INITIAL EFFECTS OF LOW-TECH RESTORATION OF WET MEADOWS IN SAGEBRUSH STEPPE

by

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

of

Master of Science

in

Fish and Wildlife Management

MONTANA STATE UNIVERSITY
Bozeman, Montana

May 2022

ACKNOWLEDGEMENTS

I would like to thank my advisor, Dr. Andrea Litt for her mentorship and guidance through graduate school and before. I would also like to thank my committee members Bok Sowell and Hayes Goosey. Bok helped develop the project and was a vital resource for guidance. Hayes was instrumental in pushing me to develop the knowledge of arthropod identification and ecology I needed to complete my research. I am thankful to Kyle Cutting for providing mentorship while in the field, and the rest of the Fish and Wildlife Service staff at Red Rock Lakes National Wildlife Refuge for helping me in too many ways to list. I would like to thank Sean Claffey for planning, designing, and implementing the restoration structures I studied, as well as the Montana Conservation Corps crews for their hard work in building the structures. This project would have been impossible without my technicians, Nick Lowing, Laura Robison, and Ethan Palen. I would like to thank the Litt Lab for their support and countless hours of editing my writing. I would especially like to thank Shane Petch for always seeming to have time to help me through all my data-wrangling woes. I am grateful for the support of my family; especially my wife, Tia, for her endless love and support that has pushed me to follow my dreams of becoming a wildlife professional. My work was supported by grants from the United States Fish and Wildlife Service, The Nature Conservancy, and Western Sustainable Agriculture Research and Education. We also respectfully acknowledge this land was once a crossroads for many nations and that many peoples used this land as a peaceful resting place long before European settlers forced them from their homes. We acknowledge the pain this causes and hope our research and stewardship honors their land, their memory, and their descendants.

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ABSTRACT

In semi-arid environments, wet meadows are important sources of late-season palatable vegetation for many wildlife species; these areas often support higher coverage and diversity of plants relative to surrounding upland environments. In the sagebrush steppe of southwest Montana, wet meadows are fed by melting snowpack. Due to climate change and land use practices, the duration and amount of moisture wet meadows receive is declining. To mitigate these changes, low-tech restoration structures, such as primitive rock dams, have been installed in six different drainages across southwest Montana. Similar structures have been studied in Colorado, where they found immediate increases in plant productivity. We used these structures within an experimental framework to compare soil moisture, vegetation structure, and vegetation composition (Chapter Two), as well as known food resources (both plants and arthropods) for sage grouse chicks and nesting sage thrashers, Brewer's sparrows, and vesper sparrows (Chapter Three) one and two years after restoration. We measured soil moisture and plant canopy coverage, as well as food resources for the focal birds during the summers of 2021 and 2022. We did not detect differences between treated and control areas in soil moisture, vegetation structure, or vegetation composition during any sampling period; however, many of our estimates for vegetation structure and composition were higher in treated than control areas two years after treatment. We also did not detect differences in plant or arthropod food resources for sage grouse chicks, nesting sage thrashers, or nesting Brewer's sparrows during any sampling period. We did find higher coverage of known plant foods for vesper sparrows in treated areas, compared to controls, during September, two years after treatment; this increase was mainly driven by Kentucky bluegrass. Given the cold climate of our study sites, more time may be needed before we can detect changes resulting from the restoration structures. Even if these low-tech solutions do not provide a "cure-all" for wet meadow restoration, changes in climate and land-use practices emphasize the continued need to find effective and practical tools to restore wet meadows in arid landscapes.

CHAPTER ONE

INTRODUCTION TO THESIS

Humans have long altered their environments to meet societal needs (Greipsson 2011), often converting habitat for other species into agricultural lands (Greipsson 2011). This conversion, has led to habitat loss and extinction of many species (Greipsson 2011). At current rates, an estimated one-third to two-thirds of all species will go extinct in the next 50 years (Greipsson 2011), and habitat loss is one of the biggest threats to species persistence (Dobson et al. 1997).

Wetlands, areas that are seasonally or permanently saturated with water (US EPA 2021), provide an important habitat component for many species (Gibbs 2000). Although wetlands cover <3% of the world's land, 30% of all plant species dwell in them and 75% of native animals rely on them at some point during the year (Knight et al. 2014; Greipsson 2011). Yet, this important ecosystem currently covers less than 50% of its historical area (Johnston 1994; Greipsson 2011). Wet meadows are a type of wetland that are saturated by water in spring and early summer, but typically dry later in the growing season (Knight et al. 2014). Even after drying, wet meadows still retain intermediate amounts of soil moisture, making them important oases in semi-arid landscapes (Knight et al. 2014; Naiman et al. 2010). Wet meadows are currently facing significant challenges in the face of climate change and many agricultural land use practices (Wenninger & Inouye 2008; Gilbert 2011; Knight et al. 2014).

In the arid Rocky Mountain west, wet meadows are often fed by temporary streams (Seager & Vecchi 2010). These streams are fed by melting high elevation snowpack (Seager & Vecchi 2010). However, climate change has resulted in reduced snowpack and a shift in phenology, such that these streams run earlier in the spring (Stewart et al. 2004), exacerbated by land use practices (Knight et al. 2014). Wet meadows typically support more plant growth than the adjacent upland environments (Knight et al. 2014), making wet meadows susceptible to overgrazing, reducing the soil-stabilizing vegetation that eventually leads to changes in water flow (Knight et al. 2014). Altered flow can result in stream channelization, subsequent reductions in water tables, and a shift towards plants drying out sooner and becoming less nutritious late in the growing season (Loheide & Gorelick 2007; Donnelly et al. 2016). As wet meadows decrease in abundance and area, animals that rely on these oases for green vegetation and water must travel farther to meet their habitat needs (Loheide & Gorelick 2007; Knight et al. 2014; Donnelly et al. 2016; Zeedyk & Clothier 2009).

Restoring wet meadows in semi-arid ecosystems may be possible using simple or low-tech solutions (Zeedyk & Clothier 2009). Low-tech restoration in wet meadows entails using natural materials, such as logs and rocks, to create dams or baffles that slow water movement and retain moisture in impaired areas. These methods are promising, but to our knowledge, only one study has examined the efficacy of these treatments, and focused on changes in vegetation productivity measured using satellite imagery (Silverman et al. 2019). When low-tech restoration structures were present, vegetation productivity increased 24% over six years (Silverman et al. 2019); much of this increase took place within one year of treatment. Although encouraging, such coarse measures of vegetation do not provide detailed inferences, such as specific changes in vegetation structure or composition, or abundance of food resources (i.e., plants and arthropods) for vertebrate species. As such, we sought to build on previous work, by comparing areas with and without low-tech restoration structures, within impaired wet meadows in sagebrush communities in Montana.

We sought to provide insights about the short-term and small-scale effects of wet-meadow restoration in sagebrush environments. In Chapter 2, we investigated how vegetation structure and composition, as well as soil moisture, changed in areas where restoration had occurred. In Chapter 3, we investigated potential changes in known food resources of greater sage-grouse (*Centrocercus urophasianus*), sage thrashers (*Oreoscoptes montanus*), Brewer's sparrows (*Spizella breweri*), and vesper sparrows (*Pooecetes gramineus*) after wet meadow restoration. In both chapters, we compared treated (areas with wet meadow restoration) and control (areas that could have been treated but were not) areas. Our results provide information about how soil moisture, vegetation, and sagebrush associated vertebrate species habitat quality changes after implementation that may be helpful to land managers planning to implement wet meadow restoration.

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CHAPTER TWO

PLANT COMMUNITY RESPONSE TO WET MEADOW RESTORATION IN SOUTHWEST MONTANA

Contribution of Authors and Co-Authors

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Contributions: Helped develop initial study idea, guided study design, assisted in securing funding, helped with data analysis, thoroughly edited manuscript

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Manuscript Information

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Restoration Ecology

Status of Manuscript:
_X__ Prepared for submission to a peer-reviewed journal
____ Officially submitted to a peer-reviewed journal
____ Accepted by a peer-reviewed journal
____ Published in a peer-reviewed journal

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Plant Community Response to Wet Meadow Restoration in Southwest Montana

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ABSTRACT

Wet meadows are important oases of palatable vegetation for many animals in semi-arid regions. These wet meadows are often fed by melting snowpack. Climatic changes can reduce snowpack, as well as initiate melting earlier in the spring. Land-use practices, such as grazing may exacerbate these changes, with implications for the abundance and size of wet meadows. Restoring the function and extent of wet meadows may be possible with simple or low-tech solutions, using natural materials (e.g., rocks) to slow water movement and retain moisture. In southwestern Montana, we used these structures within an experimental framework to compare soil moisture, vegetation structure, and vegetation composition one and two years after restoration. We did not detect differences between treated and control reaches for any response variable during any sampling period. However, despite uncertainty, canopy coverage often was higher in treated areas two years after restoration, suggesting that we may need more time to reliably detect effects. Given that our work occurred in a colder climate, with a shorter growing season than comparable studies, it may be reasonable to expect delayed effects from restoration. In addition, we were able to visually see structures slowing water, which is another promising sign that these structures eventually may be able to measurably increase soil moisture in a changing climate.

Key words: mesic restoration, low-tech restoration, Zeedyk structures, one-rock-dams, rangelands

IMPLICATIONS FOR PRACTICE

- Local environmental conditions may be an important influence on efficacy of low-tech wet meadow restoration
- Delayed effects of wet meadow restoration may be possible in colder environments with prolonged periods of freezing temperatures

MAIN TEXT

Introduction

Water is a crucial and scarce resource throughout much of the world (Knight et al. 2014; McKinstry et al. 2004). Wetlands, areas that are seasonally or permanently saturated with water (US EPA 2021), act as a link to water for many terrestrial species of plants and animals (Malanson 1993). Although wetlands cover <3% of the world's land area, 75% of native animals rely on these areas at some point during the year (Knight et al. 2014). Wet meadows are a type of wetland that is saturated by water in spring and early summer, but typically dry out later in the growing season (Knight et al. 2014). Even after drying, these meadows still may retain intermediate amounts of soil moisture, making them important oases in semi-arid landscapes (Knight et al. 2014; Naiman et al. 2010)

In the arid Rocky Mountain west, wet meadows often are fed by seasonal streams (Knight et al. 2014) and melting snow provides the primary source of water (Seager & Vecchi 2010). When functioning properly, wet meadows have high water tables, relative to the surrounding landscape, and tend to support diverse plant and animal communities (Mathewson et al. 2013; Knight et al. 2014). However, wet meadows are threatened by changes in climate patterns and land use (Gilbert 2011; Wenninger & Inouye 2008; Knight et al. 2014).

Climate warming can reduce snowpack at high elevations, as well as initiate melting of snow earlier in the spring (Seager & Vecchi 2010). For example, many rivers and ephemeral streams fed by snowmelt in the Rocky Mountains are expected to run at maximum capacity 30 to 40 days earlier by 2099 (Stewart et al. 2004). These phenological changes often are exacerbated by land use practices, such as grazing (Knight et al. 2014). Wet meadows are susceptible to overgrazing because they often support more plant growth than the surrounding upland vegetation (Knight et al. 2014). Overgrazing reduces vegetation that stabilizes soils, eventually leading to changes in water flow (Knight et al. 2014). Altered flow can result in channelization, lowered water tables, and a shift towards drying earlier and becoming less nutritious late in the growing season. Animals that need water and green vegetation may need to move longer distances to find these resources (Zeedyk & Clothier 2009; Loheide & Gorelick 2007; Donnelly et al. 2016; Knight et al. 2014). All of these changes elevate the need to restore function, by retaining moisture in wet meadows for longer periods (Stewart et al. 2004; Seager & Vecchi 2010; Silverman et al. 2019).

We may be able to restore function and increase the number and extent of wet meadows within semi-arid ecosystems using simple or low-tech restoration solutions (Zeedyk & Clothier 2009; Silverman et al. 2019). Managers can use natural materials, such as logs and rocks, to create dams or baffles that slow water movement and retain moisture in impaired areas. These methods, typically, one or some combination of three strategies: lateral control, vertical control, and plug and spread (Zeedyk & Clothier 2009; Zeedyk 2015), could help reduce the influence of climatic variability and land-use practices (Donnelly et al. 2016).

Low-tech restoration of wet meadows is new and promising, yet largely unstudied (Silverman et al. 2019). To our knowledge, only one study has examined the efficacy of these restoration methods (Silverman et al. 2019). In that study, vegetation productivity, as measured via satellite imagery, increased 24% six years after adding water-slowing structures (TNC & GCWG 2017; Zeedyk & Clothier 2009). Most of these structures were one-rock dams (Figure 1), or dams consisting of one layer of rocks across the width of the gully, one form of vertical stream control (Silverman et al. 2019; Zeedyk & Clothier 2009). Although these results are encouraging, these coarse measures of vegetation do not provide detailed inferences, such as the specific effects on species or functional groups of native plants or noxious weeds (Silverman et al. 2019).

In southwestern Montana, numerous one-rock dams and brush dams have been constructed within impaired wet meadows in sagebrush communities, with the goal of restoring function. One-rock and brush dams are constructed in a similar fashion, but the latter are made by stacking brush or tree branches in the channel bottom perpendicular to the flow of water (Zeedyk & Jansens 2006). We used these structures within an experimental framework (treated and control areas) to build on our knowledge about the restoration efficacy of these structures by quantifying potential changes in soil moisture, as well as vegetation structure and composition.

We expected that soil moisture would be comparable between treated and control areas early in the growing season (June), when most soils are saturated from snow melt, but that soil moisture would be better retained in treated areas late in the growing season (August and September). Species richness and diversity of plants tend to be relatively high in mesic meadows, especially cover of graminoids and forbs (Wallestad 1975; Debinski et al. 2000), given more available water from higher water tables. As such, we expected higher species richness, diversity, and more even distributions of plant species in treated areas, relative to controls. We also expected to find more cover of graminoids and forbs and less woody vegetation in treated areas than in untreated controls. Finally, we predicted that treated areas would support more green vegetation relative to senesced (brown) vegetation late in the growing season, compared to controls.

Methods

Site Background

Our work focused on wet meadows located around channels carved by snow melt in the sagebrush steppe of southwest Montana, in the general area of Red Rock Lakes National Wildlife Refuge (Red Rock Lakes hereafter). Wet meadows cover over 2800 ha of Red Rock Lakes (USWS 2009). The climate is characterized by long, cold winters and short summers with variable annual precipitation (USFWS 2009). Average annual temperature is 1.6° C and average annual precipitation is 50 cm (USFWS 2009), with over 60% of the water entering the wet meadows coming from snowmelt during the spring (Serreze et al. 1999).

