable to changes in moisture. Due to this, increased warming can lead to a sagebrush (Artemesia spp.) dominated landscape (Harte and Shaw, 1995). Other threats such as invasive species, conifer encroachment, and livestock grazing compound upon the impacts of climate change on sagebrush ecosystems (Heyerdahl et al., 2006; Bradley, 2010; Reisner et al., 2013).

Livestock grazing and associated agricultural activities have profoundly changed the sagebrush landscape. Higher elevation, montane environments often serve as summer pastures for livestock that are kept at lower elevations during other times of the year (Bailey, 2004). Past and current livestock grazing activities have impacted meadow environments through overgrazing, trailing of livestock through meadows, and trenching for irrigation (Hansen and Wyckoff, 1991; Runge et al., 2019). Meadows are susceptible to overgrazing and enhanced erosion by livestock due to the presence of water and palatable forage, especially in late summer (Knight et al., 2014; Randall et al., 2022). Channel incision and gully erosion lower the water table, drying the surrounding soils and making mesic sites more at risk to climate change due to a loss of natural water storage (Maestas et al., 2018).

Importance to Wildlife

Despite mesic sites making up less than 3% of the landscape, mesic sites provide critical habitat, food, and water resources for many wildlife species (Donnelly et al., 2016; Silverman et al., 2019). Around 60 to 80 percent of wildlife species are dependent on higher-moisture areas (Atamian et al., 2010). The greater sage-grouse (*Centrocercus urophasianus*, sage-grouse hereafter) is one of these species. Sage-grouse, a sagebrush obligate species that occupies elevations less than 8,000 ft, are often used as an indicator of the overall health of sagebrush ecosystems (Call, 1985; Rowland et al., 2006; Hanser and Knick, 2011). Sage-grouse chicks rely heavily on forbs and arthropods especially during their first 12 weeks of life (Klebenow and Gray, 1968; Peterson, 1970; Johnson and Boyce, 1990). Forbs remain the dominant (75%)

component of chick diet until they are greater than 3 months of age. Female sage-grouse often move their broods from breeding areas to areas with more moisture and higher forb cover once chicks are 2-3 weeks old due to seasonal drying and senescence of herbaceous vegetation in adjacent uplands in mid-summer (Peterson, 1970; Drut et al., 1994a; Fischer et al., 1996; Sveum et al., 1998; Atamian et al., 2010; Donnelly et al., 2016; Severson et al., 2022). Declining occupancy range and population size of sage-grouse throughout the 20th century is in part due to habit degradation (Connelly et al., 2004; Rowland et al., 2006; Storch, 2007; Coates et al., 2016). When high-quality brood-rearing habitat is scarce and spread out, females with broods tend to have larger home range sizes, which can result in lower chick survival (Drut et al., 1994a; Severson et al., 2022). Since high-moisture sites in semi-arid regions account for less than 3% of the landscape, they may be a limiting component of sage-grouse late brood rearing habitat (Atamian et al., 2010; Donnelly et al., 2018).

Meadow Conservation Structures

Mesic meadow restoration provides an opportunity to improve sage-grouse, and other wildlife, habitat in the face of climate change (Donnelly et al., 2018). The installation of low-tech rock restoration structures in seasonal streams has the potential to increase the extent of meadows within sagebrush ecosystems (Zeedyk and Clothier, 2014; Maestas et al., 2018; Silverman et al., 2019). By slowing and spreading water, rock structures may also reduce the impacts of climatic variability by retaining more moisture at mesic sites (Donnelly et al., 2016). This project focuses on one-rock dams described by Zeedyk and Clothier (2014) that were installed within the 1.5 million acre core sage-grouse area of Southwest Montana. One-rock dams are small restoration structures made of a single layer of rocks laid by hand on the stream bed and are typically less than 0.3 m tall (Zeedyk and Clothier, 2014).

Rock restoration structures have been widely implemented in the southwest U.S. and yet there is little data regarding their effect on vegetation or other wildlife resources. Silverman et al. (2019) were the first to attempt to assess rock structures in the Intermountain West at a site in the Gunnison Basin of Colorado. Using satellite imagery, they observed an increase in vegetation productivity by 24% and vegetation productivity was extended longer into the growing season during the four years post-treatment (Silverman et al., 2019). However, such broad measurements of vegetation do not allow for a detailed understanding of the ecological responses in wildlife habitat. In addition, past research has focused on individual restoration sites which limits the application of findings to broader landscapes. The application of rock structures in montane environments in Montana is relatively new and has not been studied. Regional research is critical as environmental variables such as topography, precipitation regimes, and soil type strongly influence vegetation, soil moisture, and how the two interact. Thus, further evidence of the effectiveness of rock structures is needed in the northern Intermountain West, especially in locations with cooler climates, and shorter growing seasons (Silverman et al., 2019).

Our research compares areas with and without restoration structures within impaired mesic meadows in sagebrush communities in Montana. We investigate whether this restoration technique is increasing plant canopy cover (total percent canopy cover and percent live canopy cover) or arthropod availability (activity density and dry biomass) of sage-grouse chick food resources in years 2-4 post installation and builds upon research conducted in years 1-2 post-installation by Sutton (2022). Our research seeks to contribute to our understanding of how and if rock restoration structures contribute to the improvement of wildlife habitat.

Methods

Study Area

In 2018 and 2019, over 300 small rock structures were constructed in high elevation sagebrush meadows within the Centennial and Medicine Lodge valleys of Beaverhead County, Montana. The climate of the region consists of short, mild summers and long, cold winters, an average annual temperature of 1.6°C, and a relatively short growing season from June through September (USFWS, 2009). Precipitation varies annually as well as between catchments but ranges from 30 to 71 cm (12 to 28 in) with May and June being the wettest months (USFWS, 2009; PRISM, 2020).

