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Hydrologic and Biologic Responses of Anthropogenically Altered Lentic Springs to Restoration in the Great Basin

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ABSTRACT

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Water is a limited and highly valued resource in the semi-arid Great Basin. Surface water sources are often small and widely spaced apart, comprising only 1-3% of the surface area of the overall landscape. Despite their small size, these springs and surrounding wet meadows have a substantial effect on the surrounding environment. Springs provide drinking water, forage and cover for livestock and wildlife, habitat for diversity of plant species and a resource for human-related activities. In recent years, many of these springs have become dewatered due to diversions of groundwater for municipal water and agriculture, and climatic shifts in precipitation affecting recharge. These hydrologic changes can cause a drop in the local water table that promotes a shift in the plant community from wetland-obligates to species that have more drought-tolerance. The root masses of the new plant community are insufficient to secure soils resulting in the erosion of the thalweg. This leads to channelization through the wet meadow, which drives the water table further underground. As degradation progresses, springs and wet meadows lose their ability to store water. The purpose of this thesis is to examine the responses of both the hydrologic and biologic factors to different springbox restoration techniques. Twenty-four spring sites were chosen in the Sheldon National Wildlife Refuge in northwestern Nevada. Each site was randomly assigned one of six different treatment designs. Variables for these studies included: surface soil moisture, soil moisture at varying depths, flow rates, water chemistry, plant community cover and frequency, biomass, wildlife visits and wildlife species numbers. We observed soil moisture increase over the majority of our sites, while flow rates only increased at the control sites. This may indicate that more water is being held in the soils around the spring source instead of being allowed to flow downstream. Biomass increased in four of our six treatments. All treatment types exhibited a similar effect on springs with none having a clearly more restorative effective than any others. This research suggests that springs in the Great Basin have unique characteristics and responses to restoration, and may need individualized approaches. Additionally, studies have shown that it may take many years for plant communities to recover after hydrologic restoration. Yearly variation caused by increased precipitation may be partially responsible for changes in hydrologic and biologic aspects of springs and wet meadows. Further data collection is needed to determine the true extent of treatment and yearly effects on spring restoration. In spite of the need for individualized approaches, restoration is possible. Simple solutions may be sufficient to recover hydrologic processes that maintain ecologic resilience.

Keywords: lentic springs, restoration, soil moisture, Great Basin, wet meadow, riparian zone

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CHAPTER 1

Hydrologic Response of Anthropogenically Altered Lentic Springs in the Great Basin

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ABSTRACT

Water in the Great Basin is a limited but valuable commodity for wildlife, plants and humanity. There are few naturally occurring surface water sources in the Great Basin, USA. Most of the water in this region comes in the form of small springs and seeps. Spring structure formation depends on the underlying geology, which influences both the amount and seasonality of discharged water. Great Basin springs develop from two lithologies: basin-filled and consolidated rock. Basin-filled springs are supplied by deep aquifers that produce consistent discharge. Consolidated rock springs are more varied, forming both multi-valley and single-valley closed systems. While both are recharged by annual precipitation, single and multi-valley systems are influenced to a greater degree by local weather patterns. The perpetuation of springs in the Great Basin is being threatened by multiple factors. Shifts in precipitation threaten the recharge of local aquifers. Water developments, such as springboxes used for livestock and agriculture, drain underlying aquifers, dewatering the spring and its riparian corridor, reducing or eliminating surface flow patterns. These modifications result in springs and wet meadows that can no longer store water in riparian soils, thus, degrading entire sites. Degradation is characterized by incised channel formation, altered plant communities and increased water lost from the soil due to evaporation. The purpose of this study is to examine the effect of different restoration techniques on the hydrology of springs and surrounding wet meadows. We selected twenty-four spring sites in the Sheldon National Wildlife Refuge in northwestern Nevada and implemented six randomly assigned treatment designs. Surface soil moisture, soil moisture at varying depths, flow rates and water chemistry were measured for this study. We observed an increase in percent soil moisture across all treatment types at the majority of our spring sites. Increased soil moisture indicates that water is being held in the riparian zone, which allows water to seep back into the soil, recharging the underlying water table. Water retention is one of the functions of a healthy spring ecosystem. The amount of flow increased at our control sites but not at any treatment. We did not determine that any of our treatments were more effective than another at restoring the hydrology of a spring. Springs in the Great Basin are very distinct and individual entities in how they respond to disturbance and restoration. Tailored restoration for each spring site is necessary to address its unique characteristics and hydrology. Yearly variation caused by increased precipitation may be partially responsible for changes in hydrology. Further data collection is needed to determine the true extent of treatment and yearly effects on spring restoration.

INTRODUCTION

Water is a limited but highly valued commodity in the semiarid sagebrush-steppe of North America's Great Basin region (Barquin and Scarsbrook 2008, Chambers and Miller 2004, Wyman et al. 2006). Ringed by the Colorado plateau, Sierra Mountains, Mojave Desert and Snake River Drainage, this geologically wrinkled landscape of basins and ranges stretches across 520,000 square kilometers of Nevada, Utah, California, Idaho and Arizona (Plume and Carlton 1988). Despite its size, the Great Basin contains few naturally occurring surface water sources. This region receives approximately 12 cm of annual precipitation on the basin floor and up to 50 cm at higher elevations, most of which comes in the form of snow. Rain in the summer and fall is limited (Swanston 1991, Welsch et al. 1995). All of the precipitation that falls in the Great

Basin either drains toward terminal water bodies (i.e. playa), infiltrates into soils where it recharges groundwater voids, is lost from the soil through evaporation, or leaves plant tissues through transpiration (Sada 2008, Lewis et al. 2003).