Our research occurred in six drainages (Little Basin Tributary 1, Little Tributary 2, Teepee Creek, Clover Tributary, Keystone Gulch, and Snowshoe Creek, Figure 2), which are spread out over more than 2000 km^2. These drainages have different characteristics, but areas within a drainage are similar. Soils are primarily sandy with large particles and low water-holding potential (Teepee Creek), clayey with high water-holding potential (Keystone Gulch), or intermediate with roughly 50% sand and 50% clay (Little Basin Tributary 1, Little Basin Tributary 2, and Clover Tributary; California Soil Resource Lab 2021). Elevation ranged from 2000 to 2300 m and slope gradients ranged from 0% (Teepee Creek) to 11% (Little Basin Tributary).

Wet meadows in this area are typically dominated by grasses, rushes, sedges, and forbs (USFWS 2009). Dominant graminoids include clustered field sedge (*Carex praegracilis*), Baltic rush (*Juncus balticus*), tufted hairgrass (*Deschampsia cespitosa*), and mat muhly (*Muhlenbergia richardsonis*; USFWS 2009). Dominant forbs vary greatly with grazing intensity and soil moisture. With a trend toward more diversity and coverage with decreased grazing intensity (USFWS 2009). Grazing in our area either takes place annually or follows a 3-year rotation (rested for 3 years then grazed on the 4th).

Experimental Design

We divided each of the six drainages into multiple experimental units, which we called reaches. A reach consisted of a section of the drainage, but the length differed based on the width of the gully, so we could ensure independence among experimental units; the length of each reach was at least 20 times the width of the gully in an average water year. We designated each reach as treatment or control and each drainage had a similar number of treatment and control reaches.

Water-slowing structures (treatment: one-rock dams and brush dams) were installed in three drainages during fall 2018 (Little Basin Tributary 1, Clover Tributary, and Keystone Gulch) and in three drainages during fall 2019 (Teepee Creek, Little Basin Tributary 2, and Snowshoe Creek).

We used identical sampling methods in treated and control reaches. Treated reaches had at least three water-slowing structures, whereas control reaches had at least three locations that could have had such structures; we centered our data collection on 3 × 2-m subplots in these locations (Figure 3). We sampled the most upstream and downstream water-slowing structures (in treated reaches) or the location where structures would have been installed (in control reaches), as well as the most central structure/location. No sampling took place within a 1.5-m buffer of each water-slowing structure (or the location where it could have occurred in control reaches) to eliminate bias associated with the disturbed ground created during construction; we placed the subplot immediately upstream of this buffer area.

During summer of 2019, we sampled soil moisture and vegetation in 16 reaches (8 treatment and 8 control) in three drainages (Little Basin Tributary 1, Clover Tributary, and Keystone Gulch); this timing of sampling captured responses one year after treatment. We sampled each reach three times (visits): in June – when all reaches typically are saturated from snow melt; in July – to capture a transitional stage from saturated to dry; and in August – to compare treated and control reaches at their driest. During summer 2020, we again sampled in June, July, and August, as well as September to better capture drying trends later in the growing season (4 total visits). During this second sampling season, we collected data in the same 16 reaches, to capture responses two years after treatment. We also sampled 38 new reaches (18 treatment and 20 control) in three additional drainages (Teepee Creek, Little Basin Tributary 2, and Snowshoe Creek) in 2020, representing one year after treatment.

Soil Moisture

We characterized soil moisture by measuring volumetric water content at a depth of 10 cm (Vegetronix VG-Meter-200, Vegetronix, Inc., Riverton, Utah). In each subplot, we sampled soil moisture along a 10-m transect, which was perpendicular to and centered on the gully, 2-m above the structure. We collected measurements at 1-m increments, with one measurement taken in the center of the gully (11 total measurements per subplot, Figure 3). Gully widths varied and we only included soil moisture measurements that fell within the bank edges of the gully in analyses.

Vegetation

We used a canopy coverage approach to quantify plant communities (Herrick et al 2005) We estimated total cover of vegetation and non-vegetation (e.g. woody debris, bare ground) within a 0.5 x 0.5-m frame randomly placed within each subplot, resulting in three estimates per reach (0-100%; Figure 3; Herrick et al 2005). If the frame fell in an area that was not representative of the surrounding vegetation (e.g., trampled vegetation from previous visits), we selected a new random location. We then identified each individual plant to the lowest taxonomic level feasible and estimated percent coverage of each functional group (eg., forbs, grasses, litter; Herrick et al 2005); the total of these estimates did not exceed 100%.

Richness was calculated as the number of individual species counted during each visit (Gurevitch et al. 2006). We used the Shannon-Wiener index ($H`$) to characterize diversity (Gurevitch et al. 2006):

$$H` = -\sum [p_i \ln(p_i)]$$

where p_i = the proportion of ith species of total observed species (Gurevitch et al. 2006). We used Shannon evenness (J) to calculate how evenly species were distributed in each reach (Gurevitch et al. 2006):

$$J = \frac{H`}{lnS}$$

where S = the total number of species observed during the visit (or richness; Gurevitch et al. 2006).

Data Analysis

To characterize the efficacy of the restoration structures, we analyzed variation in soil moisture, as well as vegetation cover (by functional group) and species composition. We averaged data collected within each reach. We completed separate analyses for: 1) data collected one year after treatment (in 2019 and 2020) and 2) data collected two years after treatment (in 2020).

We modeled each response variable as a function of treatment and visit (month) using general linear mixed models; we included a random intercept for drainage to account for repeated sampling and inherent variation among drainages. To account for variation within each drainage, we included four additional covariates: relative distance, gully slope, bank aspect, and the width-height ratio for each reach. *Relative distance* was the distance between the sampled reach and the most upstream reach in the drainage, to account for spatial trends. *Gully slope* was the average incline of sampled areas within each reach, to adjust for lower water infiltration and drier conditions with steeper slopes (Famiglietti et al. 1998; Moore et al. 1988; Nyberg 1996; Hills & Reynolds 1969). To accommodate variation in how quickly soils dry after a period of moisture recharge (Reid 1973; Famiglietti et al. 1998), we averaged the *bank aspect* (river left) of sampled areas within each reach. Finally, we computed the *width-height ratio*, dividing the width of the gully by the height, averaged for all sampled areas in the reach, to account for changes in shape leading to changes in soil saturation (Zheng et al. 2006). Larger values of this ratio denote wider and shallower gullies, which tend to be wetter than narrower and deeper gullies (Zheng et al. 2006).

Results

We did not detect differences in soil moisture between treated and control reaches during any sampling period (Table 2.1). We also were unable to detect differences in vegetation cover or composition (Tables 2.2-2.3). However, despite the uncertainty around our estimates, canopy coverage often was higher in treatment reaches compared to controls two years after treatment, for all functional groups except rushes/sedges and native grasses (Figure 2.4). Similarly, diversity and evenness were higher in treatment compared to control reaches during some summer months (Figure 2.5).

Discussion

Vegetation regeneration after a disturbance event (e.g., fire, mining) can vary greatly based on the type of disturbance, soil properties, habitat type, or climate (Greipsson 2011); the same is true for restoration efforts. Contrary to Silverman et al. (2019), we did not detect differences in vegetation characteristics immediately after treatment; several factors may have contributed to these disparate findings. First, low-tech restoration structures need flowing water to be effective (Zeedyk & Clothier 2009), such that differences in climate could lead to variation in results. Our study area receives more annual precipitation (50 cm) than the Gunnison Basin (27 cm), the location of Silverman et al. (2019), but the average annual temperatures are much lower (1.6°C in Red Rock Lakes and 3.1°C in the Gunnison Basin; USFWS 2009, Aldridge et al. 2012). Additionally, the Centennial Valley received below average precipitation (26 cm) during 2020 – the second year after installation, whereas the Gunnison Basin received similar to average precipitation in 2014, two years after installation (National Weather Service 2022). Second, timing of installation also may influence results. In the Gunnison Basin, restoration structures were installed between July and October (TNC & GCWG 2017), whereas structures were installed in our study area in October, when freezing temperatures become more common (USFWS 2009). In colder climates and areas with longer periods of freezing temperatures, plants likely grow more slowly (Li et al. 2008; Tonin et al. 2019). Therefore, local environmental conditions may be an important influence on the timing of the effectiveness of low-tech wet meadow restoration.

We also used different metrics to assess the efficacy of treatments compared to Silverman et al. (2019); they focused on comparing plant productivity based on Normalized Difference Vegetation Index (NDVI), whereas we used plant canopy coverages, collected in the field. NDVI uses satellites to measure the amount of near-infrared light reflected by green leaves, and is used as an index of plant productivity (Sellers et al. 1992). Although plant canopy coverage and productivity are likely related, canopy coverage simply measures the amount of aerial cover of a plant within a specified area (Daubenmire 1959). Further, these metrics typically represent very different spatial scales; Silverman et al. (2019) measured vegetation from the air within a 30-m grid cell, whereas our measurements were collected on the ground, in small plots at the field site. Influences of these restoration structures occur at a small scale, which would seem to require fine-scale measurements, although measuring biomass may have better captured possible vegetation changes.

Although we expected initial treatment effects to be most pronounced for soil moisture, this was not the case. We measured moisture instantaneously at a depth of 10 cm; these methods were most feasible in terms of time and cost. Soil moisture at that depth can vary greatly during the summer due to changes in air temperature and humidity (Ajmal et al. 2016). Detecting meaningful changes in soil moisture could require measuring moisture at 20-cm or deeper (Ajmal et al. 2016) or using continuous soil moisture loggers.

Restoration often takes time, such that it is uncommon to detect immediate effects of restoration efforts (Ton et al. 1998; Meyer et al. 2010; Hopple & Craft 2013; Jing et al. 2014; Cooper et al. 2017). Although we had lower sample sizes to evaluate effects two years after treatment, canopy coverage often was higher in treatment reaches compared to controls. Rushes/sedges and native grasses were exceptions to this pattern, which may be due to the most common species in these groups. Coverage of rushes and sedges was dominated by Baltic rush, a species adapted to very wet soils (Lesica et al. 2012), and our wettest reaches (i.e. reaches where we observed water running through them for most of the study period) were typically controls (T. Sutton, *pers. observ.*). Further, the lower canopy coverage of native grasses in treated reaches likely resulted because most native grasses we found were more xeric-adapted species, namely Idaho fescue (*Festuca idahoensis*).

We also observed some qualitative evidence that these structures are beginning to positively effect wet meadows. In our wettest drainages (e.g., Little Basin 1 and Little Basin 2), we observed structures slowing water and sediment deposits, when compared to control reaches

(Figure 6). With additional time, restoration may improve the function of these wet meadows in more measurable ways, allowing them to retain nutritious vegetation late in the growing season, in the face a dwindling snowpack.

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ILLUSTRATIONS

Tables

Table 2.1. Soil volumetric water content (% saturation, means and 95% CIs) in sampled reaches, one year post-treatment (summers 2019 and 2020, *n* = 53, 27 control, 26 treatment) and two years post-treatment (summer 2020, *n* = 16, 8 control, 8 treatment), southwestern Montana.

		June	**July**	**August**	**September**
1 year post-treatment	**Treatment**	43.6 (25.9 - 61.4)	30.6 (12.8 - 48.3)	21.8 (4.0 - 39.6)	16.3 (-2.1 - 34.8)
	Control	44.2 (26.3 - 62.2)	31.2 (13.3 - 49.1)	18.0 (0.0 - 35.9)	18.0 (-0.7 - 36.8)
2 years post-treatment	**Treatment**	46.8 (28.6 - 65.0)	62.4 (41.9 - 82.8)	32.0 (14.0 - 50.0)	48.2 (32.5 - 64.0)
	Control	31.4 (11.5 - 51.2)	60.7 (45.0 - 76.5)	29.6 (13.8 - 45.3)	48.0 (32.5 - 63.0)

Table 2.2. Canopy coverage of functional groups (% coverage, means and 95% CIs) in sampled reaches, one year post-treatment (summers 2019 and 2020, *n* = 53, 27 control, 26 treatment), southwestern Montana.

		June	**July**	**August**	**September**
Total Vegetation	**Treatment**	64.5 (52.4 - 76.6)	72.7 (60.7 - 84.6)	69.3 (57.3 - 81.2)	70.3 (57.6 - 83.1)
	Control	60.5 (48.5 - 72.4)	69.3 (57.2 - 81.4)	69.0 (56.8 - 81.1)	70.8 (58.0 - 83.6)
Total green vegetation	**Treatment**	61.8 (48.7 - 74.9)	67.8 (48.7 - 74.9)	50.6 (37.5 - 63.7)	27.5 (13.5 - 41.4)
	Control	65.9 (52.6 - 79.1)	65.0 (51.7 - 78.3)	46.4 (33.1 - 59.7)	28.8 (14.7 - 42.8)
Native forbs	**Treatment**	20.1 (13.2 - 26.0)	20.5 (13.6 - 27.4)	21.3 (14.3 - 28.2)	18.9 (11.4 - 26.3)
	Control	20.0 (13.0 - 27.1)	21.4 (14.4 - 28.4)	17.4 (10.4 - 24.4)	15.4 (7.9 - 22.9)
Green native forbs	**Treatment**	18.9 (12.3 - 25.5)	18.6 (12.0 - 25.2)	15.6 (9.0 - 22.2)	14.3 (7.3 - 21.4)
	Control	19.1 (12.4 - 25.8)	19.8 (13.1 - 26.5)	13.5 (6.8 - 20.2)	11.5 (4.4 - 18.7)
Rushes/sedges	**Treatment**	5.5 (-2.4 - 13.3)	7.3 (-0.6 - 15.1)	7.3 (-0.6 - 15.1)	8.7 (-5.8 - 11.5)
	Control	13.1 (5.1 - 21.1)	10.8 (2.8 - 18.8)	10.8 (2.8 - 18.8)	12.1 (4.0 - 20.1)
Green rushes/sedges	**Treatment**	6.4 (-0.3 - 13.1)	7.2 (0.4 - 13.9)	5.9 (-0.8 - 12.7)	0.1 (-0.8 - 12.7)
	Control	13.3 (6.5 - 20.1)	10.3 (3.4 - 17.1)	7.3 (0.5 - 14.2)	7.3 (0.5 - 14.9)
Native grasses	**Treatment**	3.6 (-0.7 - 7.8)	13.6 (9.9 - 17.3)	8.4 (4.6 - 12.1)	3.6 (-0.7 - 7.8)
	Control	9.6 (5.9 - 13.4)	11.4 (7.6 - 15.2)	8.3 (4.4 - 12.1)	4.7 (0.4 - 8.9)
Green native grasses	**Treatment**	5.3 (-1.4 - 11.9)	7.1 (-1.5 - 15.7)	16.8 (9.2 - 24.2)	13.7 (6.1 - 21.4)
	Control	8.2 (1.5 - 14.8)	9.8 (3.1 - 16.5)	15.0 (8.3 - 21.6)	18.0 (9.8 - 26.3)

Table 2.2. Continued.