We had 7 study catchments centered around an ephemeral drainage where water slowing structures were installed in 2018 or 2019. Catchments were located at Little Basin Tributary 24 (LB24), Little Basin Tributary 7 (LB7), Keystone Tributary (KEY), Teepee Creek (TP), Clover Tributary (CT), Clover Pass (CP), and Snowshoe Creek (SS) (Figure 3-1). All catchments were within sagebrush (Artemisia tridentate) dominated plant communities that contained wet meadow vegetation communities surrounding the drainage. However, catchments span ranges of elevation (2,000-2,250 m), aspect (north or south facing), slope (1-10%), soil type (clay and sandy loams), and drainage incision (5.5 to 14.6 width to depth ratio) (Table 3-1).

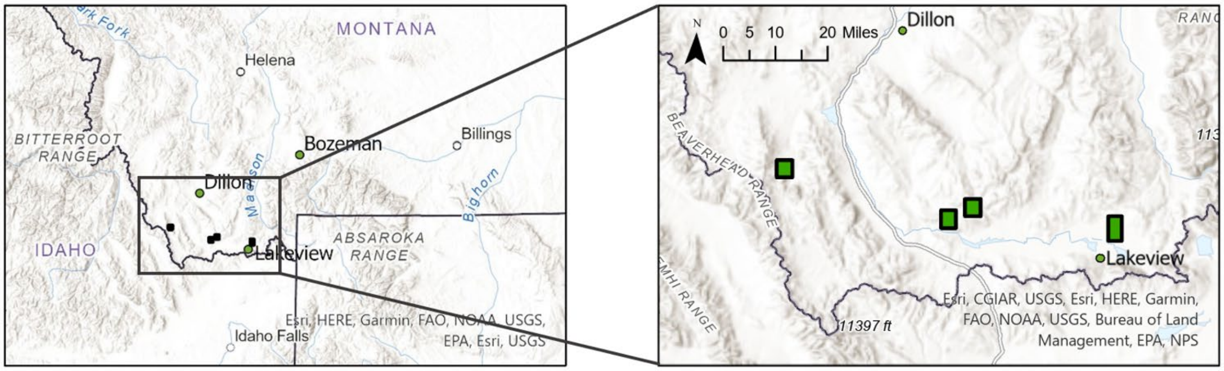

Figure 3-1: Map of four study locations in Southwest Montana where rock restoration structures were installed in intermittent drainages. Drainages were divided into treated and control reaches (n=24) to compare soil moisture from 2020 to 2022. 1 – Keystone Tributary, 2 – Little Basin Tributary 24, 3 – Little Basin Tributary 7, Clover Tributary, and Clover Pass, 4 – Teepee Creek and Snowshoe Creek.

Since our catchments spanned a geographic and topographic gradient, we collected catchment data on various physical factors that are known to influence vegetation and arthropod communities including elevation, average annual precipitation, average slope of drainage, average gully depth, and soil texture at 15 cm (Table 3-1). More detailed catchment descriptions, including information about soil moisture, can be found in Chapter Two of this thesis.

Table 3-1: Catchment characteristics (elevation, average annual precipitation, average slope, average gully depth, and soil texture at 15 cm) for seven catchments in Southwest Montana where rock restoration structures were installed. These catchments were used as study sites to compare treated reaches to untreated reaches. Catchments: Little Basin Tributary 24 (LB24), Little Basin Tributary 7 (LB7), Keystone Tributary (KEY), Teepee Creek (TP), Clover Tributary (CT), Clover Pass (CP), and Snowshoe Creek (SS). Average annual precipitation values obtained from PRISM (2010).

Catchment	Elevation	Average Annual Precipitation	Average Slope of Drainage	Average Gully Width to Depth Ratio	Soil Texture at 15cm
TP	2000 m	49 cm	1%	14.6	Sandy loam
KEY	2100 m	47 cm	5%	7.8	Clay Loam
LB7	2250 m	58 cm	7%	10.4	Loam
LB24	2200 m	45 cm	10%	5.5	Silty clay/clay loam
CT	2200 m	57 cm	4%	12.8	Loam
CP	2300 m	58 cm	3%	4.5	Loam
SS	2050 m	55 cm	2%	8.2	Sandy Loam

Treatment Design

At each of the seven catchments, a restoration professional selected locations for the installation of rock restoration structures where they would best mimic natural processes and functions in streams. Rock restoration structures are intended to slow and spread water, mimic stream riffles, and gradually raise the stream bed by increasing roughness and capturing sediment (Figure 3-2) (Zeedyk and Clothier, 2014; Sponholtz and Anderson, 2013). Each drainage channel was divided into multiple experimental units (hereafter, *reaches*). A reach was defined as a grouping of three or more water slowing structures or potential locations for water slowing structures in control, untreated, reaches. Reaches varied in length but were at least 20 times the channel/gully width (Sutton, 2022). The uppermost reach at each catchment was randomly assigned to be treated (structures built) or to be a control (no structures built) before structures were installed. Treatment versus control reaches were then systematically assigned in an alternating order going downstream from the uppermost reach.