Water that percolates within the soil and rock matrix can potentially make its way into one of many regional aquifers, the majority of which have the ability to produce at least a small amount of water during the year (Maurer et al. 2004, Plume and Carlton 1988). Many of these aquifers began forming hundreds of thousands of years ago, primarily during the Precambrian and have continued to change and develop into the water sources that they are currently (Plume and Carlton 1988). Great Basin aquifers are formed from one of two different lithologic sources: basin-filled deposits or consolidated rock (Maurer et al. 2004, Plume and Carlton 1988). Aquifers formed beneath basin-filled deposits tend to be large with the ability to store and transport massive quantities of water. These are generally productive and discharge water at a relatively consistent annual rate. Aquifers that form in and around consolidated rock are much less productive (Plume and Carlton 1988, Maurer et al. 2004). This lithologic type can be further categorized as either 1) carbonate sedimentary rock or 2) volcanic, granite, clastic (volcanic) rock. Carbonate sedimentary rock, which makes consolidated rock formations on the eastern side of the Basin, is more porous in nature and tends to create extensive, deep aquifers. Water sources located within these two aquifer types are often hydraulically connected. Groundwater is able to flow between drainages, forming sprawling multi-valley systems. The other type, volcanic consolidated rock, retains and transports the least amount of water due to its extremely non-permeable nature. This type of geology generally acts as a barrier to groundwater flow (Maurer et al. 2004, Plume and Carlton 1988, Lewis et al. 2003). Where carbonate rock encourages the creation of multi-valley interchange of groundwater, volcanic rock seals off aquifers forming closed single-valley systems (Plume and Carlton 1988). The only water discharged from these systems is collected from precipitation that flows off the surrounding uplands. There is little to no input from other groundwater sources (Plume and Carlton 1988, Maurer et al. 2004).

The majority of water sources for the Great Basin come from small, isolated springs that are dependent on the limited water sources provided by deep regional aquifers and/or local watershed recharge (Patten et al. 2008). These springs represent the only source of water available (Sada 2008) and subsequently have a disproportionate effect relative to their size on the surrounding landscape (Chambers and Miller 2004). They form pockets of high biodiversity within a more xeric landscape, including refugia for plants and wildlife that depend on higher water availability (Sada 2008, Wyman et al. 2006, Naiman et al. 1993). Additionally, vegetation can provide high-quality forage, especially during late summer and fall months when upland vegetation is limited in quality and quantity (Wyman et al. 2006, Lewis et al. 2003).

In recent years, springs throughout the Great Basin have become threatened on dual fronts: diminished inputs and reduced outputs. Precipitation provides the primary input for springs. Snowmelt provides the majority of the moisture available for riparian zones and streams (Swanston 1991), and the groundwater feeding these springs is heavily influenced by annual snowpack. Long-term shifts in precipitation patterns threaten the sustainability of springs and wet meadows throughout the Great Basin (Sada 2009, Chambers and Miller 2004). A climate that shifts away from snowfall toward spring and fall rain could affect annual aquifer recharge and potentially annual and seasonal spring flow rates (Sada 2008). More water is lost to spring runoff. Runoff can saturate soils, but does not last long enough for water to percolate through the soil profile to recharge local aquifers (Lewis et al. 2003). Such a shift in precipitation from snow to rain could potentially reduce spring discharge levels and even threaten the persistence of springs over time (Adam et al. 2009).

In addition to insufficient groundwater supply and recharge leading to earlier drying patterns (Lewis et al. 2003, Wyman et al. 2006), spring output levels are also at risk from human-related activities (i.e. irrigation, recreation, animal production) (Wyman et al. 2006, Sada et. al 2001). Excessive use of water and subsequent lowering of the water table, especially those feeding single-valley closed systems, results in the draw-down of the capillary fringe (portion of the soil profile that is not directly in contact with the water table, but experiences increased soil moisture). This can result in dewatering of the surrounding wet meadow and cause shrunken riparian zone areas (Lewis et al. 2003, Patten et al. 2008).

Dewatering results in a snowball effect that leads to altered plant community characteristics (Castelli et al. 2000, Perkins et al. 1984, Wyman et al. 2006, Chambers and Miller 2004). For example, juniper encroachment (and other woody species) can be an unintended consequence of an altered community (Wyman et al. 2006). Evapotranspiration from junipers around a wet meadow can greatly contribute to dewatering the meadow. In pinyon-juniper woodlands, 80 to 95 percent of input from precipitation is estimated to be lost due to evapotranspiration (Weltz 1987, Carlson et al. 1990). The additive effects of anthropogenic disturbance and climate change can accelerate the degradation of riparian ecosystems (Chambers and Miller 2004).

With the advent of grazing on public lands in the arid west, many springs and seeps were subjected to water developments (Fleischner 1994). This often consisted of a springbox (or headbox) installed in the riparian zone to capture water and then transport it to troughs for livestock. Such water development have resulted in degradation and dewatering of springs and wet meadows (Sada 2008, Fleischner 1994). Springboxes placed at the spring source, in particular, could cause a form of single-point incision. Single-point incision acts similarly to stream incision by lowering the “stream surface” to an elevation below the surrounding water table. Just as in stream incision, this draws down the base level of the surrounding groundwater, which alters the overall water table (Wyman et al. 2006, Chambers and Miller 2004). Drawing water away to a trough decreases the abundance of water available to the wet meadow (Patten et al. 2008, Barquin and Scarsbrook 2008). Many springboxes have overflow pipes that release excess water into the riparian zone, often several meters away from the spring source. Additionally, the water is released from a pipe opening which facilitates the formation of flow channels, which in turn restricts sheet flow. Therefore, single-point incision initially causes a drop in the water table, which is further exacerbated by the formation of a channel through the riparian corridor. The channel leads to more incision, driving the water table deeper underground. The wet meadow subsequently becomes dewatered and disconnected from its flood plain (Wyman et al. 2006, Lewis et al. 2003).

Dewatering of wet meadows leads to riparian systems that can no longer provide vital ecological functions (Wyman et al. 2006, Sada 2008). The Sheldon National Wildlife Refuge, where our study is located, has 130 identified springs and 183 water developments on many of those sites. The majority of water developments in SNWR have been abandoned. These include reservoirs, dugouts, berms, and springboxes. In conjunction with over-grazing by non-native ungulates, many of these water developments result in the dewatering and lowering the water table of springs and adjoining wet meadows (USFWS 2012, Sada 2008).