Non-native grasses	**Treatment**	14.1 (6.4 - 21.9)	20.6 (12.9 - 28.4)	17.6 (9.9 - 25.4)	24.4 (15.8 - 33.1)
	Control	13.0 (7.4 - 21.1)	15.2 (7.3 - 23.1)	17.0 (9.0 - 24.9)	26.7 (18.0 - 35.4)
Green non-native grasses	**Treatment**	14.9 (8.2 - 21.6)	19.1 (12.4 - 25.8)	12.9 (6.1 - 19.6)	4.9 (-2.7 - 12.4)
	Control	14.3 (7.4 - 21.1)	14.4 (7.5 - 21.3)	9.9 (3.0 - 16.9)	5.0 (-2.6 - 12.6)
Non-native forbs	**Treatment**	8.6 (4.6 - 12.6)	6.8 (2.9 - 10.8)	3.6 (-0.4 - 7.6)	0.3 (-4.0 - 4.6)
	Control	5.3 (1.2 - 9.3)	5.9 (1.9 - 10.0)	3.9 (-0.17 - 7.9)	0.6 (-3.8 - 4.9)
Green non-native forbs	**Treatment**	8.9 (5.2 - 12.7)	5.5 (1.7 - 9.2)	1.8 (-2.0 - 5.6)	-0.1 (-4.1 - 4.0)
	Control	5.8 (2.0 - 9.6)	5.2 (1.4 - 9.1)	2.9 (-0.9 - 6.7)	0.2 (-3.9 - 4.3)

Table 2.3. Species richness, Shannon-Weiner diversity, and species evenness (means and 95% CIs) in sampled reaches, one year post-treatment (summers 2019 and 2020, *n* = 53, 27 control, 26 treatment), southwestern Montana.

		June	**July**	**August**	**September**
Richness	**Treatment**	21.3 (15.8 - 26.9)	21.8 (16.3 - 27.4)	18.8 (13.2 - 24.3)	17.1 (11.3 - 22.9)
	Control	22.8 (17.2 - 28.4)	24.4 (18.8 - 30.0)	20.0 (14.4 - 25.6)	18.3 (12.5 - 24.1)
Diversity	**Treatment**	1.46 (1.07 - 1.85)	1.54 (1.15 - 1.93)	1.47 (1.08 - 1.85)	1.40 (0.99 - 1.81)
	Control	1.62 (1.23 - 2.01)	1.76 (1.37 - 2.16)	1.70 (1.31 - 2.09)	1.62 (1.21 - 2.03)
Evenness	**Treatment**	45.9 (36.3 - 55.4)	48.8 (39.3 - 58.4)	49.7 (40.1 - 59.2)	51.9 (41.8 - 62.1)
	Control	51.9 (42.3 - 61.6)	55.2 (45.5 - 64.8)	57.9 (48.2 - 67.6)	59.6 (49.3 - 69.8)

Figures

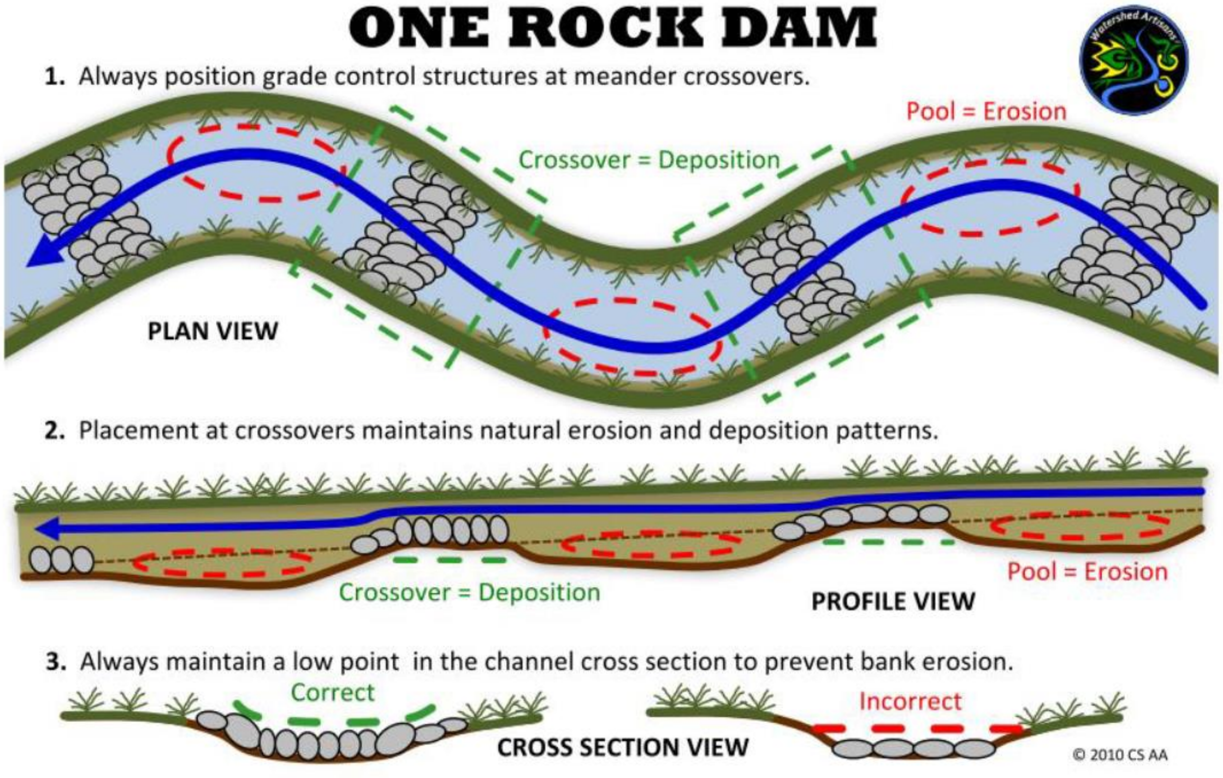

Figure 2.1. Visual representation of one-rock dams (Sponholtz & Anderson 2013).

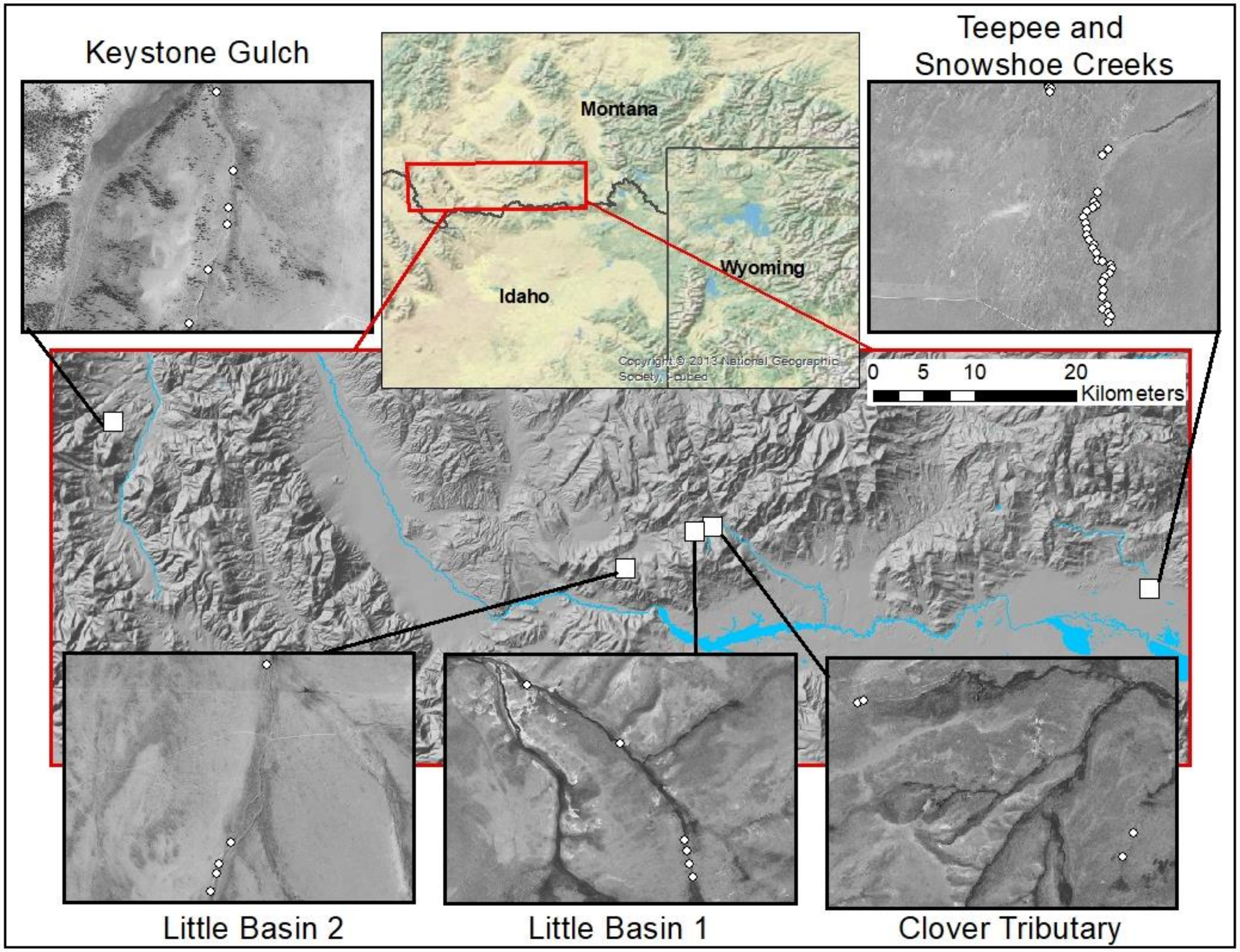

Figure 2.2. Map of sampled drainages, southwestern Montana. Each white dot represents one reach of wet meadow restoration.

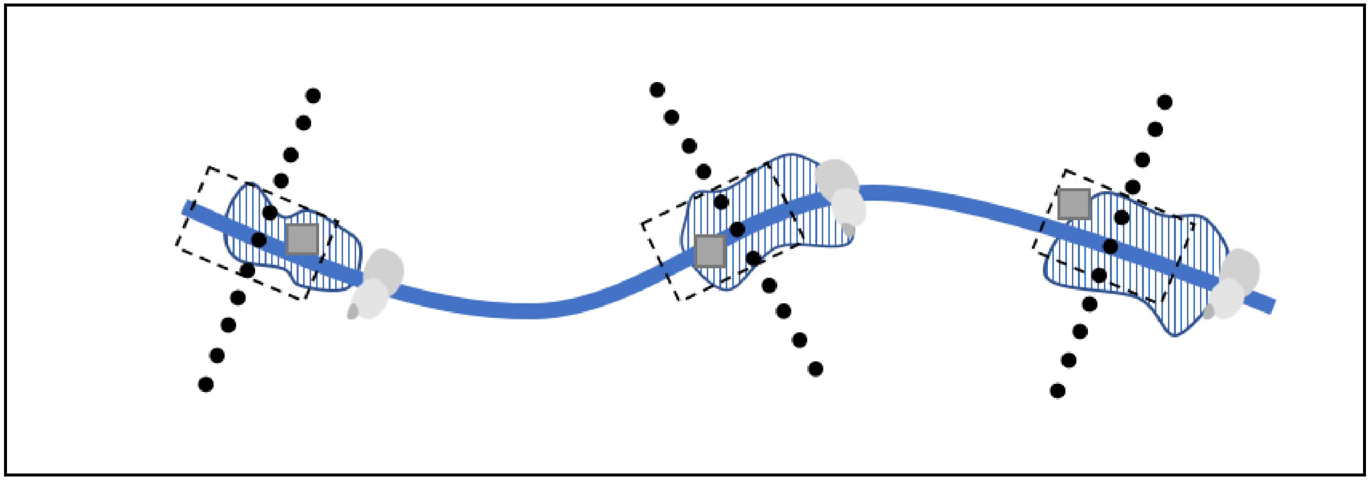

Figure 2.3. Visual representation of a reach, our experimental unit, and sampling areas including the sampling area (black hashed line), vegetation frame (gray square), and soil moisture readings (black circles) southwestern Montana.

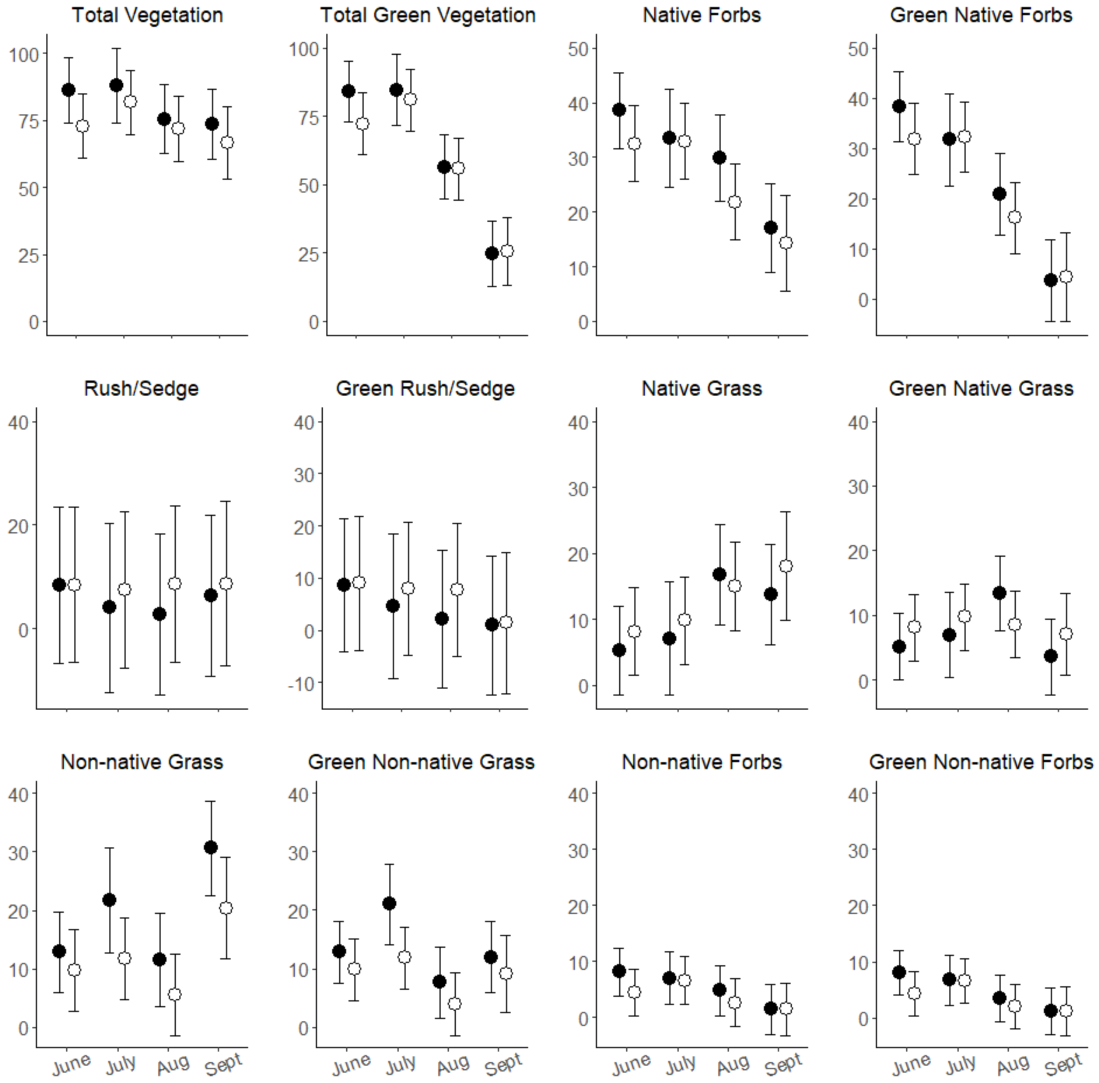

Figure 2.4. Canopy cover of each functional group (means and 95% CIs) in treatment (black) and control (white) reaches, two years post-treatment (summer 2020, n = 16, 8 control, 8 treatment), southwestern Montana.