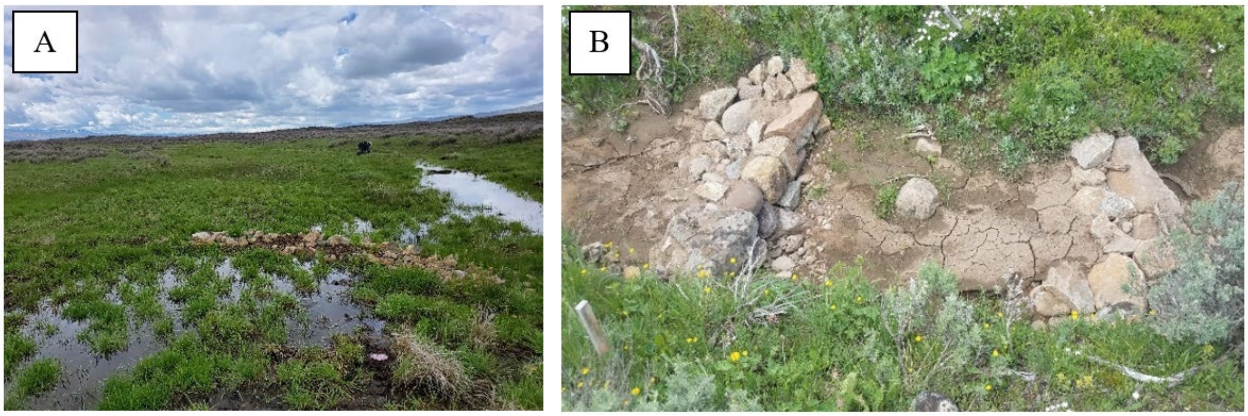

Figure 3-2: Examples of low-tech rock structures constructed in Southwest Montana. (A) Water pooling behind restoration structures in June 2022 at Teepee Creek. (B) Sediment collection upstream of restoration structures in June 2020 at Little Basin Tributary 24. Photograph credits: Laura Robison.

We sampled a total of 54 reaches (26 treatment and 28 control) once per month from June through September during the summers of 2021 and 2022 between 2 to 4 years post-installation of structures. Reaches were divided between catchments as follows: LB24 – 2 treatment, 3 control; LB7 – 3 treatment, 3 control; KEY – 3 treatment, 3 control; TP – 14 treatment, 15 control; CT – 1 treatment, 1 control; CP – 1 treatment, 1 control; SS – 2 treatment, 2 control. Sampling occurred at three structures/locations within each reach; the lowermost and uppermost in each reach as well as one randomly selected in the middle. Vegetation and arthropod samples were taken upstream of the structure/location as this is the intended area of impact.

Vegetation Sampling

We sampled the vegetation communities using a canopy cover approach using a 0.5 m^2 frame (USDA, 1999). At each sampling structure/location we randomly placed the frame twice;

once between 0 and 1 m upstream (hereafter, *0 m*) and once between 1.5 and 2.5 m upstream (hereafter, *2 m*). This provided 6 vegetation estimates per reach (n=1,296/year). Within each frame, we identified plants to the lowest taxonomic class feasible based on phenological stage during the growing season. For each species within a frame, we estimated the percent cover live, percent cover senesced, and total percent cover (% live + % senesced).

The primary sage-grouse plant food resources we observed at our catchments were *Achillea millefolium* (yarrow), *Agoseris* spp. (false dandelion), *Artemisia ludoviciana* (white sage), *Astragalus* spp. (milkvetch), *Erigeron glacialis* (streamside fleabane), *Phlox* spp. (phlox), *Taraxacum officinale* (common dandelion), and *Trifolium longipes* (long-stalked clover) (Table B.1). Of these, *Achillea millefolium*, *Artemesia ludoviciana*, *Erigeron glacialis*, and *Taraxacum officinale* were the dominant species. While increased moisture benefits all these species, *Artemesia ludoviciana*, *Trifolium longipes*, and *Erigeron glacialis* preferentially grow in meadows and riparian areas.

Arthropod Sampling and Processing

We quantified arthropod communities through pitfall sampling (Greenslade, 1964). Pitfall trapping took place at each catchment for one day, once per month, for the months of June through August, when arthropods are important in sage-grouse chick diet. Pitfall trapping primarily samples ground dwelling arthropods (Yi et al., 2012) which include the orders most common to sage-grouse chick diet: Coleoptera, Hymenoptera, Orthoptera, and Lepidoptera larva (Klebenow and Gray, 1968; Peterson, 1970; Drut et al., 1994b). While pitfall trapping is limited in that insect activity and density are confounded, it continues to be widely used (Kromp, 1999; Yi et al., 2012). In fact, activity-density measurements, rather than true density, may be a more meaningful metric of arthropod availability to ground-foraging insectivorous birds such as the sage-grouse (Goosey et al., 2019).

We placed a pitfall trap at a random location between 1.5 and 2.5 m upstream of each sampling structure/location resulting in a total of three pitfall traps set at each reach (n=486/year). Pitfall traps consisted of a 9.5 cm diameter x 12 cm deep cup placed in a hole with the surrounding space backfilled with dirt so that the top of each cup was flush with the soil surface (Greenslade, 1964). We removed vegetation within a 10 cm radius ring around the cup so as not to impede the movement of ground dwelling arthropods (Greenslade, 1964). Traps were filled one-third full with propylene glycol-based antifreeze to kill and preserve trapped arthropods (Dahlgren et al., 2010; Hohbein and Conway, 2018). Pitfall traps were set for 24 hours, after which arthropods were collected.

Upon return from the field, arthropod samples were frozen. We later thawed and separated arthropods by order. Due to their importance in sage-grouse chick diet, the order Coleoptera were identified to family and the order Hymenoptera were sorted into Formicidae and non-Formicidae (Peterson, 1969; Sutton, 2022). We counted and oven-dried arthropods at 60°C for 48 hours (Ganihar, 1997; Knapp, 2012). We then weighed samples to the nearest milligram. Values obtained from pitfall trapping were treated as indices of activity-density (number captured/reach/24-hour period) (Pavuk et al., 1997).

<u>Vegetation Analysis</u>

Based on research in Montana, Idaho, and Oregon, sage-grouse chicks in areas dominated by *Artemesia tridentata* select a diversity of forbs from the families Brassicaceae (mustards), Fabaceae (legumes), Polemoniaceae (phlox), Asteraceae (daisies), and Liliaceae (lilies)

(Klebenow and Gray, 1968; Peterson, 1970; Drut et al., 1994a). We developed models using total canopy coverage and live canopy coverage of specific genera known to be sage-grouse food resources as the response variables (Curran et al., 2015).

We modeled the response variables as a function of treatment (treated or control), month (June through September), year (2021 and 2022), and distance from structure (0 or 2 m) using linear mixed effects models. To account for the nested nature of our study design and the inherent variation among drainages, we included catchment, reach, and location within reach (upstream, middle, or downstream) as random variables. Canopy cover values were log-transformed to adjust for lack of normality and heteroskedasticity in the data.