Water developments, such as piping and diversion, can lead to reduced and even eliminated spring flows (Erman 2002). Springs in this condition no longer store water, trap sediments, recharge local aquifers or provide forage and drinking water for wildlife (Wyman et al. 2006, Lewis et al. 2003). Restoring the underlying hydrology by removing an unused water development structure that impedes the natural movement of water will allow the water table to reconnect with the floodplain of the wet meadow (Sada 2008). Recovery of the hydrologic structure of springs is necessary for recolonization of natural wet meadow plant communities (Barquin and Scarsbrook 2008, Schumm 1977, Jensen et al. 1989), and a restoration of the ecosystem services they provide (Sada 2008). Restoration will benefit wildlife species reliant on wet meadows for food, cover, and drinking water (Oakley et al. 1985). Restoration will improve the quality of riparian zone, which strongly influences wildlife usage (Stevens et al. 1997).

The purpose of our study is to assess the influence of different springbox restoration techniques on hydrology of springs and adjoining wet meadows in Sheldon National Wildlife Refuge. Specifically, we will characterize spring flow, water chemistry and soil moisture responses to springbox removal and soil treatment. We hypothesize that treatment designs that mimic the natural hydrology and structure of pre-disturbance springs will be more effective at facilitating hydrologic restoration in degraded springs and wet meadows.

We predict that springs exhibiting hydrologic restoration will reach three milestones. First, surface soil moisture will increase as more water is discharged at ground level in sheet flows. Riparian soils will act as sponges, soaking in excess discharge resulting in increased soil moisture (Lewis et al. 2003). Second, higher flow rates will occur as water previously diverted to troughs returns to the main flow being discharged into the wet meadow (Wyman et al. 2006, Barquin and Scarsbrook 2008, Sada 2008). Finally, as water is held in saturated riparian soils, it will percolate back through the soil profile and recharge underlying local aquifers (Lewis et al. 2003). This will lead to a rising water table. A rising water table facilitates the recolonization and establishment of wetland-obligate plant communities dominated by sedges and rushes (Castelli et al. 2000, Hammersmark et al. 2009, Cowley 1997).

METHODS

Site Description

The Sheldon National Wildlife Refuge is located in the northwestern corner of Nevada (Figure 1.1), straddling Humboldt and Washoe counties (center point 41.806413, -119.232577). The elevation ranges from 1326 to 2183 meters ABS (Collins 2016). The majority of the SNWR lies on the Columbia Plateau basalt shelf formed during the Holocene and Eocene epochs (USFWS 2012, Plume and Carlton 1988). This basalt covers 210,000 km^2 of western Oregon and Washington, eastern Idaho, and northern Nevada. Basalt forms from volcanic lava producing highly porous substrate, considered the most productive of volcanic rock aquifers (Plume and Carlton 1988). Dominant vegetation consists of a low salt-desert shrubland, primarily greasewood (*Sarcobatus vermiculatus*; 1326 m) on the northeastern corner of SNWR before sharply rising onto a basalt-shelf plateau consisting of sagebrush-steppe shrubland (Rodgers and Tiehm 1979). Higher elevations support aspen (*Populus tremuloides*) stands, western juniper (*Juniperus occidentalis*) woodlands and curly-leaf mountain mahogany (*Cercocarpus ledifolius*) forests (Collins 2016).

The lithology of SNWR is almost completely consolidated rock of volcanic origin. Hydrologically-closed or single-valley systems are common for this type of aquifer. These systems are recharged by snowmelt infiltration (Plume and Carlton 1988, Plume and Carlton 1988, USFWS 2012). Most of the higher elevation springs in SNWR have aquifers that are localized and small. During early spring, snowmelt percolates through fractured basalt, often traveling long distances before resurfacing after coming in contact with an impermeable rock layer composed of volcanic tuff (USFWS 2012). Volcanic tuff forms when volcanic rock, ash and magma are thrown into the air by an eruption. The ejected material falls back to earth and is compacted/cemented into rock (Plume and Carlton 1988).

Hydrologically, the refuge is within the Great Basin, particularly the Alvord sub-drainage in the east and the Guano sub-drainage in the west (Herbst 1996, Omernik and Gallant 1987). To date, the U.S. Fish and Wildlife Service has identified and inventoried over 130 springs, most of which are in the western and southern portions of the refuge (USFWS 2012). A majority of the springs inventoried in SNWR have had some kind of water development constructed directly at the spring source or within the adjoining wet meadow. These come in a variety of sizes and styles from springboxes and headboxes to small dugouts and large reservoirs. Early pioneers to the area constructed many of these water developments to capture water for agricultural use (Sada et al. 2001, Wyman et al. 2006, USFWS 2012). Water developments continued after the refuge was established to benefit wildlife species (Hazeltine 1959, USFWS 2012). Most are now abandoned and left to corrode (USFWS 2012). The unintended consequences remain to the present day in the form of dewatered wet meadows, altered plant communities, formation of incised channels, invasive species, and loss of ecosystem functions (USFWS 2012, Wyman et al. 2006, Chambers and Miller 2004, Prichard et al. 1994, Sada 2008).

Site Selection

We selected study sites from a list of altered springs identified by the U.S. Fish and Wildlife Service in the Sheldon National Wildlife Refuge in northwestern Nevada (Figure 1.1). Sites ranged in elevation between 1737 m and 2042 meters above sea level. Sites were selected based on the type of water development and accessibility to roads. We excluded sites in the northeastern salt-desert systems due to lower elevation, plant community and underlying hydrogeology. Water developments on the refuge included dugouts, berms, reservoirs and springboxes. For this study, we concentrated on sites developed with a single springbox (Figure 1.2) as these were the most numerous type of development in the study area. We selected twenty-four springbox sites from the list of available springboxes. We narrowed our choice of sites to springboxes that consisted of a corrugated metal pipe roughly a meter in diameter and two meters deep with one outflow pipe that went off to a trough (which may or may not have been removed) and an overflow pipe that released excess water into the riparian corridor at various distances from the springbox. Some initially chosen sites were rejected because the springbox had been removed from the site already and the records not updated or the design/look/materials/function of the springbox was different from other springboxes in the study.