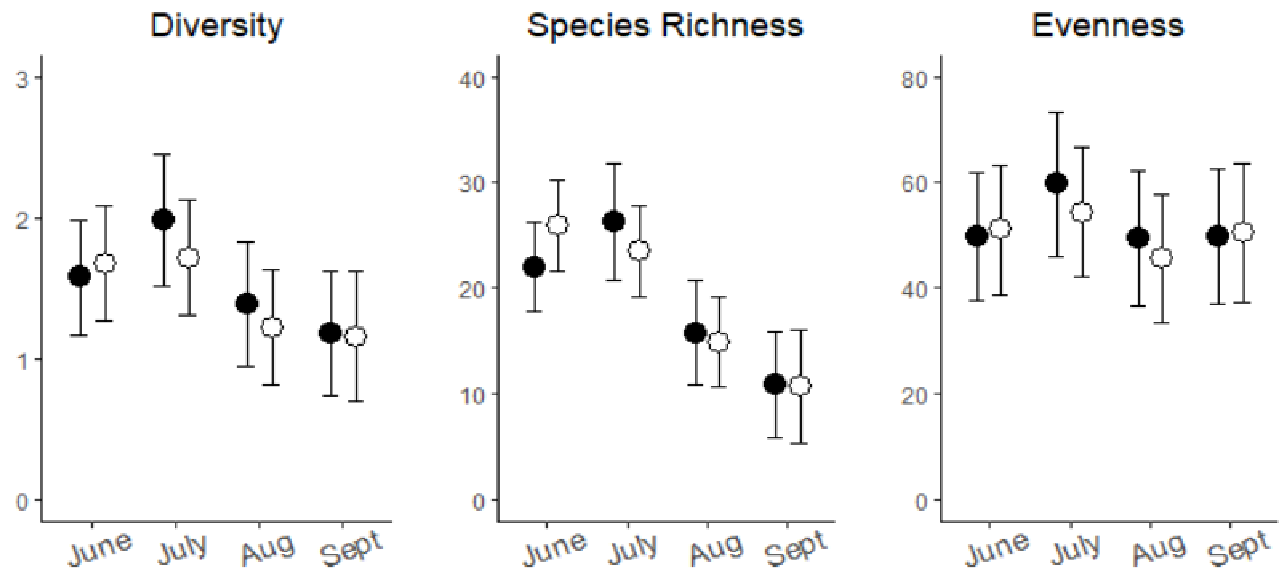

Figure 2.5. Shannon-Weiner diversity, species richness, and species evenness (means and 95% CIs) in treatment (black) and control (white) reaches, two years post-treatment (summer 2020, n = 16, 8 control, 8 treatment), southwestern Montana.

Figure 2.6. A. Water being slowed at a one-rock dam, Little Basin 1, July 2019, southwestern Montana. Photo by Thomas Sutton. B. Untreated control reach, where orange paint (right side) represents where a structure would have been placed, July 2019, southwestern Montana. Photo by Thomas Sutton. C. Sediment deposit above water slowing structures, August 2020, southwestern Montana. Photo by Laura Robison.

CHAPTER THREE

ASSESSING FOOD RESOURCES OF FOUR SAGEBRUSH-ASSOCIATED BIRDS AFTER LOW-TECH WET MEADOW RESTORATION

Contribution of Authors and Co-Authors

Manuscript in Chapter 3

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Contributions: Implemented the study, collected and analyzed data, wrote manuscript

Co-Author: Andrea R. Litt

Contributions: Helped develop initial study idea, guided study design, assisted in securing funding, helped with data analysis, thoroughly edited manuscript

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Manuscript Information

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Restoration Ecology

Status of Manuscript:
_X__ Prepared for submission to a peer-reviewed journal
____ Officially submitted to a peer-reviewed journal
____ Accepted by a peer-reviewed journal
____ Published in a peer-reviewed journal

Society for Ecological Restoration

Assessing food resources of four sagebrush-associated birds after low-tech wet meadow restoration

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Author Contributions: TS implemented the study, collected and analyzed data, and wrote manuscript; AL assisted with data analysis; AL, BS developed initial study idea and secured funding; AL, BS, HG guided study design and reviewed manuscript.

ABSTRACT

The sagebrush steppe is home to over 200 vertebrate species. Wet meadows are an important habitat component for many of these animals, including the greater sage grouse, which relies on these areas in late summer when upland vegetation senesces. In Montana, seasonal streams feed wet meadows in late spring and early summer as high-elevation snow melts. Increased temperatures and overgrazing has led to wet meadows drying earlier in the year and the need to explore restoration solutions. We used low-tech restoration methods within an experimental framework to retain water in wet meadows; we compared plant and arthropod foods of sage grouse chicks in treated and control areas. We also compared foods that three other bird species consume during the nesting period: sage thrashers, Brewer's sparrows, and vesper sparrows. We did not detect differences in arthropod foods for any bird species or plant foods for sage grouse chicks during any sampling period. We did detect higher coverage of plant foods for vesper sparrows two years after treatment during September. These plant foods were dominated by Kentucky bluegrass, a plant adapted to growing in moist soils, providing promise that these structures are starting to retain green vegetation later in the season. Our results reflect a need to better understand the efficacy of wet meadow restoration, especially variation in the time needed to detect effects. With more time and continued monitoring, these structures may improve habitat for sagebrush-associated species in the face of climate change.

Key words: mesic restoration, sage grouse, vesper sparrow, Brewer’s sparrow, sage thrasher, Zeedyk structures

IMPLICATIONS FOR PRACTICE

- Improvement of habitat for some sagebrush associated birds may take longer than two years, warranting the need for continued monitoring
- Assessing food resources of wildlife species is a helpful metric for success in restoration treatments

MAIN TEXT

Introduction

The semi-arid sagebrush steppe covers a vast area in western North America, providing a home for more than 200 vertebrate species (Knight et al. 2014). Seven of these species cannot survive in any other environment (Knight et al. 2014), yet even these sagebrush obligates require a mosaic of vegetation communities to meet their habitat needs (Klebenow 1969; Wallestad 1975; Howe et al. 2000). For example, moisture is limited in the sagebrush steppe and animals may need to look elsewhere for water sources (Knight et al. 2014; Greipsson 2011; Aldridge & Brigham 2007). Access to water often is provided by wetlands (Knight et al. 2014; Greipsson 2011) or wet meadows, which are saturated with water during the spring, but typically dry later in the summer (Knight et al. 2014). Even after drying, wet meadows retain more moisture than their surroundings, making them important oases of plant and arthropod foods for many species associated with the sagebrush steppe (Knight et al. 2014; Naiman et al. 2010; Greipsson 2011).

The greater sage grouse (*Centrocercus urophasianus*, sage grouse hereafter) is a sagebrush-obligate species, yet wet meadows also are an important component of their habitat (Klebenow 1969; Wallestad 1975; Schreiber et al. 2015). In late summer, when upland vegetation senesces, sage grouse move to moist areas like wet meadows to find palatable green vegetation and arthropods (Drut et al. 1994; Wallestad 1975; Schreiber et al. 2015). The foods provided by wet meadows can be especially important for survival of sage grouse chicks (Schreiber et al. 2015; Dunn & Braun 2007; Drut et al. 1994). In captive-reared sage grouse, chicks experience 100% mortality when arthropods are completely excluded from their diet (Johnson & Boyce 1990). Further, brood-rearing hens most commonly select vegetation in wet meadows within a landscape dominated by big sagebrush (*Artemisia tridentata;* Schreiber et al. 2015).

Seasonal streams feed wet meadows in late spring and early summer as snow melts at high elevation (Knight et al. 2014; Seager & Vecchi 2010). Yet, increased average annual temperatures has reduced the depth of the snowpack and melts snowpack earlier in the year (Seager & Vecchi 2010), causing wet meadows to receive less moisture for shorter periods of time. Wet meadows also have been susceptible to overgrazing by livestock, given the abundance of palatable vegetation (Knight et al. 2014). This removal of vegetation can destabilize soils (Knight et al. 2014), eventually altering the flow of seasonal streams and resulting in increased area of upland vegetation communities (Zeedyk & Clothier 2009; Loheide & Gorelick 2007; Donnelly et al. 2016).

Decreases in the extent of wet meadows can lower habitat quality for sage grouse, due to reduced abundance of late season forage (Wallestad 1975; Gregg et al. 1993; Schreiber et al. 2015). When wet meadows, and the important food resources they provide, are less abundant, sage grouse hens with broods have larger home range sizes, which relates to decreased chick survival (Gregg et al. 1993; Drut et al. 1994). Increasing the extent of wet meadows through restoration may improve habitat for sage grouse chicks, thus bolstering populations (Silverman et al. 2019; Schreiber et al. 2015; Gregg et al. 1993), as well as benefiting other wildlife species (Rowland et al. 2006). Additionally, sage grouse now occupy roughly half of their historic range (Schroeder et al. 2004), elevating the need to restore and conserve wet meadows.

We used low-tech restoration methods (Zeedyk & Clothier 2009) within an experimental framework to slow water movement and retain moisture in wet meadows (Chapter Two). To assess habitat-related changes resulting from these restoration efforts, we compared food resources (both plants and arthropods) for sage grouse chicks in treated and control areas throughout the summer growing season. Although we were focused on sage grouse, we also explored the potential that restoration activities could influence food resources for other birds with similar preferences for nesting habitat (Rowland et al. 2006). We considered two other sagebrush obligates: sage thrashers (*Oreoscoptes montanus*) and Brewer's sparrows (*Spizella breweri*), as well as vesper sparrows (*Pooecetes gramineus*), a species associated with sagebrush landscapes in western North America (Paige & Ritter 1999). For each of these species, we were interested in understanding how known arthropod and plant foods changed with wet meadow restoration.

Sage grouse, sage thrashers, Brewer’s sparrows, and vesper sparrows all eat a diet of diverse arthropods (Evans 1964; Howe et al. 2000; Petersen & Best 1986; Peterson 1970; Klebenow & Gray 1968). Additionally, sage thrashers, Brewer’s sparrows, and vesper sparrows mostly consume arthropods while nesting (Rotenberry 1980; Petersen & Best 1986; Howe et al. 2000). Forbs are important food plants for sage grouse chicks, and consuming a greater proportion of forbs, relative to sagebrush, could contribute to increased chick survival (Drut et al. 1994). Sage thrashers and Brewer’s sparrows only eat trace amounts of vegetation while nesting (Howe et al. 2000; Petersen & Best 1986), whereas vesper sparrows eat a variety of seeds (Rotenberry 1980; Evans 1964).

In spring, when most areas in southwest Montana are saturated from melting snow, we did not expect to detect differences in arthropod or plant foods between treated and control areas early in the growing season. However, soils begin to dry later in the growing season; as such, we expected arthropod and plant foods would be more abundant in treated areas, compared to untreated controls. More abundant food could improve habitat quality for each of these bird species.

Methods

Site Background

Our work focused on wet meadows located around channels carved by snow melt in the sagebrush steppe of southwest Montana, in the general area of Red Rock Lakes National

Wildlife Refuge (Red Rock Lakes hereafter). Wet meadows cover over 2800 ha of Red Rock Lakes (USFWS 2009). The climate is characterized by long, cold winters and short summers with variable annual precipitation (USFWS 2009); average annual temperature is 1.6° C and average annual precipitation is 50 cm (USFWS 2009). More than 60% of the water entering the wet meadows comes from snowmelt during the spring (Serreze et al. 1999).

Our research occurred in six drainages (Little Basin Tributary One, Little Basin Tributary Two, Teepee Creek, Clover Tributary, Clover Tributary, Keystone Gulch, and Snowshoe Creek, Figure 1), which are spread out over more than 2000 km^2. These drainages have different characteristics, but areas within a drainage are similar. Soils are primarily sandy with large particles and low water-holding potential (Teepee Creek), clayey with high water-holding potential (Keystone Gulch), or intermediate with roughly 50% sand and 50% clay (Little Basin Tributary 1, Little Basin Tributary 2, and Clover Tributary; California Soil Resource Lab 2021). Elevation ranged from 2000 to 2300 m and slope gradients ranged from 0% (Teepee Creek) to 11% (Little Basin Tributary Two).

Wet meadows in this area are typically dominated by grasses, rushes, sedges, and forbs (USFWS 2009). Dominant graminoids include clustered field sedge (*Carex praegracilis*), Baltic rush (*Juncus balticus*), tufted hairgrass (*Deschampsia cespitosa*), and mat muhly (*Muhlenbergia* richardsonis; USFWS 2009). Dominant forbs vary greatly with grazing intensity and soil moisture, with a trend toward more diversity and coverage with decreased grazing intensity (USFWS 2009). These drainages are grazed annually or follow a three-year rotation (rested for three years, then grazed on the fourth).

Experimental Design

We divided each of the six drainages into multiple experimental units, which we called reaches. A reach consisted of a section of the drainage, but the lengths differed so we could ensure independence among experimental units; the length of each reach was at least 20 times the width of the gully in an average water year. We designated each reach as treatment or control; each drainage had a similar number of treatment and control reaches. Water-slowing structures (treatment: one-rock dams and brush dams) were installed in four drainages during fall 2018 (Little Basin Tributary One, Clover Tributary, Clover Pass Tributary, Keystone Gulch) and in three drainages during fall 2019 (Teepee Creek, Little Basin Tributary Two, and Snowshoe Creek).

We used identical sampling methods in treated and control reaches. Treated reaches had at least three water-slowing structures, whereas control reaches had at least three locations that could have had such structures; we centered our data collection on three by two-meter plots in these locations (Figure 2). We sampled the most upstream and downstream water-slowing structures (in treated reaches) or the location where structures would have been installed (in control reaches), as well as the most central structure/location. No sampling took place within a 1.5-m buffer of each structure/location to eliminate bias associated with the disturbed ground created during construction; we placed the subplot immediately upstream of this buffer area.