Arthropod Analysis

The arthropods consumed by sage-grouse chicks are primarily from the orders Coleoptera, Orthoptera, Hymenoptera (Formicidae), and Lepidoptera (larva) (Klebenow and Gray, 1968; Peterson, 1970). We combined the activity density (total number of individual arthropods) for these groups in each reach for each visit and used this as a response variable. We also combined the dry biomass for these groups in each reach for each visit and used this as a second response variable. Both the activity density and dry biomass values were log-transformed to adjust for lack of normality and heteroskedasticity in the data.

We modeled each response variable as a function of treatment (treatment or control), month (June through August), and year (2021 and 2022) using linear mixed effects models. We included catchment as a random variable to account for repeated sampling and variation between drainages.

Results

Vegetation

We estimate a 27% (SE=8%) increase in the total canopy cover ($p=0.004$) and a 30% (SE=9%) increase in the live canopy cover ($p=0.003$) across both years (Figure 3-3; Figure 3-4; Table B.2; Table B.3). There was no treatment by year interaction ($p>0.10$) and no treatment by month interaction ($p>0.10$). However, canopy cover in 2022 was greater than it was in 2021 ($p<0.001$). There was no evidence of a difference in plant cover between the 0 m and 2 m sampling locations for treatment reaches or for control reaches in either year ($p>0.10$).

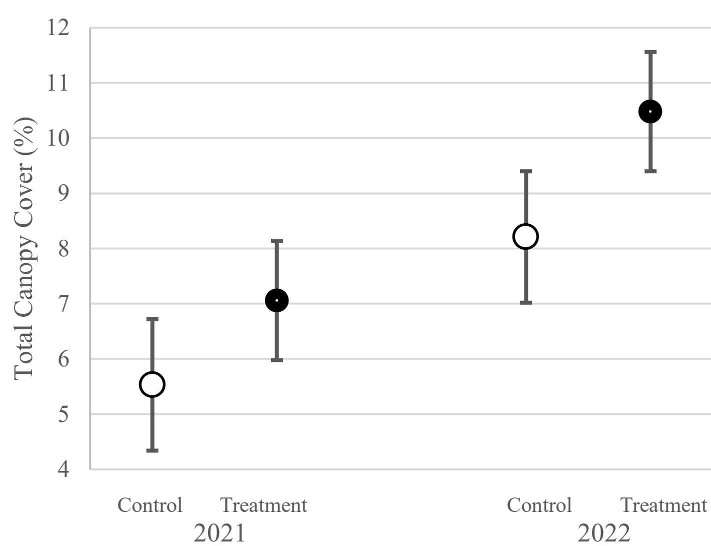

Figure 3-3: Total canopy cover (means +/- SE) of sage-grouse plant food resources at treated and control reaches across 7 catchments in 2021 and 2022 (n=54, 26 treatment, 28 control). Treated reaches had significantly higher total cover across both years ($p=0.004$). There were no treatment by year or treatment by month interactions ($p>0.10$). Treatment reaches are those where rock restoration structures were installed. Differences were tested using linear mixed effects models.

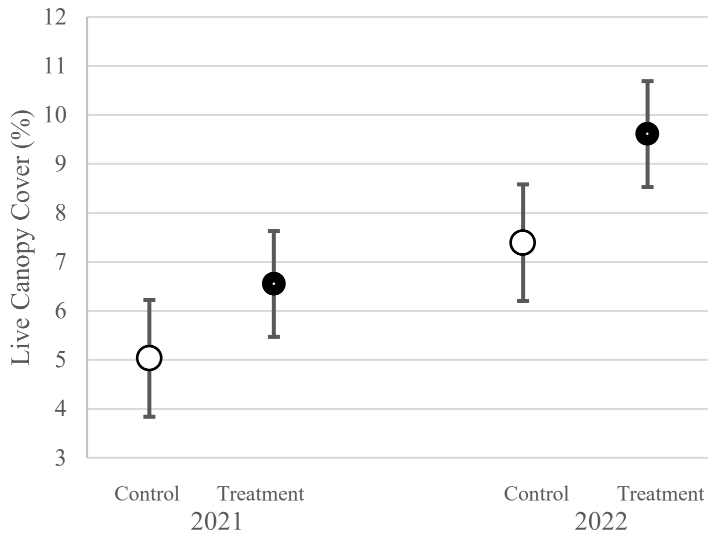

Figure 3-4: Live canopy cover (means +/- SE) of sage-grouse plant food resources at treated and control reaches across 7 catchments in 2021 and 2022 (n=54, 26 treatment, 28 control). Treated reaches had significantly higher live cover across both years (p=0.003). There were no treatment by year or treatment by month interactions (p>0.10). Treatment reaches are those where rock restoration structures were installed. Differences were tested using linear mixed effects models.

Arthropods

We did not detect differences between control and treatment reaches in arthropod food resources (activity density or biomass) for sage-grouse chicks (Table B.4; Table B.5). However, month (p=0.0126) and year (p=0.0016) had a significant impact on average activity density and there was an interaction between month and year (p=0.0131) (Figure 3-5; Table B.4). Average biomass per reach was higher in 2021 (0.47 g, 95% CI: 0.31 to 0.71 g) than it was in 2022 (0.15 g, 95% CI: 0.10 to 0.23 g) (p=0.0209) (Figure 3-6; Table B.5).