Study Design

Our twenty-four spring sites were divided into four groups of six sites (Figure 1.3). Within each group, spring sites were randomly assigned one of six treatments: control, capped pipes, sand-filled, sand-filled with springbox casing removed, gravel-filled, and gravel-filled with springbox casing removed. Groups were organized based on the amount of flow measured during pre-treatment data collection consisting of high flow (0.030-0.005 ft^3/sec), medium-high flow (0.004-0.0017 ft^3/sec), medium-low flow (0.0016-trace ft^3/sec), and low flow (trace-0 ft^3/sec).

Treatment Type Descriptions

We designed six different treatments (Figure 1.4) with varying cost and effort needed for implementation.

1) Control: Springbox and all outflowing pipes were left unaltered to account for hydrologic variation (by flow group) between pre- and post-treatment periods.

2) Capped Pipes: Springbox outflow and overflow pipes were capped and underground pipes were removed. Soils were compacted to remove air pockets created during pipe removal, eliminating a water flow path. A metal lid was secured to the springbox, perforated with small escape holes to allow water to flow out while preventing rodents from getting in. In cases where the metal casing of the springbox extended above the ground, holes were created around the perimeter of the springbox at ground level to prevent water from climbing above ground level. This design was the simplest and most cost-effective treatment. However, the hydrologic pressure, which is greater within a solid column of water than when mixed with a substrate, could cause the spring to collapse under the weight of the water (Hopkins Personal communication).

3, 5) Sand or gravel-filled: Springbox was filled with coarse quarry sand or gravel of varying particle sizes. Before materials were added the internal outflow and overflow pipes were capped to prevent water from escaping through residual pathways formed by the pipes. The springbox was then filled with sand or gravel and any excess springbox casing present above ground level was cutoff level with the surrounding terrain. All excess underground piping was removed. Soils were compacted to collapse soil voids created during pipe removal. Our reasoning for this treatment design was that the natural scaffolding allowing the water to climb vertically to the surface had been disrupted by the installation of the springbox. Therefore, to prevent water from moving laterally into the surrounding soil, the metal casing of the springbox was left to provide a structure to force the water to move vertically and onto the surface. Over time, the metal casing will corrode (in some cases it already has) and water will have access to horizontal flow. The material fill, sand or gravel, would form an artificial ladder mimicking the geologic scaffolding creating a path for water to climb (Sada 2008).

4, 6) Sand or gravel-filled with springbox casing removed (hereafter sand SBR or gravel SBR): Each springbox casing was removed and the remaining hole was filled with either sand or gravel that had been obtained from an on-site quarry. An excavator was used to remove the casing and pipes (surface and underground). The filled hole was leveled with the surrounding terrain. Finally, soils were compacted to fill in voids left by pipe removal. These treatments removed all man-made structures. This allowed complete freedom for the water to move in any direction and to possibly create new flow patterns. However, this may pull water away from the surface.

Hydrologic Measurements

Surface Soil Moisture

We measured surface soil moisture to measure the ability of each sites' riparian zones to capture discharge from the spring source. Riparian soils act as sponges, making water available for vegetation and wildlife and forming the largest freshwater reservoir on Earth (Lewis et al. 2003). Soils that become fully saturated at the surface allow water to percolate back through the soil to recharge groundwater (Lewis et al. 2003).

Soil moisture measurements were collected three times during the field season (May, June and August) during both pre- and post-treatment periods. We measured percent soil moisture of the soil surface using a DYNAMAX© ML3 Theta Probe. Measurements were taken over a depth from the soil surface down to 5 cm. If the probe could not be easily inserted into the ground due to rocks or roots, it was pulled out and an alternate sample location was selected adjacent to the initial placement, repeated until a measurement could be taken. At some locations (13%), the measurement could not be taken due to a high concentration of rocks in the soil. The first transect for this dataset was chosen by standing on top of the springbox and randomly selecting a degree between 0 and 360. An upwards of 20 m transect was placed along the random position and three other transects were placed a 90 degree intervals. Transects ran to the edge of the riparian zone and then five additional meters into upland vegetation. Up to five points were randomly selected within the riparian zone and then systematically measured five meters into the upland. If the riparian zone extended beyond twenty meters then the upland measurements were discarded. We limited the maximum extent of our study to twenty meters in order to concentrate specifically on the influence of the springbox and spring source. Beyond that distance, data may become influenced by other aquifers and groundwater inputs. High elevation springs in SNWR are small (>3 acres), and all of our sites fall within this high elevation range (1737-2042 m) (USFWS 2012). These same transects and points were repeated for post-treatment measurements.

Deep-Pit Soil Moisture Measurements

At the center of each springbox, a random bearing was chosen, with two others at 120° intervals. At approximately one meter from the springbox along each transect, a 45 cm deep pit was dug. Pits were dug to this depth because wetland obligate species grow where the water table is at a depth of 0-50 cm (Castelli 2000). Soil moisture measurements were sampled at 0 cm, 15 cm, 30 cm, and 45 cm depths using a percent soil moisture detection probe (Theta probe®). These measurements were taken to characterize soil moisture at depths accessible by plant roots through capillary action (Lewis et al. 2003). When water filled the pit, we determined that we had reached the water table. Deep-pit soil moisture measurements were sampled 2-3 times at each site during the pre- and post-treatment field seasons with new random locations selected for each sampling session to avoid bias from previously dug pits.

Flow

We measured flow by capturing channeled surface water in a graduated cylinder and timing the volume. We selected the cylinder size based on approximate discharge rates at each spring (10 ml, 250 ml, 750 ml, 2.5 L or 4 L). Measurements were repeated twelve times and the longest and shortest times were discarded to account for variations in flow rate and human error. Sites where water was present but did not appear to be flowing were marked as trace. At some sites, either the trough pipe or the outflow pipe were easily accessible. By blocking the overflow pipe inside the springbox, all discharge could be channeled through the outflow pipe and flow measurements were taken at that location when available. This measurement was collected at each site 2-3 times (May, June and August) during the pre-treatment period, and repeated the same number of times during post-treatment data collection.