During the summer of 2019, we sampled 16 reaches (8 treatment, and 8 control) in four drainages (Little Basin Tributary One, Clover Tributary, Clover Pass Tributary, and Keystone Gulch); this timing of sampling captured responses one year after treatment. During summer 2020, we collected data in the same 16 reaches, to capture responses two years after treatment. We also sampled 38 new reaches (18 treatment and 20 control) in three additional drainages

(Teepee Creek, Little Basin Tributary Two, and Snowshoe Creek) in 2020, representing one year after treatment. We sampled vegetation and arthropods in each reach three times (visits): once in each of the months of June, July, and August, to capture the gradient of senescence throughout the summer. During the summer of 2020, we added one additional vegetation sampling visit during the month of September.

Arthropod Sampling

To ensure we captured both ground- and vegetation-dwelling arthropods, we used a combination of vacuum (Dietrick 1961) and pitfall (Greenslade 1964) sampling to characterize arthropod communities in treatment and control reaches. We combined samples from each of the methods collected within a reach during the same visit.

Vacuum sampling - To capture flying and vegetation-dwelling arthropods, we used a vacuum/blower (Stihl SH56C, Stihl, Inc., Waiblingen, Germany) with window screen attached to the end (Davis et al. 2014). One vacuum sample was collected at a random location within each sampling plot, for a total of three for each experimental unit. At the selected location, we placed a 0.25-m^2 plastic barrel with an 800-micron screen covering on top, to prevent arthropods from escaping (Kruess & Tscharntke 2002). We ran the vacuum for 30 seconds within that barrel at each sampling location (Davis et al. 2014; Brook et al. 2008). To reduce bias from disturbing vegetation, we ensured vacuum sampling was the first task completed after arriving at the experimental unit (Standen 2000).

Pitfall sampling - We used pitfall trapping to capture ground-dwelling arthropods (Greenslade 1964). Ideally, pitfall traps should have a diameter between 6.5 and 15 cm, to allow for relatively efficient sampling without drastically increasing effort (Work et al. 2002). With this in mind, we used 9.5 x 12-cm plastic cups (Solo Cup Company, Lake Forest, Illinois) dug deep enough to be flush with the ground (Greenslade 1964). Additionally, we removed vegetation and debris from the area immediately surrounding the pitfall trap to allow for free movement of ground-dwelling arthropods (Greenslade 1964). We partially filled each pitfall trap with propylene glycol (LowTox Antifreeze/Coolant, Prestone Products Corporation, Lake Forest, Illinois) to kill and preserve trapped insects (Hohbein & Conway 2018).

We placed one pitfall trap within each sampling unit, for a total of three for each experimental unit, for one 24-hour period. For the first visit (June), we randomly selected a location within each sampling plot. For the second (July) and third (August) visits, we reused the pitfall hole dug during the first visit to reduce soil disturbance.

Arthropod Processing

After collection, arthropods were cleaned of debris and stored in 90% ethanol until further processing. We identified each collected arthropod to order. We measured all collected arthropods from frons to the tip of the abdomen to the nearest millimeter, then converted arthropod length to estimated biomass using taxon-specific length/mass regression equations (Rogers et al. 1977; Davis et al. 2014). We computed an estimated biomass by order for each reach, combining data from the three sampled locations during one visit. We focused our analysis on known important arthropod foods (see Data Analysis).

Vegetation Sampling

To quantify vegetation in treated and untreated reaches, we used a 0.5 x 0.5-m frame to estimate cover of important food resources for sage-grouse chicks. We measured vegetation in one random location within each sampling unit, resulting in three vegetation estimates in each experimental unit for each visit (Figure 2). If the frame fell in the area affected by pitfall trapping (i.e., over removed vegetation or pitfall hole), we selected another random location in an unaffected area.

We characterized vegetation in treated and untreated reaches based on a canopy coverage approach (Coulloudon et al. 1999). We estimated total coverage of vegetation and non-vegetation (e.g., woody debris, bare ground) within each frame (0-100%; Coulloudon et al. 1999). We then identified each individual plant to the lowest taxonomic level feasible and estimated canopy coverage for each (0-100%). We focused our analysis on known important plant food resources (see Data Analysis).

Data Analysis

We used information from the literature to characterize important arthropod and plant foods for our bird species of interest. Although we aimed to find diet information from nearby areas, we used references from other locations as needed.

Arthropod models. - Sage grouse chicks overwhelmingly consume arthropods belonging to the orders of Hymenoptera (mainly Formicidae), Coleoptera, Orthoptera, and larval Lepidoptera (Klebenow & Gray 1968; Peterson 1970; Gregg & Crawford 2009). Nesting sage thrashers and Brewer's sparrows primarily consume arthropods belonging to Araneae, Lepidoptera, Coleoptera, Diptera, Orthoptera, and Hymenoptera (Howe et al. 2000; Petersen & Best 1986). We used the combined estimated biomass (in µg) for important arthropod foods in each reach for each visit as the response variable for sage grouse chicks, sage thrashers, and Brewer's sparrows. Vesper sparrows tend to consume arthropods according to their relative abundance (Evans 1964). As such, we used the combined biomass of all collected arthropods as the response variable for vesper sparrows.

Vegetation models. - In Montana, Idaho, and Oregon, chicks primarily consume forbs belonging to the families Brassicaceae, Fabaceae, Polemoniaceae, Asteraceae, and Lilaceae in areas dominated by big sagebrush (*Artemisia tridentata*; Patterson 1952; Klebenow & Gray 1968; Drut, Pyle, et al. 1994; Peterson 1970; Wallestad 1975; Martin 1970). We did not have a published diet assessment for vesper sparrows in our study area. Instead, we used research from Michigan, which found that nesting vesper sparrows and their chicks consumed seeds from species in the genera *Cyperus, Danthonia, Deschampsia, Panicum, Poa, Setaria, Sorghastrum, Asclepias, Oxalis,* and *Polygonum* (Evans 1964). We compared canopy coverage of known important genera within these families as the response variable in models for sage grouse chicks and nesting vesper sparrows. In southwestern Montana, vesper sparrows typically nest through late August (Davis 1961) and migrate south in middle to late September (Skaar 1969). Given that food resources might still be useful to the birds until migration, we examined their nesting food resources from June through September.

Model structure. – We developed separate models by bird species (sage grouse chicks, nesting Brewer’s sparrows, nesting sage thrashers, or vesper sparrows) and food type (plants or arthropods). We also completed separate analyses for: 1) data collected one year after treatment (in 2019 and 2020) and 2) data collected two years after treatment (in 2020). We modeled each response variable as a function of treatment (treated or control) and visit (month) using general linear mixed models; we included a random intercept for drainage to account for repeated sampling and inherent variation among drainages. To account for variation within each drainage, we included four additional covariates: relative distance, gully slope, bank aspect, and the width-height ratio for each reach. *Relative distance* was the distance between the sampled reach and the most upstream reach in the drainage, to account for potential spatial trends. *Gully slope* was the average incline of the sampled areas within each reach, to adjust for lower water infiltration and drier conditions with steeper slopes (Hills & Reynolds 1969; Moore et al. 1988; Nyberg 1996; Famiglietti et al. 1998). To accommodate variation in how quickly soils dry after a period of moisture recharge (Reid 1973; Famiglietti et al. 1998), we averaged the *bank aspect* (river left) of sampled areas within each reach. Finally, we computed the *width-height ratio*, dividing width of the gully by the height, averaged for all sampled areas in the reach, to account for changes in shape leading to changes in soil saturation (Zheng et al. 2006). Larger values of the ratio denote wider and shallower gullies, which tend to be wetter than narrower and deeper gullies (Zheng et al. 2006).

Results

We did not detect differences in important forage items for sage grouse chicks, nesting Brewer's sparrows, or nesting sage thrashers between treated and control reaches at any time post treatment (Tables 3.1 and 3.2). Similarly, we did not detect differences in arthropod foods for vesper sparrows at any time post-treatment or important plant foods for vesper sparrows for most sampling periods after treatment (Tables 3.1 and 3.2). However, we did find 13% higher coverage of important plant foods for vesper sparrows in treatment reaches (33.7%, 95% CI = 27.9 to 39.4) compared to control reaches (20.7%, 14.5 to 26.9) in September, two years after treatment (Figure 3.3).

Discussion

Wildlife and plants recover and regenerate at different rates after disturbance events and restoration efforts; these differences depend on many factors including the type and severity of disturbance, soil properties, and climate (Greipsson 2011). We sampled these reaches one and two years after treatment, but have yet to detect differences in soil moisture or coverage of any plant functional group (Chapter Two). Although previous research in the Gunnison Basin suggests that vegetation characteristics can change immediately after wet meadow restoration (Silverman et al. 2019), colder temperatures at our study sites may require that water-slowing structures need to be in place for longer periods before detecting treatment effects. As such, it is perhaps not surprising that we detected few differences in plant or arthropod foods for these bird species in the first years after treatment.

We did detect higher canopy coverage of plant foods for nesting vesper sparrow in treatment reaches than control during the month of September, two years after treatment. Kentucky bluegrass (*Poa pratensis*) dominated this plant group (97%) and is a very mesic-adapted plant, thriving in soils with intermediate amounts of moisture (Lesica et al. 2012). Detecting higher canopy coverage of these plants provides some indication that the restoration structures are beginning to retain more moisture during drier periods of the summer, which in turn, can provide important foods for these bird species.

Many of the plant species that provide important forage for our birds of interest also may provide habitat for arthropods (Niemela et al. 1992; Xiao-Dong & Hong-Zhang 2006; Stamps et al. 2009; Borchard et al. 2013; Perez-Sanchez et al. 2018; Pinedo-Escatel & Moya-Raygoza 2018). Although arthropods have short generation times and can respond rapidly to ecological changes, some time lag is likely required between detecting changes in vegetation characteristics and subsequent changes in arthropod populations. Continued monitoring of these areas seems warranted to detect potential changes. We also identified arthropods at a reasonably coarse level of taxonomic resolution, which could mask varied habitat needs and responses to wet meadow restoration (Niemela et al. 1992; Xiao-Dong & Hong-Zhang 2006; Stamps et al. 2009; Borchard et al. 2013; Perez-Sanchez et al. 2018; Pinedo-Escatel & Moya-Raygoza 2018). Identifying arthropods to family or genus could provide important insights.

Although previous work showed immediate changes in vegetation after wet meadow restoration (Silverman et al. 2019), our results suggest delayed effects may be more likely in some environments, especially to detect changes in arthropods. Continued post-treatment monitoring may show that these structures are improving wet meadows and, in turn, benefitting the wildlife that rely on them. Even if these low-tech solutions do not provide an instant “cure-all” for wet meadow restoration, changes in climate and land-use practices emphasize the need to find effective and practical tools to improve habitat for sagebrush-associated species.

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ILLUSTRATIONS

Tables

Table 3.1. Means (and 95% CIs) from models assessing important arthropod (biomass in µg) and plant foods (percent canopy coverage) for different bird species in sampled reaches, one year post-treatment (summers 2019 and 2020, *n* = 53, 27 control, 26 treatment), southwestern Montana. We sampled plants in June, July, August, and September, and arthropods in June, July, and August.

Bird species	Food type		June	July	August	September
Sage grouse	Arthropods	**Treatment**	40.6 (-89.8 - 171.0)	123.7 (6.3 - 241.1)	565.0 (445.2 - 684.8)	
		Control	111.1 (-18.0 - 240.2)	186.5 (60.2 - 312.8)	470.0 (343.6 - 596.3)	
	Plants	**Treatment**	6.6 (4.5 - 8.8)	6.5 (4.3 - 8.7)	8.5 (6.3 - 10.8)	8.1 (5.1 - 11.0)
		Control	6.5 (4.2 - 8.7)	7.1 (4.9 - 9.3)	7.3 (4.9 - 9.6)	7.5 (4.3 - 10.6)
Sage thrasher and Brewer’s sparrow	Arthropods	**Treatment**	93.3 (-28.3 - 214.9)	204.6 (98.8 - 310.4)	619.3 (508.6 - 729.9)	
		Control	132.1 (18.5 - 245.7)	304.6 (191.0 - 418.2)	527.1 (413.5 - 640.7)	
Vesper sparrow	Arthropods	**Treatment**	174.4 (46.7 - 302.1)	214.0 (102.8 - 325.1)	664.6 (548.4 - 780.8)	
		Control	159.7 (40.3 - 279.0)	317.0 (197.7 - 436.4)	575.9 (456.8 - 695.3)	
	Plants	**Treatment**	14.9 (10.1 - 19.8)	16.3 (11.5 - 21.2)	16.0 (11.2 - 20.8)	25.9 (20.2 - 31.6)
		Control	15.0 (10.0 - 20.0)	13.4 (8.5 - 18.3)	16.6 (11.6 - 21.5)	24.8 (19.2 - 30.4)

Table 3.2. Means (and 95% CIs) from models assessing important arthropod (biomass in µg) and plant foods (percent canopy coverage) for different bird species in sampled reaches, two years post-treatment (summer 2020, *n* = 16, 8 treatment, 8 control), southwestern Montana. We sampled plants in June, July, August, and September, and arthropods in June, July, and August

Bird Species	**Food Type**		**June**	**July**	**August**	**September**
Sage grouse	Arthropods	**Treatment**	48.1 (-111.0 - 207.2)	123.0 (-86.9 - 332.9)	200.7 (16.4 - 385.1)	
		Control	86.2 (-73.6 - 246.0)	110.7 (-49.2 - 270.5)	501.8 (342.0 - 661.6)	
	Plants	**Treatment**	10.0 (7.0 - 13.0)	6.0 (2.1 - 9.9)	6.7 (3.3 - 10.2)	3.8 (-0.53 - 8.1)
		Control	6.0 (2.7 - 9.2)	6.7 (3.5 - 10.0)	7.8 (4.6 - 11.0)	6.9 (2.6 - 11.2)
Sage thrasher and Brewer's sparrow	Arthropods	**Treatment**	83.2 (-90.9 - 257.3)	223.2 (-6.6 - 453.1)	253.5 (51.7 - 455.3)	
		Control	135.4 (-39.5 - 310.3)	168.8 (-6.1 - 343.7)	568.0 (393.1 - 742.9)	
Vesper sparrow	Arthropods	**Treatment**	108.6 (-70.7 - 287.9)	221.97 (-14.7 - 458.7)	260.7 (52.9 - 468.5)	
		Control	171.7 (-8.4 - 351.9)	179.3 (-0.8 - 359.4)	598.7 (418.5 - 778.8)	