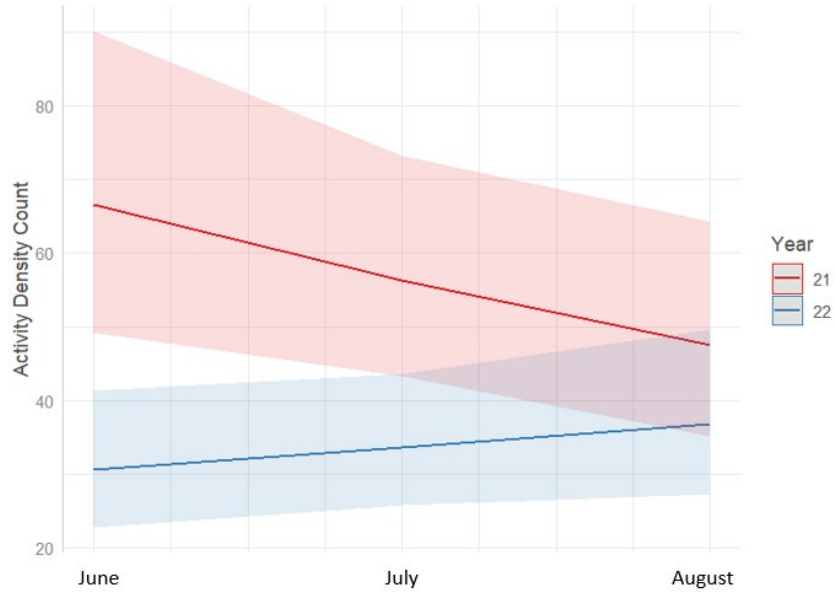

Figure 3-5: Activity density (means and 95% CIs) of sage-grouse arthropod food resources across 7 catchments in 2021 and 2022 (n=54). Both month (p=0.0126) and year (0.0016) significantly impacted activity density. There was also a month by year interaction (p=0.0131).

Discussion

Our results indicate that meadow restoration using low-tech rock structures in ephemeral drainages was an effective way to increase sage-grouse chick vegetation resources. Not only did treatment with restoration structures increase total percent cover but it also increased the percent cover of living plants. Since these metrics are highly correlated, we recommend only evaluating total cover in the future. Sutton (2022) evaluated the restoration sites 1-2 years after structures were installed while our study evaluates the restoration sites 2-3 years after structures were installed. Our results differ from those found by Sutton (2022) who detected no difference in sage-grouse plant food resources between treatment and control reaches. This may be because we added a second vegetation cover measurement closer to the structure. However, since we did not find a significant difference between our 0 m and 2 m frames, the difference in results is more likely because it takes time for restoration to have an effect. This delayed response in vegetation change agrees with preliminary results from research conducted in Colorado suggesting that generally it takes 3 years to see a response to structure installation (TNC, 2017). It also indicates that further change may occur as more time passes.

It is worth noting that each restoration structure has a relatively small area of impact due to the size of the gullies and the limited water flow through the ephemeral stream. While we did observe higher canopy cover at treated reaches, overall percent cover of sage-grouse chick plant resources is relatively low with a median cover of 7% at control reaches. This means, as an example, a 27% increase of 7% cover is an increase of 2% cover. While this may be a small increase when looking at the impacted area of one structure, it could be a large amount of vegetation if experienced across an entire drainage. This emphasizes that to have a meaningful impact on sage-grouse food resources, rock restoration structures need to be implemented in a series, throughout a drainage, and across an entire watershed.

It is difficult to make direct comparisons between our findings and other studies investigating rock restoration structures due to differing sampling methods. Other studies looking at vegetation responses to similar restoration techniques have looked at coarser metrics using satellite imagery, but there are no other studies that look at specifically increasing sage-grouse resources or any other targeted plant communities with rock structures. However, it is likely that vegetation cover (discussed in more detail by Lowing (2023)) is correlated with sage-grouse food resource cover. If this is true, then our findings are similar to the 24% increase in vegetation productivity post-restoration in the Gunnison Basin in Colorado using NDVI imagery (Silverman et al., 2019). In the future, it could be interesting to use a combined approach that integrates

NDVI imagery with species specific surveys to directly relate changes that have been observed using these different methods.

We observed higher vegetation resource cover in 2021 versus 2022. This is likely due to 2021 having lower overall precipitation and thus water availability. In 2021, Jefferson Basin, the watershed that contains our study catchments, received 79% of normal annual precipitation. In 2022, Jefferson Basin received 99% of normal annual precipitation (USDA, 2022). Despite this difference, treatment was impactful regardless of amount of annual precipitation since there was no interaction between treatment and year. In addition, the treatment was effective throughout the entire summer since there was no interaction between month and treatment. This supports the use of rock structures in the face of a changing climate as it is an indication that they will remain effective in the below average precipitation years that will inevitably become more frequent. While we were specifically investigating food resources for sage-grouse, the increase in plant food sources at treated reaches is an indication that mesic vegetation resources for other wildlife species may also be affected.

While we did detect a difference in the plants important to sage-grouse chick diet, we did not detect a difference in the abundance or availability of arthropods important to sage-grouse chick diet between treatment and control reaches. Sutton (2022) also did not find any differences in the biomass of arthropods in 2019 and 2020. This may be because individual responses to restoration occur at different rates. Arthropod communities are highly influenced by vegetation communities and not exclusively soil moisture (Wenninger and Inouye, 2008). Since Sutton (2022) did not find any changes in vegetation cover or community composition in 2019 and 2021, it is likely that more time is needed to see an arthropod response to the vegetation changes we observed in 2021 and 2022. Interestingly, we found a decrease in arthropod activity density and biomass from 2021 to 2022 which is the opposite of the vegetation responses. However, there may have been changes to the arthropod community tied to specific families or species that have unique habitat requirements that we did not capture due to our focus on sage-grouse chick food resources and only identifying arthropods to order (Yu et al., 2006; Stamps et al., 2009; Borchard et al., 2013).

Other low-tech restoration methods are also being implemented in this region and provide opportunities for comparison within the sagebrush steppe of Southwest Montana. For instance, beaver dam analogs (BDAs) were installed in a creek near our study areas between Little Basin Tributary 7 and Teepee Creek (Askam et al., 2022). BDAs are another low-tech restoration technique with a similar premise to the structures used in our research but are typically implemented in perennial streams and rivers that have more water flow than the systems where one-rock dams are used. BDAs can be highly effective, but in this case, four years post treatment, they did not find an increase in late-season vegetation in locations where BDAs were installed. Askam et al. (2022) suggest this may be because vegetation was not water limited prior to BDA installation. This implies that there may be more opportunities for improving vegetation in this region where water is more limited, such as the ephemeral drainages where our study took place.