Water Chemistry

Water samples were collected from each spring site in early summer during both pre-treatment and post-treatment. Whenever possible, water samples were collected as close to the spring source as possible. In some cases, the samples were collected where the water exited an outflow pipe. The outflow pipe siphoned water from below the surface in the springbox and therefore closer to the spring source. The pipe was buried underground, so the water in the pipe was protected from atmospheric contamination until it emerged. Samples were collected and stored in sealed nalogen bottles to prevent evaporation. Water analyses were conducted by the Brigham Young University water chemistry laboratory. Using cavity ring-down spectrometry, the ratio of oxygen-18($d^{18}O$) to oxygen-16 ($d^{16}O$) for each sample was calculated and then compared to Vienna Standard Mean Ocean Water (VSMOW), which has an isotopic ratio of one for $d^{18}O/d^{16}O$. VSMOW is the standard machinery used to determine $d^{18}O$ ratios. The difference in $d^{18}O$ ratios of our samples and VSMOW were plotted compared to the Global Meteoric Water Line (GMWL). The slope of the GMWL predicts how isotopes naturally separate within the atmosphere, and can be used as an indicator of a water source's origin.

Water Location

We used 0.25 m resolution satellite imagery to digitize the wet zone created by each springbox prior to treatment. The boundaries pre-treatment and post- were plotted on the same map to create a visual representation of the shift in wetted area for each spring (Figure 1.10).

Statistical Analysis

All datasets were organized and analyzed using the same methodology. Each site and its measurements were labeled with one of each category (i.e. flow group, treatment type, and spring type). For example, Bateman spring was labeled: highest flow group, gravel-filled treatment and single spring type. We measured variables along four transects at each site. Recorded variables along each transect were averaged across the entire site, producing a single value for each site per year. If multiple measurements for a variable were taken during the field season, we averaged all measurements together to produce a single value for each site per year. Pre-treatment variables were compared to post-treatment values using an analysis of covariance. Pre-treatment values were used as a covariate. Due to the individual and diverse nature of our study sites, we interpreted p-values of 0.1 significant changes. Our dependent variable was the change between years (pre-treatment vs. post-treatment). We used Analysis of Variance (ANOVA) to investigate whether or not there was a difference between pre-treatment and post-treatment based on categories of spring type, flow group or treatment type. We further analyzed the individual elements of each category using a test of least square means; again asking was post-treatment different from pre-treatment (our dependent variable). Finally, we wanted to test whether our treatments were different from our control sites. We ran a difference of least square means to test for differences in the amount of change between the two years when comparing treatments to the control (or other treatments). The results were adjusted for multiple comparisons using a Tukey-Kramer adjustment. We used SAS© for the statistical analysis of our data.

For individual springs, we used an overall standard f-test to analyze the difference between pre-treatment and post-treatment. We also ran a least square means to show whether our estimates for each year significantly differed from zero. P-values of 0.1 or less were considered significant for this test. Data and results were discarded if there was insufficient information to properly run a statistical analysis.

RESULTS

Surface Soil Moisture

Between pre-treatment and post-treatment, soil moisture increased for combined upland and riparian sites (Figure 1.5). We detected an increase in percent soil moisture for both spring types (complexes $p=0.0163$, single $p=0.0195$) and all flow groups (high $p=0.0133$, low $p=0.0036$, medium-high $p=0.0310$, medium-low $p=0.0586$). Soil moisture was higher in the control group and the gravel-filled and sand SBR treatment sites (Table 1.1). Although gravel-filled and sand SBR treatments showed increased soil moisture, the amount of change between pre- and post-treatments did not vary from the change in the control group (Table 1.1). When the measurements were split out into riparian and upland only analyses, these treatment effect changes are no longer significant. Increases and decreases in the soil moisture in riparian and upland zones can be seen across the board when considered on a site-by-site basis (Figure 1.5). Eleven of our sites had increased soil moisture in the riparian zone (Table 1.2), and seventeen sites displayed changes in soil moisture in upland zones between pre-treatment and post-treatment.

Three of the treatments lost soil moisture but the others were increased. In accordance with the other results for overall soil moisture, nearly all the springs in this study exhibited increased soil moisture within a 20-meter radius of the spring source.

Deep Soil Moisture

Soil moisture increased for every spring type, flow group and treatment type at all four depths (Table 1.3). Differences were also detected on a site-by-site basis (Figure 1.6). Fourteen springs showed increased soil moisture at 0 cm (Figure 1.6). This was expected as we saw similar results in our surface soil measurements. At 15 cm, sixteen sites had increased moisture content with seven of those sites reaching the water table. Increased soil moisture occurred at a depth of 30 cm for fourteen springs. Only ten springs had increased soil moisture at a depth of 45 cm. Yearly effect analysis determined that the change in the treatment did not differ from the change in the control (Table 1.3).

Flow

Our results showed that there were no change in the flow rates discharged at our sites, except for the control group (p=0.0057). Round Mountain and Tomato springs measured an increase in flow rates after treatment. Round Mountain had trace amounts of flow in 2016 and increased by 0.000878 ft^3/sec (p=0.0337). Tomato also increased from trace amounts of flow to 0.000835 ft^3/sec (p=0.0021). Other sites (Beebee p=0.0646, Mule p=0.0561 and Rock Spring p=0.0078), however, decreased in their flow rates in post-treatment. When considering yearly effects, none of the treatments measured changes in flow that varied from the controls (Table 1.4).

Water Chemistry

Our samples were all isotopically depleted in comparison to VSMOW, with $d^{18}O$ ratios ranging between -10.34 to -17.01 (Figure 1.7). The only outlier measurements came from

Meadowlark spring in pre-treatment with a $d^{18}O$ of -10.34. This could be due to possible evaporation in the sample, but this is unlikely because the value is still very close to the GMWL.

Because most of our springs had an average $d^{18}O$ value, they are fed by winter precipitation and that warm-weather monsoonal precipitation has little effect on spring flow overall (Nelson 2018).