Figures

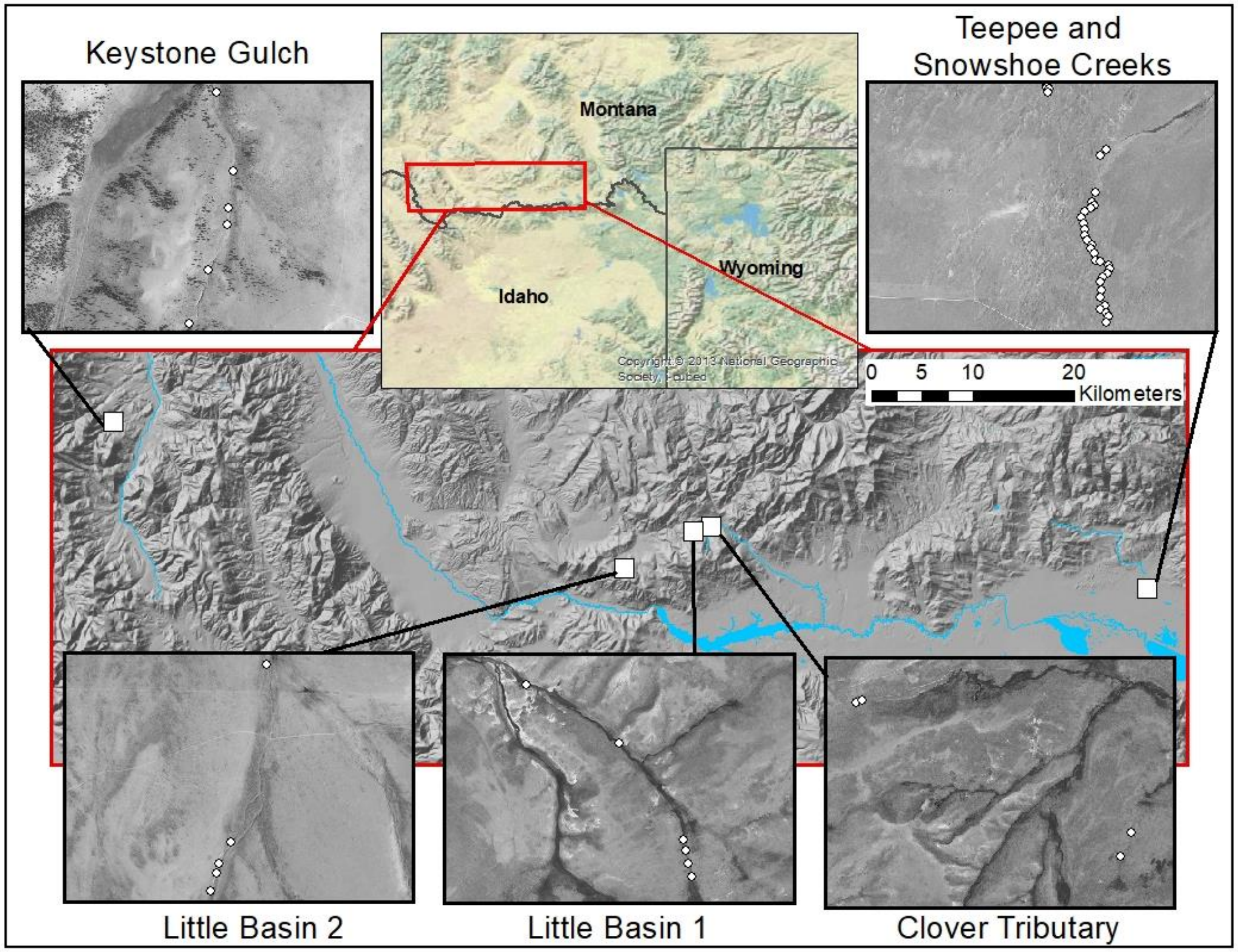

Figure 3.1. Map of sampled drainages, southwestern Montana, 2019-2020. We examined the efficacy of wet meadow restoration; each white dot represents 1 treatment or control reach.

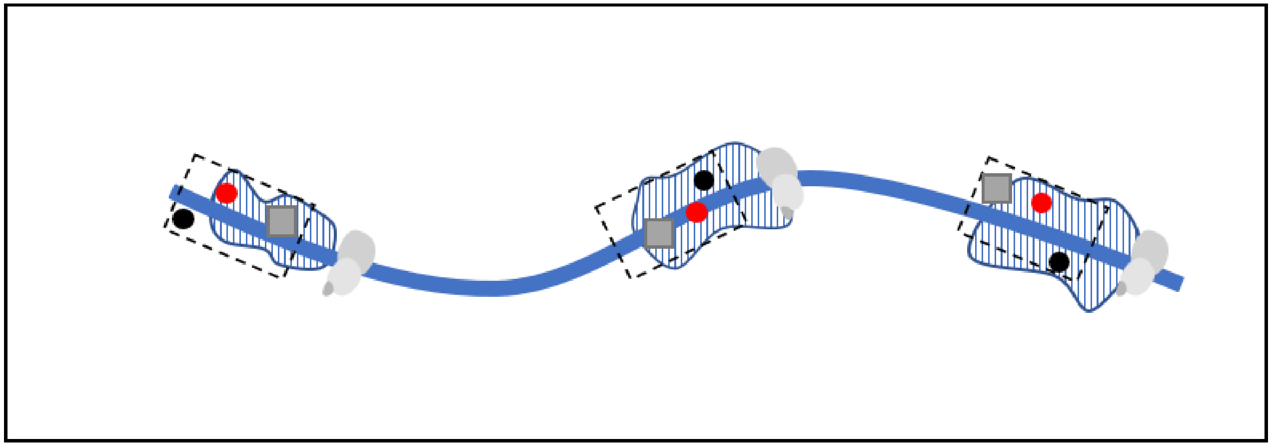

Figure 3.2. Visual representation of a reach, our experimental unit, including the sampling area (black hashed line), vegetation frame (gray squares), pitfall sample (black circles), vacuum sample (red circles), southwestern Montana, 2019-2020. This image shows a treated reach (see depictions of the one-rock dams), but we used the same sampling layout for control reaches. Not to scale.

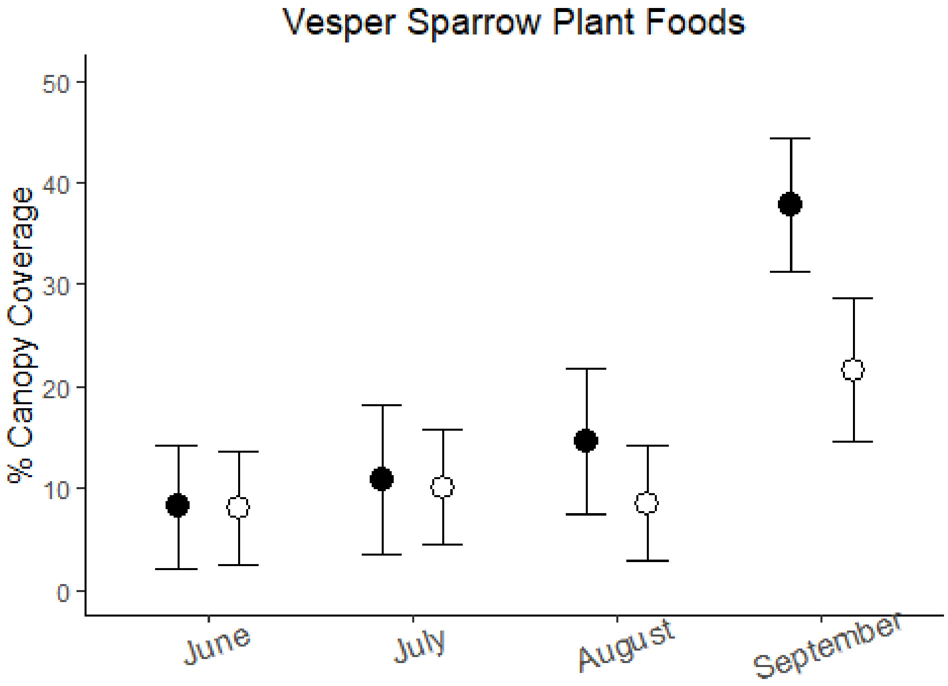

Figure 3.3. Canopy coverage (%) of important plant foods for nesting vesper sparrows (means and 95% CIs) in treatment (black) and control (white) reaches, two years post-treatment (summer 2020, n = 16, 8 treatment, 8 control), southwestern Montana.

CHAPTER FOUR

CONCLUSION OF THESIS

General Conclusions

Throughout the western United States, low-tech methods have been implemented to restore degraded wet meadows and improve habitat for species that rely on these areas, with promising indications of success (Zeedyk & Clothier 2009; Silverman et al. 2019; Hammersmark et al. 2010). We were unable to detect differences in soil moisture, vegetation structure or composition (Chapter Two), or food resources for sage grouse chicks or nesting sage thrashers and Brewer’s sparrows in the first two years after treatment (Chapter Three). However, values for canopy coverage, diversity, and evenness often were higher in treatment reaches compared to controls two years after treatment (Chapter Two) and we were able to detect differences in known food resources of nesting vesper sparrows during September, two years after restoration (Chapter Three). Additionally, we observed slowing water and sediment deposits for some structures. These observations suggest more time may be needed to detect change in our system.

Low-tech solutions are a relatively new method of wet meadow restoration (Zeedyk & Clothier 2009) and continued monitoring is crucial to developing our understanding. Comparing findings from the growing number of studies focused on these tools also will help us to understand the factors that may alter efficacy and the time needed to detect effects. Timing of installation and the local environmental conditions may be important influences on the effectiveness of low-tech wet meadow restoration and variation in the time needed to detect effects.

We also are still learning which methods are most appropriate for post-treatment monitoring; our findings provide some new insights. For vegetation, we recommend exploring changes in vegetation biomass, as an alternative response metric. To understand changes in arthropods (as food resources or as another community of interest), focusing on finer taxonomic resolution (e.g., family, genus) could minimize masking responses of disparate groups.

Compared to other restoration methods (Zeedyk & Clothier 2009), the low cost and relative speed of installation make a compelling case for low-tech solutions in wet meadows. If these low-tech solutions do not provide a "cure-all", there is a continued need to find effective and practical tools feasible for widespread use. Climate change, overgrazing, and habitat conversion (Knight et al. 2014), combined with the importance of wet meadows to many plant and wildlife species, suggest an "all hands on deck" approach is needed to restore wet meadows in arid landscapes and improve habitat for sagebrush-associated species (Silverman et al. 2019).

Management Implications

Not detecting effects of wet meadow restoration one or two years post-treatment is an important finding for this relatively new method of restoration. Our results provide valuable information to land managers planning to implement low-tech methods for wet meadow restoration. As with many restoration projects (Ton et al. 1998; Meyer et al. 2010; Hopple & Craft 2013; Jing et al. 2014; Cooper et al. 2017), one-rock-dams and brush dams may not be an instant "cure-all" for wet meadows. They are one of many tools used to restore and conserve wet meadows in sagebrush environments (Zeedyk 2015; Zeedyk & Clothier 2009), and their effectiveness may depend on environmental factors. Evidence of restoration efforts may not materialize for several years, especially for projects implemented in environments similar to southwestern Montana. Post-treatment monitoring is important to detect potential changes, including increases in invasive plants.

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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 models comparing soil volumetric water content for 1 and 2 years post-treatment.

	Variable/Covariate	Estimate	SE	*t*	*P*
1 year post-treatment *df* = 181	Intercept (June - Control)	67.12	10.54	6.37	<0.001
	Relative Distance	-0.02	0.00	-7.01	<0.001
	Average Gully Slope	0.99	1.65	0.60	0.548
	River Left Bank Aspect	0.00	0.03	0.09	0.931
	Reach Width/Height	-0.67	0.19	-3.61	<0.001
	July - Control	-13.02	4.55	-2.86	0.005
	August - Control	-26.24	4.55	-5.76	<0.001
	September - Control	-26.17	5.14	-5.10	<0.001
	June - Treatment	-0.57	4.71	-0.12	0.903
	July - Treatment	-0.07	6.56	-0.01	0.992
	August - Treatment	4.40	6.56	0.67	0.504
	September - Treatment	-1.15	7.29	-0.16	0.875
2 years post-treatment *df* = 39	Intercept (June - Control)	19.43	14.55	1.34	0.189
	Relative Distance	0.03	0.01	2.53	0.015
	Average Gully Slope	-2.70	3.01	-0.90	0.375
	River Left Bank Aspect	0.06	0.09	0.69	0.496
	Reach Width/Height	0.25	0.56	0.45	0.659
	July - Control	29.36	12.03	2.44	0.019
	August - Control	-1.80	12.03	-0.15	0.882
	September - Control	16.56	12.03	1.38	0.177
	June - Treatment	15.38	12.86	1.20	0.239
	July - Treatment	-13.76	17.73	-0.78	0.442
	August - Treatment	-12.98	17.08	-0.76	0.452
	September - Treatment	-15.10	16.53	-0.91	0.367

Table A.2. Estimates (and standard errors, test statistics, and P-values) from models comparing coverage of plant functional groups in treated and control reaches, 1 year post-treatment.

df = 175	Variable/Covariate	Estimate	SE	*t*	*P*
Total Vegetation	Intercept (June - Control)	65.24	7.33	8.91	<0.001
	Relative Distance	0.00	0.00	-0.96	0.337
	Average Gully Slope	-0.62	1.28	-0.48	0.631
	River Left Bank Aspect	0.01	0.02	0.36	0.719
	Reach Width/Height	0.04	0.14	0.27	0.238
	July - Control	4.78	3.88	-0.22	0.788
	August - Control	4.44	3.56	1.25	0.214
	September - Control	6.26	3.93	1.59	0.113
	June - Treatment	-4.03	3.60	-1.12	0.264
	July - Treatment	7.41	5.05	1.47	0.143
	August - Treatment	4.34	5.10	0.85	0.396
	September - Treatment	3.60	5.72	0.63	0.529
Total Green Vegetation	Intercept (June - Control)	69.34	8.05	8.61	<0.001
	Relative Distance	0.00	0.00	-1.89	0.061
	Average Gully Slope	0.73	1.42	-0.51	0.608
	River Left Bank Aspect	0.02	0.02	0.83	0.410
	Reach Width/Height	-0.19	0.16	-1.18	0.238
	July - Control	-0.87	3.88	-0.22	0.823
	August - Control	-19.50	3.93	-4.96	<0.001
	September - Control	-37.11	4.34	-8.55	<0.001
	June - Treatment	-4.08	3.98	-1.03	0.306
	July - Treatment	6.85	5.57	1.23	0.220
	August - Treatment	8.32	5.62	1.48	0.141
	September - Treatment	2.78	6.31	0.44	0.660

Table A.2. Continued.