Initial modeling completed in this study did not detect differences in treatment effects between catchments despite them spanning a gradient of environmental and abiotic conditions. This indicates that rock structures are potentially suited for most ephemeral drainages that have similar characteristics to those in this study (elevation: 2,000-2,250 m, aspect: north or south facing, slope: 1-10%, soil type: clay and sandy loams, and drainage incision: 5.5 to 14.6 width to depth ratio). Further investigation into our data may reveal insights into mechanisms that are driving vegetation productivity in mesic meadow systems. By tying in reach specific data regarding soil characteristics, topography, underlying geology, monthly precipitation, elevation, and average temperatures, we hope to better understand where rock restoration structures may be most effective. Despite this, our research shows that this low-tech restoration technique is one tool we can use to help create resilient landscapes in the face of climate change.

CHAPTER FOUR

CONCLUSIONS

Maintaining the hydrologic and ecological function of mesic sites is important when thinking about a changing future climate. One way to provide this is through meadow restoration and maintenance. In degraded landscapes across the semi-arid west, practitioners are using low-tech methods in attempts to restore degraded streams and meadows. Despite their broad application, the effects of these methods are notoriously under-researched due to a lack of resources and difficulty in implementing meaningful experimental designs (Silverman et al., 2019). Our results indicate one method of restoration, constructing small rock structures in ephemeral drainages, may improve the water storage and wildlife resources that meadows provide. Since soil moisture is a main driver of meadow vegetative communities, it is possible that the increase residence time after rainfall events we identified in Chapter One is promoting the increase in cover of plants that are food resources for sage-grouse chicks we identified in Chapter Two. This is valuable as soil moisture has been shown to moderate the effects of climate warming on the spring phenology of grasslands (Liu et al., 2022) and may be able to moderate the effects of climate warming on the loss of moisture-sensitive forbs.

While we did find a measurable difference in soil moisture and vegetation components in the areas surrounding rock structures, the differences were relatively small. Many of the metrics we measured such as seasonal soil moisture and arthropod availability showed no difference between treated and control reaches. This is important information for land managers looking for ways to restore meadows. Low-tech rock structures have promise for improving mesic meadows, but they are not an instant fix; recovery takes time. Due to the sedimentation and pooling we observed behind structures, it is possible that in future years there will be evidence of a larger change in the landscape due to the rock restoration structures. This is supported by the differences detected between treatment and control in years 3-4 post-installation that were not detected by Sutton (2022) in years 1-2 post-installation. In addition, variation in the time needed to detect effects may be greatly influenced by environmental conditions. In our study, drought years in 2020 and 2021 may have slowed the response time as some of the streams did not flow during spring snowmelt in 2021.

Although not the target question of this research, environmental factors can influence the efficacy of rock restoration structures. Locations for implementation should be selected wisely since local environmental conditions may influence low-tech restoration techniques. Our study catchments represented a diverse array of abiotic factors, but overall they were very similar in environmental conditions. This limits our ability to diagnose locations where rock structures may be most effective as the difference in treatment effect between catchments is likely to be small. Further exploration of what is driving components of mesic meadow systems by looking at more catchments and catchments that span a more diverse spread of abiotic factors would be valuable. This knowledge could help practitioners understand where rock structures are most useful and impactful. In locations where rock structures are potentially less effective, there may be other methods they can be combined with to have a larger impact.

In this study, we selected monitoring methods that targeted soil moisture and ecological resources tied to a specific species, the sage-grouse. There are other methods that may be more fitting for post-treatment monitoring depending on the extent and goals of the restoration project. These may include looking at pre-and-post NDVI imagery, measuring vegetation biomass, or quantifying changes in channel morphology. In the future, it would be interesting to look specifically at facultative mesic species that may be more closely associated and affected by changes in soil moisture. Focusing on a finer taxonomic resolution of arthropods could also reveal responses to restoration that were not detected in the coarse, family-level approach we took in this study.

The importance of high-elevation meadows to our watersheds, combined with the threats of climate change and human-caused habitat degradation, suggest that the restoration of wet meadows in arid landscapes should be a priority issue (Knight, 2014; Silverman et al., 2019). The results from our study provide valuable information to land managers looking for ways to enhance the quality of sagebrush rangelands and improve the extent and resilience of mesic habitats. The low cost and relative speed of installing rock structures provides a promising method for improving wet meadows across a vast landscape. Our results show that rock structures are beneficial in sagebrush steppe ephemeral drainages that have similar characteristics to those in this study (elevation: 2,000-2,250 m, aspect: north or south facing, slope: 1-10%, soil type: clay and sandy loams, and drainage incision: 5.5 to 14.6 width to depth ratio). When implemented as a series within a drainage and across an entire watershed, this restoration technique can have a meaningful impact on water retention time and wildlife vegetation resources at a landscape scale.

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APPENDICES

APPENDIX A

SUPPLEMENTARY TABLES FOR CHAPTER TWO

Table A.1: Estimates (and standard errors, test statistics, and P-values) from linear mixed model comparing average soil moisture across all catchments. Soil moisture probes were placed at 6 reaches within each catchment and at 0 m, 1 m, and 3 m from the gully center.