Water Location

After treatment, there was a very clear shift in the wetted area that was established underneath a trough (or pipe if the trough had been removed) to an increased or new wetted area around the spring source. Pre-treatment wetted areas were formed by water being funneled to a trough by an outflow pipe and excess water being released into the original wet meadow by an overflow pipe (Figure 1.9). Except for the controls, every treatment capped both pipes, cutting off water flowing to those pre-treatment wetted areas. This shift could be seen using satellite imagery and mapping where the wetted area was seen in pre-treatment and where it had shifted to in post-treatment. The wetted area of some sites shifted, while some did not (Figure 1.10).

DISCUSSION

We hypothesized that a spring on the path to hydrologic restoration would reach three milestones. First, we would see an increase in soil moisture on the surface. Second, flow rates would increase due to more water discharged into the wet meadow around the source. Third, the water table would rise (Cowley 1997, Wyman et al. 2006, Lewis et al. 2003).

The largest hydrologic change observed was the increased surface soil moisture. Eight of our spring sites changed from having bare, dry soil to pooled surface water at or near the spring source. Seven of these sites also experienced the development of a riparian channel spreading downstream compared to no channel flow in post-treatment. Soil moisture increased over all spring types, flow groups and most of our treatments (complexes $p=0.0163$, single $p=0.0195$, high $p=0.0133$, medium-high $p=0.0310$, medium-low $p=0.0586$). The treatments that had the greatest influence on soil moisture were the control group, gravel-filled, and sand SBR (Figure 1.5). We believe that leaving the metal casing in the ground may have been involved in increasing surface soil moisture. The impermeable metal forced water to the surface, instead of allowing it to flow out horizontally where it would have been less effective. In a natural system, geologic strata at the site would have created a natural funnel to the surface where the spring would have emerged (Maurer et al. 2004). Installation of the springbox disrupted the natural formation; therefore, leaving the metal casing may act as a structural replacement that mimics the funneling effect of the underlying geologic structure (Sada 2008, Patten et al. 2008).

No single treatment increased surface soil moisture in riparian zones or uplands alone. However, when examining our sites individually, we detected differences in soil moisture content (Figure 1.5). For example, Little Fish increased in soil moisture in the riparian zone alone from an average of 4% to approximately 80% soil moisture ($p<0.0001$). Of the 19 sites that had riparian zones in pre-treatment, 11 saw increased soil moisture in post-treatment. Increased soil moisture in the riparian zone suggests that more water is available for wetland species to persist in the plant community. Soil moisture in the upland tells a more interesting story. Twenty-three or our twenty-four sites had a riparian zone with a radius smaller than 20 m. In these cases, we measured 5 meters into the upland zone to try to characterize changes to the extent of the riparian zone. The extent of riparian zones are dynamic (Lewis et al. 2003) and can change with any shift in the underlying hydrology of a site (Gray et al. 1992). Of those twenty-three sites, fifteen had increased post-treatment soil moisture content. Three upland zones decreased in soil moisture. An increase in soil moisture may indicate that the riparian zone is expanding as moisture moves outward through sheet flow across the surface (Wyman et al. 2006, Patten et al. 2008) and through capillary action (Lewis et al. 2003). We kept our measurements identical during each field season, so where our upland measurements began along the transect reflects where the uplands began before treatment. Conversely, decreased soil moisture in the uplands may indicate that riparian zones area are diminishing. None of these three decreasing sites had increased riparian zone soil moisture. This indicated that moisture is contracting or consolidating in the riparian zone of these sites. It is important to note that we also observed increases in soil moisture in half of our control groups, which may be a result of the wetter year in post-treatment. However, we suggest that at least some of that change is due to our treatments and not just a yearly climatic effect (Welsch et al. 1995, Wyman et al. 2006).

On a site-by-site basis, our most dramatic increases in soil moisture in both the upland and riparian zones occurred in sites where the area around the springbox or spring source was extremely dry pre-treatment (less than 10%). These sites increased in soil moisture anywhere from 25% to 75% post-treatment. Some of our increased surface soil moisture may be a result of higher precipitation in the post-treatment period than the year before. However, the presence of surface water shifted closer to the spring source at many of our sites (Figure 1.10). While climatic changes may have contributed to increased soil moisture, the physical shift in spring discharge location can be wholly attributed to our treatment implementation.

Our second measure of restoration success was the change in flow rates. Our data did not bear this out. Overall, only flow for the control group increased ($p=0.0057$). This is in part due to the presence of more water being discharged in general due to higher precipitation during the winter between pre-treatment (2016) and post-treatment (2017). Cumulative precipitation for the 2016 and 2017 water years were nearly identical (25.1 cm and 24.94 cm, respectively). However, looking at the NRCS Soil Climate Analysis Data (SCAN) for both years on May 1 we can see a clear difference in the cumulative precipitation. As our field season began in May of 2016, SNWR had 14.33 cm of precipitation, but by May of 2017 SNWR had already received 18.14 cm of precipitation (USDA 2017). This was a 26% increase from the pre-treatment period. Pre-treatment came on the tail end of several dry years and the amount of water being transported and discharged at the study springs may have been heavily altered by diminished regional aquifers (Lewis et al. 2003, Welsch et al. 1995). After being recharged over the winter, the flow rates of post-treatment may be more indicative of the natural water potential of these sites. If this scenario was accurate, then we question why flow rates did not increase for all sites. One option is that many of our sites that had diminished flow rates also experienced increased pooling in and around the spring source (Patten et al. 2008). It is possible that the gross flow rates themselves had not changed, but that the water was no longer flowing in a channelized pattern typical of stream systems (Hancock 2002). These sites may have morphed from a lotic system to a more lentic-type system where the water flows in a more spread out sheet flow pattern. The water in sheet flows cover a wider area and has a slower velocity, allowing it to infiltrate back into the ground (Naiman and Decamps 1997, Lewis et al. 2003).

Improved infiltration and water holding capacity are prime functions of a healthy riparian zone (Lewis et al. 2003, Wyman et al. 2006). These systems can function as a colossal organic sponge, soaking up discharged water and holding it for slow release into springs or riparian channels (Wyman et al. 2006, Lewis et al. 2003). With slower discharge, water is more available for plant growth and animal use over longer time period into the summer (Wyman et al. 2006).