Native Forbs	Intercept (June - Control)	21.58	4.35	4.97	<0.001
	Relative Distance	0.00	0.00	0.37	0.713
	Average Gully Slope	0.43	0.79	0.55	0.585
	River Left Bank Aspect	0.00	0.01	-0.23	0.816
	Reach Width/Height	-0.16	0.09	-1.80	0.073
	July - Control	1.36	2.19	0.62	0.534
	August - Control	-2.65	2.22	-1.20	0.233
	September - Control	-4.63	2.45	-1.89	0.060
	June - Treatment	0.02	2.25	0.01	0.993
	July - Treatment	-0.94	3.15	-0.30	0.766
	August - Treatment	3.85	3.18	1.21	0.227
	September - Treatment	3.43	3.56	0.96	0.337
Green Native Forbs	Intercept (June - Control)	20.21	4.13	4.90	<0.001
	Relative Distance	0.00	0.00	-0.22	0.826
	Average Gully Slope	0.56	0.74	0.75	0.454
	River Left Bank Aspect	0.00	0.01	0.00	0.997
	Reach Width/Height	-0.14	0.08	-1.69	0.092
	July - Control	0.73	2.06	0.35	0.723
	August - Control	-5.58	2.08	-2.68	0.008
	September - Control	-7.56	2.30	-3.29	0.001
	June - Treatment	-0.16	2.10	-0.08	0.938
	July - Treatment	-1.05	2.95	-0.36	0.722
	August - Treatment	2.28	2.98	0.77	0.444
	September - Treatment	2.98	3.34	0.89	0.372
Rush/Sedge	Intercept (June - Control)	3.43	5.22	0.66	0.512
	Relative Distance	0.00	0.00	-0.01	0.990
	Average Gully Slope	0.55	1.02	0.54	0.592
	River Left Bank Aspect	0.04	0.02	2.00	0.047
	Reach Width/Height	0.15	0.12	1.31	0.192
	July - Control	-2.33	2.89	-0.81	0.420
	August - Control	-1.02	2.92	-0.35	0.727
	September - Control	-3.59	3.22	-1.11	0.266
	June - Treatment	-7.59	2.95	-2.57	0.011
	July - Treatment	4.10	4.14	0.99	0.323
	August - Treatment	4.19	4.18	1.00	0.317
	September - Treatment	0.95	4.69	0.20	0.839

Table A.2. Continued.

Green Rush/Sedge	Intercept (June - Control)	4.91	4.46	1.10	0.271
	Relative Distance	0.00	0.00	0.30	0.764
	Average Gully Slope	0.39	0.87	0.45	0.656
	River Left Bank Aspect	0.03	0.02	2.06	0.040
	Reach Width/Height	0.10	0.10	1.05	0.292
	July - Control	-3.03	2.46	-1.23	0.220
	August - Control	-5.98	2.49	-2.40	0.017
	September - Control	-9.84	2.75	-3.58	0.000
	June - Treatment	-6.86	2.52	-2.73	0.007
	July - Treatment	3.74	3.53	1.06	0.290
	August - Treatment	5.48	3.56	1.54	0.125
	September - Treatment	3.46	4.00	0.87	0.387
Native Grass	Intercept (June - Control)	7.04	3.43	2.05	0.0418
	Relative Distance	0.00	0.00	-3.08	0.002
	Average Gully Slope	-0.24	0.71	-0.33	0.741
	River Left Bank Aspect	0.01	0.01	1.15	0.251
	Reach Width/Height	0.26	0.09	2.94	0.003
	July - Control	2.99	2.21	1.35	0.177
	August - Control	5.02	2.23	2.25	0.025
	September - Control	5.62	2.46	2.29	0.023
	June - Treatment	-1.91	2.25	-0.85	0.397
	July - Treatment	3.38	3.16	1.07	0.286
	August - Treatment	-0.16	3.19	-0.05	0.959
	September - Treatment	6.36	3.58	1.78	0.077
Green Native Grass	Intercept (June - Control)	9.96	2.72	3.66	<0.001
	Relative Distance	0.00	0.00	-2.23	0.026
	Average Gully Slope	-0.53	0.56	-0.95	0.345
	River Left Bank Aspect	0.01	0.01	0.63	0.531
	Reach Width/Height	0.09	0.07	1.35	0.179
	July - Control	1.78	1.69	1.05	0.295
	August - Control	-1.37	1.71	-0.80	0.423
	September - Control	-4.98	1.89	-2.64	0.009
	June - Treatment	-1.95	1.73	-1.13	0.260
	July - Treatment	4.13	2.43	1.70	0.091
	August - Treatment	2.06	2.45	0.84	0.401
	September - Treatment	0.87	2.75	0.32	0.751

Table A.2. Continued.

Non-native Grass	Intercept (June - Control)	15.59	5.35	2.91	0.004
	Relative Distance	0.00	0.00	0.86	0.389
	Average Gully Slope	-0.30	1.08	-0.28	0.780
	River Left Bank Aspect	-0.01	0.02	-0.69	0.489
	Reach Width/Height	-0.08	0.12	-0.65	0.519
	July - Control	2.28	3.12	0.73	0.465
	August - Control	4.05	3.15	1.29	0.200
	September - Control	13.80	3.48	3.97	<0.001
	June - Treatment	1.19	3.19	0.37	0.709
	July - Treatment	4.25	4.47	0.95	0.343
	August - Treatment	-0.52	4.51	-0.11	0.909
	September - Treatment	-3.49	5.06	-0.69	0.491
Green Non-native Grass	Intercept (June - Control)	17.11	4.73	3.62	<0.001
	Relative Distance	0.00	0.00	-1.08	0.281
	Average Gully Slope	-0.15	0.96	-0.16	0.871
	River Left Bank Aspect	0.00	0.02	0.11	0.913
	Reach Width/Height	-0.11	0.11	-1.02	0.311
	July - Control	0.14	2.81	0.05	0.961
	August - Control	-4.30	2.84	-1.52	0.131
	September - Control	-9.22	3.13	-2.94	0.003
	June - Treatment	0.65	2.87	0.23	0.820
	July - Treatment	4.07	4.02	1.01	0.313
	August - Treatment	2.27	4.06	0.56	0.576
	September - Treatment	-0.83	4.56	-0.18	0.856
Non-native Forbs	Intercept (June - Control)	8.46	2.54	3.33	0.001
	Relative Distance	0.00	0.00	0.29	0.775
	Average Gully Slope	-0.74	0.48	-1.55	0.122
	River Left Bank Aspect	-0.01	0.01	-0.73	0.468
	Reach Width/Height	-0.08	0.05	-1.56	0.119
	July - Control	0.65	1.33	0.49	0.623
	August - Control	-1.39	1.34	-1.04	0.301
	September - Control	-4.73	1.48	-3.19	0.001
	June - Treatment	3.34	1.36	2.46	0.014
	July - Treatment	-2.43	1.90	-1.28	0.203
	August - Treatment	-3.65	1.92	-1.90	0.059
	September - Treatment	-3.62	2.15	-1.68	0.094

Table A.2. Continued.

Green Non-native Forbs	Intercept (June - Control)	8.19	2.34	3.51	<0.001
	Relative Distance	0.00	0.00	-0.63	0.532
	Average Gully Slope	-0.65	0.42	-1.56	0.121
	River Left Bank Aspect	0.00	0.01	-0.01	0.990
	Reach Width/Height	-0.08	0.05	-1.70	0.091
	July - Control	-0.53	1.15	-0.46	0.643
	August - Control	-2.88	1.16	-2.47	0.014
	September - Control	-5.56	1.29	4.32	<0.001
	June - Treatment	3.17	1.18	2.69	0.007
	July - Treatment	-2.93	1.65	-1.77	0.077
	August - Treatment	-4.28	1.67	-2.57	0.011
	September - Treatment	-3.45	1.87	-1.85	0.066

Table A.3. Estimates (and standard errors, test statistics, and P-values) from models comparing coverage of plant functional groups in treated and control reaches, 2 years post-treatment.

DF = 39	Variable/Covariate	Estimate	SE	*t*	*P*
Total Vegetation	Intercept (June - Control)	74.61	8.95	8.34	<0.001
	Relative Distance	0.01	0.01	1.32	0.195
	Average Gully Slope	-2.03	1.69	-1.20	0.236
	River Left Bank Aspect	0.03	0.05	0.65	0.521
	Reach Width/Height	-0.22	0.34	-0.65	0.522
	July - Control	8.77	5.37	1.64	0.110
	August - Control	-1.08	5.37	-0.20	0.841
	September - Control	-6.39	6.27	-1.02	0.314
	June - Treatment	13.28	5.41	2.45	0.019
	July - Treatment	-6.81	8.26	-0.82	0.415
	August - Treatment	-9.55	7.91	-1.21	0.234
	September - Treatment	-6.28	8.46	-0.74	0.463
Total Green Vegetation	Intercept (June - Control)	72.19	8.31	8.69	<0.001
	Relative Distance	0.01	0.01	1.89	0.066
	Average Gully Slope	-0.95	1.57	-0.61	0.546
	River Left Bank Aspect	0.02	0.04	0.48	0.634
	Reach Width/Height	-0.25	0.32	-0.77	0.444
	July - Control	8.60	4.99	1.73	0.092
	August - Control	-16.58	4.99	-3.33	0.002
	September - Control	-46.66	5.82	-8.02	<0.001
	June - Treatment	11.71	5.03	2.33	0.025
	July - Treatment	-8.01	7.68	-1.04	0.303
	August - Treatment	-10.87	7.35	-1.48	0.147
	September - Treatment	-12.55	7.86	-1.60	0.118

Table A.3. Continued.

Native Forbs	Intercept (June - Control)	44.62	5.17	8.63	<0.001
	Relative Distance	-0.03	0.01	-5.23	<0.001
	Average Gully Slope	4.75	1.46	3.26	0.002
	River Left Bank Aspect	-0.12	0.04	-3.10	0.004
	Reach Width/Height	-0.19	0.24	-0.82	0.416
	July - Control	0.50	4.83	0.10	0.918
	August - Control	-10.67	4.83	-2.21	0.033
	September - Control	-18.22	5.52	-3.30	0.002
	June - Treatment	6.07	4.87	1.25	0.220
	July - Treatment	-5.53	7.38	-0.75	0.458
	August - Treatment	2.03	7.11	0.29	0.777
	September - Treatment	-3.19	7.58	-0.42	0.676
Green Native Forbs	Intercept (June - Control)	41.45	5.25	7.90	<0.001
	Relative Distance	-0.02	0.01	-4.27	<0.001
	Average Gully Slope	4.24	1.48	2.87	0.007
	River Left Bank Aspect	-0.10	0.04	-2.49	0.017
	Reach Width/Height	-0.19	0.24	-0.78	0.439
	July - Control	0.33	4.89	0.07	0.946
	August - Control	-15.75	4.89	-3.22	0.003
	September - Control	-27.46	5.60	-4.90	0.000
	June - Treatment	6.46	4.94	1.31	0.198
	July - Treatment	-6.99	7.49	-0.93	0.356
	August - Treatment	-1.81	7.21	-0.25	0.804
	September - Treatment	-7.19	7.69	-0.94	0.355
Rush/Sedge	Intercept (June - Control)	16.93	9.89	-1.71	0.095
	Relative Distance	0.04	0.01	8.57	0.000
	Average Gully Slope	-3.66	1.53	-2.39	0.022
	River Left Bank Aspect	0.18	0.04	4.22	<0.001
	Reach Width/Height	0.09	0.33	0.28	0.779
	July - Control	-1.00	4.85	-0.21	0.838
	August - Control	0.13	4.85	0.03	0.980
	September - Control	0.23	5.68	0.04	0.968
	June - Treatment	-0.08	4.90	-0.02	0.987
	July - Treatment	-3.39	7.47	-0.45	0.653
	August - Treatment	-5.76	7.15	-0.81	0.425
	September - Treatment	-2.25	7.66	-0.29	0.770

Table A.3. Continued.

Green Rush/Sedge	Intercept (June - Control)	-11.83	8.38	-1.41	0.166
	Relative Distance	0.04	0.00	8.44	<0.001
	Average Gully Slope	-2.92	1.29	-2.26	0.030
	River Left Bank Aspect	0.15	0.04	4.24	<0.001
	Reach Width/Height	0.01	0.28	0.04	0.966
	July - Control	-1.00	4.09	-0.25	0.808
	August - Control	-1.29	4.09	-0.32	0.754
	September - Control	-7.60	4.79	-1.59	0.120
	June - Treatment	-0.37	4.13	-0.09	0.929
	July - Treatment	-2.96	6.30	-0.47	0.641
	August - Treatment	-5.15	6.03	-0.86	0.398
	September - Treatment	-0.09	6.46	-0.01	0.989
Native Grass	Intercept (June - Control)	9.44	5.21	1.81	0.078
	Relative Distance	-0.01	0.00	-1.88	0.068
	Average Gully Slope	0.22	1.33	0.17	0.869
	River Left Bank Aspect	0.00	0.04	-0.10	0.922
	Reach Width/Height	0.09	0.23	0.42	0.681
	July - Control	1.63	4.35	0.37	0.711
	August - Control	6.79	4.35	1.56	0.126
	September - Control	9.89	5.02	1.97	0.056
	June - Treatment	-2.89	4.39	-0.66	0.514
	July - Treatment	0.21	6.67	0.03	0.975
	August - Treatment	4.73	6.41	0.74	0.464
	September - Treatment	-1.41	6.84	-0.21	0.837
Green Native Grass	Intercept (June - Control)	10.88	3.98	2.73	0.009
	Relative Distance	-0.01	0.00	-1.51	0.140
	Average Gully Slope	0.06	1.03	0.06	0.954
	River Left Bank Aspect	0.02	0.03	-0.62	0.539
	Reach Width/Height	0.03	0.17	0.20	0.845
	July - Control	1.63	3.35	0.49	0.630
	August - Control	0.46	3.35	0.14	0.892
	September - Control	-1.11	3.87	-0.29	0.776
	June - Treatment	-2.99	3.38	-0.88	0.382
	July - Treatment	0.18	5.14	0.04	0.972
	August - Treatment	7.80	4.94	1.58	0.122
	September - Treatment	-0.45	5.27	-0.09	0.932

Table A.3. Continued.

Non-native Grass	Intercept (June - Control)	14.30	5.18	2.76	0.009
	Relative Distance	-0.01	0.01	-1.66	0.105
	Average Gully Slope	-0.92	1.46	-0.63	0.531
	River Left Bank Aspect	-0.04	0.04	-0.97	0.339
	Reach Width/Height	0.30	0.24	1.28	0.208
	July - Control	1.98	4.83	0.41	0.684
	August - Control	-4.13	4.83	-0.85	0.399
	September - Control	10.63	5.53	1.92	0.062
	June - Treatment	3.16	4.87	0.65	0.520
	July - Treatment	6.84	7.40	0.93	0.360
	August - Treatment	2.79	7.12	0.39	0.698
	September - Treatment	7.12	7.59	0.94	0.354
Green Non-native Grass	Intercept (June - Control)	11.94	4.09	2.92	0.006
	Relative Distance	-0.01	0.00	-1.49	0.145
	Average Gully Slope	-0.13	1.08	-0.12	0.902
	River Left Bank Aspect	-0.03	0.03	-1.05	0.302
	Reach Width/Height	0.24	0.18	1.32	0.194
	July - Control	1.98	3.54	0.56	0.579
	August - Control	-5.92	3.54	-1.67	0.102
	September - Control	-0.75	4.08	-0.18	0.856
	June - Treatment	3.00	3.57	0.84	0.406
	July - Treatment	6.18	5.43	1.14	0.262
	August - Treatment	0.72	5.21	0.14	0.892
	September - Treatment	-0.10	5.56	-0.02	0.985
Non-native Forbs	Intercept (June - Control)	6.02	2.97	2.03	0.050
	Relative Distance	0.00	0.00	0.38	0.703
	Average Gully Slope	-1.01	0.51	-1.98	0.055
	River Left Bank Aspect	0.01	0.01	0.83	0.409
	Reach Width/Height	-0.06	0.11	-0.60	0.553
	July - Control	2.19	1.62	1.35	0.185
	August - Control	-1.79	1.62	-1.10	0.276
	September - Control	-3.01	1.90	-1.59	0.121
	June - Treatment	3.66	1.64	2.24	0.031
	July - Treatment	-3.24	2.50	-1.30	0.202
	August - Treatment	-1.49	2.39	-0.62	0.537
	September - Treatment	-3.65	2.56	-1.43	0.162

Table A.3. Continued.