Fixed Effects	Estimated Coefficient	SE	df	t value	P value
(Intercept)	0.7357	0.03928	2.739e+01	18.731	<0.001
Treatment	-0.01617	0.01115	1.098e+01	-1.450	0.175
Month	-0.05446	0.003980	6.608e+03	-13.683	<0.001
2020	-0.1360	0.03318	6.608e+03	-4.098	<0.001
2021	-0.3245	0.03359	6.608e+03	-9.661	<0.001
2022	-0.02937	0.03353	6.608e+03	-0.876	0.381
Month by 2020	0.004333	0.004188	6.609e+03	1.035	0.301
Month by 2021	0.03427	0.004265	6.609e+03	8.035	<0.001
Month by 2022	0.005867	0.004256	6.609e+03	-1.378	0.168

Random effects	Variance	SD
Distance	0.0009740	0.03121
Reach	0.0004040	0.02010
Pair	0.0004144	0.02036
Catchment	0.0017766	0.04215
Residual	0.0044331	0.06658

(Number of observations: 6686; 4 catchments; 12 pairs of reaches; 24 reaches; 3 distances)

Table A.2: Estimates (and standard errors, test statistics, and P-values) from linear mixed model comparing drying rates of individual reaches. Soil moisture probes were placed at 6 reaches within each catchment and at 0 m, 1 m, and 3 m from the gully center.

Fixed Effects	Estimated Coefficient	SE	df	t value	P value
(Intercept)	-0.23256	0.03199	3.39467	-7.269	0.0035
Treatment	-0.03171	0.02318	139.00462	-1.368	0.1736
1 m	0.03547	0.02393	132.21632	1.482	0.1406
3 m	0.05027	0.02393	132.82452	2.101	0.0375
Treatment by 1 m	0.04261	0.03299	132.91510	1.292	0.1987
Treatment by 3 m	0.03413	0.03301	133.81237	1.034	0.3029

Random effects	Variance	SD
Pair	0.001464	0.03827
Catchment	0.003413	0.05842
Year	0.001234	0.03513
Residual	0.007671	0.08759

(Number of observations: 169; 4 catchments; 12 pairs of reaches; 24 reaches; 3 distances)

Table A.3: Estimates from cumulative link mixed model comparing odds of an increase (I), stabilization (S), or decrease (D) after a rainfall event at treatment and control reaches. Soil moisture probes were placed at 6 reaches within each catchment and at 0 m, 1 m, and 3 m from the gully center.

Fixed Effects	Estimated Coefficient	SE	z value	p
Treatment	1.0929	0.3253	3.359	0.0008
Random effects	Variance	SD		
Catchment	0.000	0.000		
Event	6.4317	6.102		
Pair	2.9865	1.7282		
Distance	0.9616	0.9806		
Residual	20.19	4.494		
Threshold	Estimate	SE	z value	
D|S	-6.3026	0.9674	-6.515	
S|T	-2.1205	0.6475	-3.275	

(Number of observations: 300; 3 distances from gully center; 12 pairs of reaches; 36 events; 4 catchments)

Table A.4: Estimates (and standard errors, test statistics, and P-values) from linear mixed model comparing length of soil moisture increase across all catchments. Rainfall events were identified by looking for increases in soil moisture that occurred at more than one reach.

Fixed Effects	Estimated Coefficient	SE	df	t value	P value
(Intercept)	15.7460	3.0650	27.2111	5.137	<0.001
Treatment	1.6731	0.6115	165.2459	2.736	0.0069
Little Basin 24	-1.7407	3.9614	25.1572	-0.439	0.6641
Little Basin 7	-6.9385	4.5126	26.4782	-1.538	0.1360
Teepee	-1.2459	3.9269	24.3189	-0.317	0.7538
1 m	-0.4565	0.7876	173.3137	-0.583	0.5604
3 m	-0.3017	0.8233	176.2153	-0.366	0.7145
Random effects	Variance	SD			
Pair	17.55	4.190			
Event	37.23	6.102			
Residual	20.19	4.494			

(Number of observations: 216; 12 pairs of reaches; 27 events)

APPENDIX B

SUPPLEMENTARY TABLES FOR CHAPTER THREE

Table B.1: Plant genera/species considered sage grouse food resources along with their frequency of occurrence (number of frames they appeared in).

Plant	**Frequency**	**Plant**	**Frequency**
Achillea millefolium	406	Mertensia oblongifolia	19
Agoseris spp.	154	Medicago sativa	36
Antennaria spp.	74	Phlox spp.	188
Artemisia ludoviciana	396	Ranunculus abortivus	2
Astragalous spp.	107	Ranunculus acris	13
Brassicaceae spp.	50	Ranunculus glaberrimus	46
Collinsia parviflora	8	Ranunculus eschscholtzii	46
Erigeron glabellus	503	Taraxacum officinale	772
Eriogonum heracleodies	6	Tragopogon dubious	24
Delphinium occidentale	2	Trifolium spp.	159
Lomatium spp.	1		

Table B.2: Estimates (and standard errors, test statistics, and P-values) from linear mixed model comparing log total canopy cover of sage grouse plant food resources between treated reaches and control reaches in Southwest Montana. Treatment reaches are those where rock restoration structures were installed.

Fixed Effects	Estimated Coefficient	SE	df	t value	P value
(Intercept)	-0.35520	0.39715	195.2	-0.894	0.37221
Treatment	0.24435	0.08237	41.78	2.967	0.00496
Distance from Structure (2 m)	-0.02029	0.04892	1423	-0.415	0.67833
Month	0.22522	0.04801	1424	4.691	<0.001
Year	1.71164	0.23253	1420	7.361	<0.001
Treatment by 2 m	-0.04015	0.07046	1421	-0.570	0.56890
Month by Year	-0.17757	0.03104	1424	-5.721	<0.001
Random effects	Variance	SD			
Structure Location	0.06004	0.2450			
Reach	0.03045	0.1745			
Pair	0.05055	0.2248			
Catchment	0.15024	0.3876			
Residual	0.46217	0.6798			

(Number of observations: 1535; 7 catchments; 28 pairs of reaches; 54 reaches; 2 distances)

Table B.3: Estimates (and standard errors, test statistics, and P-values) from linear mixed model comparing log live canopy cover of sage grouse plant food resources between treated reaches and control reaches in Southwest Montana. Treatment reaches are those where rock restoration structures were installed.