Our method of measuring flow rates would not be effective in capturing this phenomenon, hence the apparent changes.

Three of our sites had decreased flow rates during post-treatment periods. Beebee, a spring in the low flow group that produced only 0.00053 ft^3/sec of flow during the pre-treatment period, was capped. In post-treatment, no flow was measureable and the catchment basin did not fill with water to a level sufficient to escape out holes in the springbox. We suggest that the volume of water discharge was insufficient to outpace evaporation (Lewis et al. 2003). Therefore, even though the amount of discharge may have remained the same, the treatment at this site trapped the water in the springbox (Barquin and Scarbrook 2008, Sada 2008). Some water did soak into the surrounding soil profile as determined by an increase in soil moisture at zero ($p=0.0191$) and 15 cm ($p=0.0606$) depths (Lewis et al. 2003).

Mule, a spring with low flow in pre-treatment had pipes plugged and the springbox filled with gravel. In post-treatment, there was no longer any flow being produced from this springbox. Mule spring was part of a spring complex with multiple springs bubbling up to the surface within close proximity to one another. A likely explanation is that Mule's diminished surface flow resulted from the re-routing of flow that joined subterranean discharge from other underground spring systems nearby (Plume and Carlton 1988, Lewis et al. 2003, Maurer et al. 2004, Patten et al. 2008). Water follows the path of least resistance through the soil profile following capillary action as it is pulled toward plant roots and other soil water (Lewis et al. 2003, Maurer et al. 2004). There is no photographic evidence or other records of what Mule looked like before development, but if it was a single spring source originally, the process of installing a springbox could have caused a partial collapse of the original spring (Erman 2002) leading to the formation of a post-development spring complex.

Finally, two springs, Tomato and Round Mountain (Figure 1.8), were dry in pre-treatment, however, post-treatment these sites both developed lentic pooling around the spring source with measureable flow downhill (Patten et al. 2008). Both were single spring systems where the entirety of the spring source was encased in the water development. Blocking outflow pipes concentrated water back at the spring source potentially providing sufficient hydrostatic pressure to drive water to the surface at the spring site (Sada 2008, Lewis et al. 2003). At Tomato, we could actually observe the spring bubbling up through the sand.

The third and final restoration conditions that we would expect to see is a decrease in the depth to the water table (Lewis et al. 2003, Wyman et al. 2006, Barquin and Scarsbrook 2008, Sada 2008). We attempted to capture this by digging pits down to 45 cm and measuring the percentage of soil moisture at 0, 15, 30 and 45 cm as we descended the soil profile (Figure 1.6). When water flowed into our pit, we had reached the water table. In a study by Castelli et al. (2000), wetland-obligate communities were shown to grow when the water table was at a depth of 0-50 cm. Plant communities in degraded wet meadows with dropping water tables shift from wetland-obligates species to species that prefer drier conditions, such as grasses and shrubs (Perkins et al. 1984, Chambers and Miller 2004). Restoration that decreases the depth to the water table to 50 cm or less provides the conditions needed for recolonization of wetland species (Wyman et al. 2006, Barquin and Scarsbrook 2008). Soil moisture increased at every depth and in every category indicating a rehydration of riparian soils throughout the soil profile (Table 1.3). Rehydration of riparian soils results in more water stored in wet meadow ecosystems (Lewis et al. 2003).

Similar to surface soil moisture, differences also appeared on a site-by-site basis. Fourteen springs showed increased soil moisture at the 0 cm mark. This is expected as we saw similar results in our surface soil measurements. In post-treatment, five sites that did not have standing water around the spring source pre-treatment had reached the water table at 0 cm with an average of 66% increase in soil moisture. We also saw an average increase of 51% over those fourteen sites. With the exception of Little Catnip, all of our sites had water tables at an average depth of greater than 45 cm pre-treatment. In post-treatment, five springs had a depth to water table of 0 cm, seven had 15 cm or less, ten had 30 cm or less. At sixteen of our sites, the water table had decreased in depth to 30-45 cm with nine of those sites having significantly increased soil moisture content. An increase in surface soil moisture and a decrease in depth to the water table at our sites show that restoration has an effect on the hydrology of riparian systems.

Changes in soil moisture and flow reflect the treatment effects at our sites. However, we also accounted for the influence of unpredictable variation in precipitation from year to year. This was accomplished by comparing the amount of change in precipitation between our treatments to the amount of change observed in our control groups. For almost every variable, our treatments did not vary from the controls. As mentioned previously, the post-treatment period was approximately 26% wetter than the pre-treatment period. This suggests that the lack of difference observed at our sites were driven by increased moisture and not restorative measures. Collectively, our treatments appeared to cause little to no change. However, we do not believe that this negates the changes seen in the treatment effects. It does emphasize the need for multiple years of data collection, preferably with precipitation levels that are similar to pre-disturbance climatic conditions and changes in the underlying hydrology resulting from restoration. Changes at individual sites were too drastic in some cases to attribute entirely to climate. Precipitation increased by 26%, but many of our driest sites experienced increases of surface soil moisture between 26-75% increases. Changes in soil moisture and flow resulting from treatment effect may have been masked by pooling sites in the analysis with drier sites cancelling each other out.

In spite of increased precipitation, one aspect of our study that can be wholly attributed to treatment effect is the shifting in location where water emerges onto the landscape (Figure 1.10). By capping the pipes in all of our treatments, water was no longer diverted away from the spring source. Many sites had water reoccurring on the surface around the spring source where it had not been flowing pre-treatment. At some sites, water disappeared altogether from the surface of the spring and wet meadow. A potential explanation is that the flow became subsurface, moving horizontally through the soil profile instead of being funneled up to the surface. Additional years of data collection of flow rates, water table depths and soil moisture values will be needed to determine the long-term effects of restored springs that have subsurface flows.