Green Non-native Forbs	Intercept (June - Control)	5.34	2.83	1.89	0.067
	Relative Distance	0.00	0.00	0.28	0.781
	Average Gully Slope	-0.86	0.51	-1.69	0.099
	River Left Bank Aspect	0.01	0.01	0.75	0.457
	Reach Width/Height	-0.03	0.11	-0.27	0.787
	July - Control	2.19	1.61	1.36	0.181
	August - Control	-2.29	1.61	-1.43	0.162
	September - Control	-3.15	1.88	-1.68	0.102
	June - Treatment	3.59	1.62	2.21	0.033
	July - Treatment	-3.38	2.48	-1.36	0.180
	August - Treatment	-2.22	2.37	-0.94	0.353
	September - Treatment	-3.58	2.54	-1.41	0.166

Table A.4. Estimates (and standard errors, test statistics, and P-values) from models comparing species richness, Shannon-Weiner diversity, and species evenness in treated and control reaches, 1 year post-treatment.

df = 175	Variable/Covariate	Estimate	SE	*t*	*P*
Species Richness	Intercept (June - Control)	23.57	3.17	7.43	<0.001
	Relative Distance	0.00	0.00	1.44	0.152
	Average Gully Slope	-1.27	0.46	-2.76	0.006
	River Left Bank Aspect	0.01	0.01	1.19	0.235
	Reach Width/Height	-0.12	0.05	-2.43	0.016
	July - Control	1.58	1.24	1.27	0.205
	August - Control	-2.86	1.25	-2.29	0.024
	September - Control	-4.56	1.38	-3.30	0.001
	June - Treatment	-1.52	1.27	-1.20	0.233
	July - Treatment	-1.08	1.78	-0.61	0.546
	August - Treatment	0.31	1.79	0.17	0.864
	September - Treatment	0.32	2.01	0.16	0.874
Shannon-Weiner Diversity	Intercept (June - Control)	1.58	0.23	6.93	<0.001
	Relative Distance	0.00	0.00	0.39	0.701
	Average Gully Slope	-0.11	0.04	-2.94	0.004
	River Left Bank Aspect	0.00	0.00	2.27	0.025
	Reach Width/Height	-0.01	0.00	-1.80	0.073
	July - Control	0.15	0.10	1.50	0.137
	August - Control	0.09	0.10	0.86	0.391
	September - Control	0.00	0.11	0.01	0.989
	June - Treatment	-0.16	0.10	-1.55	0.123
	July - Treatment	-0.07	0.14	-0.47	0.640
	August - Treatment	-0.08	0.14	-0.55	0.580
	September - Treatment	-0.06	0.16	-0.38	0.708

Table A.4. Continued.

Species Evenness	Intercept (June - Control)	49.72	5.87	8.47	<0.001
	Relative Distance	0.00	0.00	-0.67	0.505
	Average Gully Slope	-2.83	1.04	-2.72	0.007
	River Left Bank Aspect	0.04	0.02	2.44	0.016
	Reach Width/Height	-0.07	0.12	-0.61	0.546
	July - Control	3.23	2.85	1.13	0.259
	August - Control	5.98	2.89	2.07	0.040
	September - Control	7.64	3.19	2.40	0.018
	June - Treatment	-6.07	2.92	-2.08	0.039
	July - Treatment	-0.26	4.09	-0.06	0.950
	August - Treatment	-2.18	4.13	-0.53	0.597
	September - Treatment	-1.57	4.63	-0.34	0.735

Table A.5. Estimates (and standard errors, test statistics, and P-values) from models comparing species richness, Shannon-Weiner diversity, and species evenness in treated and control reaches, 2 years post-treatment.

df = 39	Variable/Covariate	Estimate	SE	*t*	*P*
Species Richness	Intercept (June - Control)	34.06	3.24	10.51	<0.001
	Relative Distance	-0.01	0.00	-3.19	0.003
	Average Gully Slope	-1.56	0.89	-1.76	0.086
	River Left Bank Aspect	0.00	0.02	-0.06	0.951
	Reach Width/Height	-0.14	0.15	-0.95	0.346
	July - Control	-2.50	2.92	-0.86	0.397
	August - Control	-11.00	2.92	-3.77	0.001
	September - Control	-15.26	3.35	-4.55	0.000
	June - Treatment	-3.96	2.95	-1.35	0.186
	July - Treatment	6.78	4.47	1.52	0.138
	August - Treatment	4.81	4.30	1.12	0.271
	September - Treatment	4.15	4.59	0.90	0.372
Shannon-Weiner Diversity	Intercept (June - Control)	2.11	0.30	6.95	<0.001
	Relative Distance	0.00	0.00	-2.12	0.040
	Average Gully Slope	-0.16	0.06	-2.72	0.010
	River Left Bank Aspect	0.00	0.00	1.10	0.278
	Reach Width/Height	-0.01	0.01	-1.02	0.315
	July - Control	0.04	0.18	0.24	0.810
	August - Control	-0.45	0.18	-2.46	0.019
	September - Control	-0.51	0.21	-2.41	0.021
	June - Treatment	-0.09	0.18	-0.51	0.617
	July - Treatment	0.36	0.28	1.27	0.211
	August - Treatment	0.25	0.27	0.95	0.349
	September - Treatment	0.12	0.29	0.41	0.683

Table A.5. Continued.

Species Evenness	Intercept (June - Control)	57.73	8.53	6.77	<0.001
	Relative Distance	0.00	0.01	-0.66	0.512
	Average Gully Slope	-5.14	1.48	-3.49	0.001
	River Left Bank Aspect	0.08	0.04	2.09	0.044
	Reach Width/Height	-0.27	0.31	-0.86	0.395
	July - Control	3.39	4.68	0.73	0.473
	August - Control	-5.50	4.68	-1.18	0.246
	September - Control	-0.62	5.47	-0.11	0.911
	June - Treatment	-1.16	4.72	-0.25	0.807
	July - Treatment	6.39	7.20	0.89	0.380
	August - Treatment	5.08	6.89	0.74	0.466
	September - Treatment	0.59	7.38	0.08	0.937

APPENDIX B

SUPPLEMENTARY TABLES FOR CHAPTER THREE

Table B.1: Estimates (and standard errors, test statistics, and P-values) from models comparing food resources of sage grouse chicks and nesting sage thrashers, Brewer's sparrows, and vesper sparrows) in treated and control reaches, 1 year post-treatment.

		Variable/Covariate	Estimate	SE	*t*	*P*
Sage grouse	Arthropods	Intercept (June - Control)	-574.82	281.89	-2.04	0.044
	df = 105	Relative Distance	0.21	0.07	2.99	0.004
		Average Gully Slope	19.17	46.76	0.41	0.683
		River Left Bank Aspect	0.97	0.97	1.00	0.320
		Reach Width/Height	35.23	7.15	4.93	0.000
		July - Control	177.04	173.58	1.02	0.310
		August - Control	865.98	173.58	4.99	<0.001
		June - Treatment	-170.72	181.90	-0.94	0.350
		July - Treatment	-43.03	244.98	-0.18	0.861
		August - Treatment	169.37	246.710	0.69	0.494
	Plants	Intercept (June - Control)	4.10	1.71	2.40	0.018
	df = 144	Relative Distance	0.00	0.00	3.03	0.003
		Average Gully Slope	0.61	0.34	1.80	0.075
		River Left Bank Aspect	0.00	0.01	-0.60	0.552
		Reach Width/Height	0.04	0.05	0.92	0.360
		July - Control	0.63	1.05	0.60	0.548
		August - Control	0.81	1.09	0.74	0.459
		September - Control	1.00	1.52	0.66	0.513
		June - Treatment	0.17	1.04	0.16	0.873
		July - Treatment	-0.76	1.48	-0.51	0.610
		August - Treatment	1.14	1.54	0.74	0.458
		September - Treatment	0.43	2.11	0.21	0.837
Sage Thrasher and Brewer's Sparrow	Arthropods	Intercept (June - Control)	-470.49	291.92	-1.61	0.110
	df = 106	Relative Distance	0.20	0.07	2.77	0.007
		Average Gully Slope	-8.27	48.17	-0.17	0.864
		River Left Bank Aspect	1.17	1.01	1.16	0.247
		Reach Width/Height	28.10	6.94	4.05	<0.001
		July - Control	330.65	177.12	1.87	0.065
		August - Control	958.52	177.12	5.41	<0.001
		June - Treatment	-105.99	186.02	-0.57	0.570
		July - Treatment	-156.68	250.86	-0.63	0.534
		August - Treatment	89.92	253.60	0.36	0.724

Table B.1. Continued

Vesper Sparrow	Arthropods *df* = 106	Intercept (June - Control)	-500.35	296.47	-1.69	0.094
		Relative Distance	0.21	0.07	2.83	0.006
		Average Gully Slope	7.30	48.92	0.15	0.882
		River Left Bank Aspect	1.34	1.02	1.31	0.192
		Reach Width/Height	27.51	7.04	3.91	<0.001
		July - Control	317.91	179.88	1.77	0.080
		August - Control	1001.40	179.88	5.57	<0.001
		June - Treatment	-52.32	188.92	-0.28	0.782
		July - Treatment	-215.91	254.76	-0.85	0.399
		August - Treatment	16.23	257.55	0.06	0.950
Vesper Sparrow	Plants *df* = 159	Intercept (June - Control)	13.71	4.01	3.42	0.001
		Relative Distance	0.00	0.00	0.21	0.835
		Average Gully Slope	0.23	0.83	0.28	0.778
		River Left Bank Aspect	0.01	0.01	0.71	0.477
		Reach Width/Height	-0.03	0.09	-0.27	0.789
		July - Control	-0.54	2.43	-0.22	0.826
		August - Control	0.92	2.46	0.37	0.710
		September - Control	11.51	2.69	4.28	<0.001
		June - Treatment	-0.16	2.57	-0.06	0.950
		July - Treatment	2.13	3.49	0.61	0.543
		August - Treatment	0.62	3.54	0.18	0.860
		September - Treatment	2.75	3.91	0.70	0.483

Table B.2. Estimates (and standard errors, test statistics, and P-values) from models comparing food resources of sage grouse chicks and nesting sage thrashers, Brewer's sparrows, and vesper sparrows) in treated and control reaches, 2 years post-treatment..

		Variable/Covariate	Estimate	SE	*t*	*P*
Sage Grouse	Arthropods *df* = 30	Intercept (June - Control)	275.46	227.60	1.21	0.236
		Relative Distance	-0.24	0.22	-1.12	0.271
		Average Gully Slope	7.06	64.87	0.11	0.914
		River Left Bank Aspect	0.54	1.78	0.31	0.762
		Reach Width/Height	-5.76	10.86	-0.53	0.600
		July - Control	128.89	195.96	0.66	0.516
		August - Control	563.47	195.96	2.88	0.007
		June - Treatment	-67.40	198.03	-0.34	0.736
		July - Treatment	24.53	300.64	0.08	0.936
		August - Treatment	-447.70	288.85	-1.55	0.132
	Plants *df* = 33	Intercept (June - Control)	2.11	2.35	0.90	0.375
		Relative Distance	0.00	0.00	1.17	0.250
		Average Gully Slope	-0.16	0.66	-0.24	0.814
		River Left Bank Aspect	0.02	0.02	1.26	0.216
		Reach Width/Height	0.08	0.10	0.76	0.456
		July - Control	0.76	2.23	0.34	0.735
		August - Control	1.83	2.23	0.82	0.417
		September - Control	0.95	2.65	0.36	0.723
		June - Treatment	4.03	2.18	1.85	0.074
		July - Treatment	-4.76	3.29	-1.45	0.158
		August - Treatment	-5.09	3.18	-1.60	0.119
		September - Treatment	-7.18	3.68	-1.95	0.060
Sage Thrasher and Brewer's Sparrow	Arthropods *df* = 30	Intercept (June - Control)	377.47	243.39	1.55	0.131
		Relative Distance	-0.28	0.23	-1.24	0.225
		Average Gully Slope	-3.75	69.38	-0.05	0.957
		River Left Bank Aspect	0.94	1.90	0.63	0.625
		Reach Width/Height	-9.84	11.62	-0.85	0.404
		July - Control	180.12	209.56	0.86	0.397
		August - Control	613.27	209.56	2.93	0.007
		June - Treatment	-58.68	211.77	-0.28	0.784
		July - Treatment	93.21	321.50	0.29	0.774
		August - Treatment	-458.76	308.89	-1.49	0.148

Table B.2. Continued

Vesper Sparrow	Arthropods *df* = 30	Intercept (June - Control)	400.19	248.27	1.61	0.118
		Relative Distance	-0.30	0.23	-1.28	0.210
		Average Gully Slope	2.10	70.77	0.03	0.977
		River Left Bank Aspect	0.90	1.94	0.47	0.645
		Reach Width/Height	-8.30	11.85	-0.70	0.489
		July - Control	149.64	213.76	0.70	0.489
		August - Control	600.82	213.76	2.81	0.009
		June - Treatment	-71.69	216.02	-0.33	0.742
		July - Treatment	99.32	327.95	0.30	0.764
		August - Treatment	-467.31	315.08	-1.48	0.149
	Plants *df* = 37	Intercept (June - Control)	4.86	4.26	1.14	0.261
		Relative Distance	0.00	0.00	0.17	0.866
		Average Gully Slope	-0.09	1.25	-0.07	0.945
		River Left Bank Aspect	-0.01	0.03	-0.23	0.819
		Reach Width/Height	0.32	0.19	1.67	0.104
		July - Control	2.07	3.89	0.53	0.598
		August - Control	0.48	3.89	0.12	0.902
		September - Control	13.54	4.45	3.05	0.004
		June - Treatment	0.11	4.07	0.03	0.979
		July - Treatment	0.60	6.05	0.10	0.922
		August - Treatment	5.91	6.05	0.98	0.335
		September - Treatment	16.11	6.19	2.60	0.013