Fixed Effects	Estimated Coefficient	SE	df	t value	P value
(Intercept)	-0.35915	0.40138	257.316	-0.895	0.3717
Treatment	0.26271	0.08358	42.45	3.143	0.0031
Distance from Structure (2 m)	-0.03767	0.05018	1381	-0.751	0.4529
Month	0.21561	0.04941	1387	4.364	<0.001
Year	1.99615	0.23866	1384	8.364	<0.001
Treatment by 2 m	-0.02113	0.07237	1381	-0.292	0.7704
Month by Year	-0.21841	0.03198	1387	-6.831	<0.001

Random effects	Variance	SD
Structure Location	0.06209	0.2492
Reach	0.03021	0.1738
Pair	0.04184	0.2045
Catchment	0.13090	0.3618
Residual	0.47460	0.6889

(Number of observations: 1495; 7 catchments; 28 pairs of reaches; 54 reaches; 2 distances)

Table B.4: Estimates (and standard errors, test statistics, and P-values) from linear mixed model comparing log activity density of sage grouse arthropod food resources between treated reaches and control reaches in Southwest Montana. Treatment reaches are those where rock restoration structures were installed.

Fixed Effects	Estimated Coefficient	SE	df	t value	P value
(Intercept)	54.4138	15.8379	258.6488	3.436	0.0007
Treatment	-0.2405	0.1706	50.6593	-1.410	0.1647
Month	-5.6449	2.2460	258.5690	-2.513	0.0126
Year	-2.3429	0.7361	258.5690	-3.183	0.0016
Month by Year	0.2608	0.1044	258.5690	2.498	0.0131

Random effects	Variance	SD
Reach	0.294236	0.54344
Catchment	0.007278	0.08531
Residual	0.566423	0.75261

(Number of observations: 314; 7 catchments; 54 reaches)

Table B.5: Estimates (and standard errors, test statistics, and P-values) from linear mixed model comparing log biomass of sage grouse arthropod food resources between treated reaches and control reaches in Southwest Montana. Treatment reaches are those where rock restoration structures were installed.

Fixed Effects	Estimated Coefficient	SE	df	t value	P value
(Intercept)	4.2577	2.1307	265.1992	1.998	0.0467
Treatment	-0.1780	0.1812	47.7355	-0.982	0.3309
Month	-0.5532	0.3012	261.3050	-1.837	0.0647
Year	-3.1068	1.3369	260.8109	-2.324	0.0209
Month by Year	0.2815	0.1896	260.7010	1.484	0.1389

Random effects	Variance	SD
Reach	0.1191	0.3451
Catchment	0.1163	0.3410
Residual	1.8743	1.3691

(Number of observations: 314; 7 catchments; 54 reaches)

Table B.6: Characteristics (distance, slope, width to depth ratio, and aspect) of each reach that was sampled during 2021 and 2022. These covariates were used in the initial model selection process. Experimental unit is formatted with the catchment name followed by the reach identifier. Treatment reaches are those where rock restoration structures were installed in Southwest Montana.

Reach	Treatment	Distance from Top Reach	Gully Slope	Width to Depth Ratio	Aspect of Right Bank
CloverPassTrib_3	Treatment	0	1	35	160
CloverPassTrib_4	Control	32	2	18	174
CloverTrib1_1	Treatment	0	3	9	196
CloverTrib1_2	Control	152	3	17	117
KeystoneTrib1_1	Control	0	2	7	145
KeystoneTrib1_A	Treatment	408	2	7	99
KeystoneTrib1_3	Control	371	2	10	76
KeystoneTrib1_B	Treatment	478	1	6	50
KeystoneTrib1_C	Treatment	986	6	7	85
KeystoneTrib1_6	Control	830	1	10	76
LBTrib7_1	Treatment	0	3	9	137
LBTrib7_2	Control	48	2	12	113
LBTrib7_3	Treatment	96	1	10	40
LBTrib7_4	Control	150	1	7	16
LBTrib7_5	Treatment	577	2	13	35

LBTrib7_6	Control	964	3	10	120
Teepee_1	Control	0	0	12	210
Teepee_2	Treatment	50	0	11	313
Teepee_3	Control	455	0	47	249
Teepee_4	Treatment	564	1	31	299
Teepee_5	Control	642	0	24	253
Teepee_6	Treatment	706	0	37	344
Teepee_7	Control	734	0	17	312
Teepee_8	Treatment	844	0	27	291
Teepee_9	Control	911	0	22	153
Teepee_10	Treatment	974	0	17	204
Teepee_11	Control	1080	2	9	228
Teepee_12	Treatment	1142	2	7	225
Teepee_13	Control	1196	1	7	209
Teepee_14	Treatment	1268	1	8	311
Teepee_15	Control	1328	1	9	205
Teepee_16	Treatment	1388	1	10	229
Teepee_17	Control	1439	2	8	175
Teepee_18	Treatment	1598	1	12	236
Teepee_19	Control	1655	1	7	246
Teepee_20	Treatment	1704	1	9	287
Teepee_21	Control	1756	1	12	129
Teepee_22	Treatment	1810	1	11	186
Teepee_23	Control	1979	2	12	178
Teepee_24	Treatment	2055	2	9	260
Teepee_25	Control	2134	2	9	203
Teepee_26	Treatment	2210	2	12	199
Teepee_27	Control	2284	2	9	222
Teepee_28	Treatment	2351	1	8	310
Teepee_29	Control	2431	2	9	239
Snowshoe_1	Control	0	1	9	233
Snowshoe_2	Treatment	24	2	8	177
Snowshoe_3	Control	77	2	8	157
Snowshoe_4	Treatment	98	2	8	188
LBSect24_1	Control	0	5	5	112
LBSect24_2	Treatment	61	6	6	127
LBSect24_3	Control	97	5	6	113
LBSect24_4	Treatment	183	4	6	115
LBSect24_6	Control	871	4	5	122