Management Implications

All treatment techniques were similar in their effectiveness or ineffectiveness. Any restoration approach that prevents water from leaving the riparian zone (filling in ditches, capping pipes) will have the effect of raising surface soil moisture levels (Sada 2008, Barquin and Scarbrook 2008). Simply capping pipes and allowing the catchment container to fill with water is inexpensive and increases water availability. However, as seen with very low flowing springs, simply capping pipes could potentially trap water in the catchment basin, especially when evaporation rates outpace discharge. This prevents water from connecting with its floodplain (Wyman et al. 2006). Riparian zones will not become saturated, allowing for groundwater recharge and will lead to a shift in plant communities from wetland to drier species (Prichard et al. 1994, Wyman et al. 2006, Sada 2008, Fleischner 1994, Chambers and Miller

2004, Perkins et al. 1984). Additionally, capping pipes alone may create a hazard to small-bodied wildlife (i.e. rats, voles and mice) by creating a deep pool that they can drown in (Andrew et al. 2001). Several times during our study, both before and to a lesser extent after treatment, we encountered springboxes that held decaying carcasses of rodents and birds that climbed into the catchment through a hole or dry pipe and drowned. This also poses a health risk for human use. Even though capping pipes is an effective restoration technique, if funding allows, we recommend leaving the metal casing in the ground and filling it with a substrate, such as sand or gravel, whichever is available. This provides a scaffolding for water to climb and reach the surface, and provides structure to protect against possible collapse of the spring source (Sada 2008). Filling the springbox reduces the potential of drowning small animals and contaminating the water source. Although not addressed in this study, having a good precipitation year following treatment efforts, or long-term precipitation patterns, may have a large influence on the success of hydrologic restoration (Sada 2008, Lewis et al. 2003, Wyman et al. 2006, Welsch et al. 1995). A well-recharged regional aquifer at the beginning of restoration may "prime the pump" and allow for more successful continual groundwater recharge and recruitment of healthy riparian vegetation in the long run (Wyman et al. 2006). Further years of data collection may help to determine the long-term efficacy of restoration.

In the process of restoring a spring and wet meadow system, a riparian zone in another area created by diverted water may be imperiled (Figure 1.10). Many of our spring sites had pipes leading from a few to hundreds of meters away to a trough. With pipe capping and trough removal, the wetted area that supports riparian vegetation surrounding the trough will experience drying conditions and a likely shift in plant community structure (Prichard et al. 1994, Chambers and Miller 2004). We will discuss the effects of restoration on plant communities in Chapter 2.

Understanding the underlying geohydrology of a site is imperative to understanding and predicting how spring systems will react to shifts and variations in climatic situations. Surveys from USGS reveal that springs in the western half of SNWR are formed from consolidated rock of volcanic origin (Plume and Carlton 1988, USFWS 2012). This lithologic type often forms single-valley closed systems that react similarly to our spring sites (Plume and Carlton 1988). Our water chemistry data provides further evidence that discharge for these springs comes from winter precipitation events as all $d^{18}0$ values are very close to the slope predicted by GMWL (Figure 1.7). The GMWL slope predicts how atmospheric isotopes separate in nature. Oxygen-18 is a stable indicator of spring precipitation origins because it does not exchange oxygen with the rocks surrounding underground aquifers (Nelson 2018). For hydrologically closed single-valley systems, oxygen-18 is a valuable indicator. However, some springs are supplied by deep aquifers or from mixed sources. Oxygen-18 would not be useful in determining these spring's origins. To truly assess the sources of a spring, we recommend further investigation using tritium and carbon-14 (14C) to determine the recharge interval of these springs. Tritium is present in water that is less than 75 years old due to atomic activity. 14C would be able to determine if the water is ancient water and comes from deep aquifers (Nelson 2018). When springs are supplied by a mixture of deep aquifer water and precipitation, we would expect them to have a more consistent flow year round that are not as effected by seasonal shifts in precipitation (Plume and Carlton 1988). These would be more drought resistant. Pure precipitation dependent springs are more responsive to climatic changes (Sada 2008, Chambers and Miller 2004). With less winter precipitation, those springs may dry up earlier and earlier in the year. Springs that are fed by ancient water or have a larger catchment basin should be the focus of restoration is resources are limited, as they will provide a more consistent source of water (Plume and Carlton 1988).

Conclusion

Springbox reconstruction and site relocation can lead to improved surface soil and local aquifer hydrology (Sada 2008, Wyman et al. 2006). However, our study indicates that there are no one-size-fits all restoration methods for lentic springs and seeps in the Great Basin. As long as springbox reconstruction methods put water back on the surface around the spring source, percent soil moisture should increase (Sada 2008). Increased flowrates are not a true indicator of spring health, as they do not account for shifts in channel flow to sheet flow. Diminished flowrates are not an indication of lost production, but in combination with higher soil moisture, show that the riparian zone is acting as a bio-sponge. This sponge captures water from the spring and holds it so that it can be released slowly throughout the year (Wyman et al. 2006, Lewis et al. 2003, Patten et al. 2008).

Treatment type may have a greater effect on water developments that completely encapsulate the spring source than those that have just been sunken into the water table nearby. However, without records of what these sites looked like before development, we can only speculate about their original condition. Many sites that are a part of a spring complex now, particularly those with another spring emerging from the ground meters from the springbox, may have begun as a single spring system. The very act of installing the springbox may have caused the original outlet to partially collapse, the new spring nearby resulted from the aquifer creating a new outlet for discharge. This could explain why many of the springboxes with this configuration have a rather robust satellite spring next to a subpar producing water development. It seems illogical for these underperforming developments to have been installed where they were with such a good source so nearby. This leads us to believe that the water developments were the original spring source at the site and that the complex of springs developed later.

Although we detected changes in our treatment effects, yearly effects appear to indicate that higher precipitation was the driving force behind these changes. With only one year of comparison data, it may be premature to definitively confirm the cause of the noted changes. More data collection is needed during years with precipitation similar to both pre-treatment and post-treatment periods. What is evident is that adequate precipitation is beneficial for improving the hydrologic function when recovering springs and seeps, and may be an integral ingredient in restoration efforts of these valuable Great Basin ecosystems.

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FIGURES

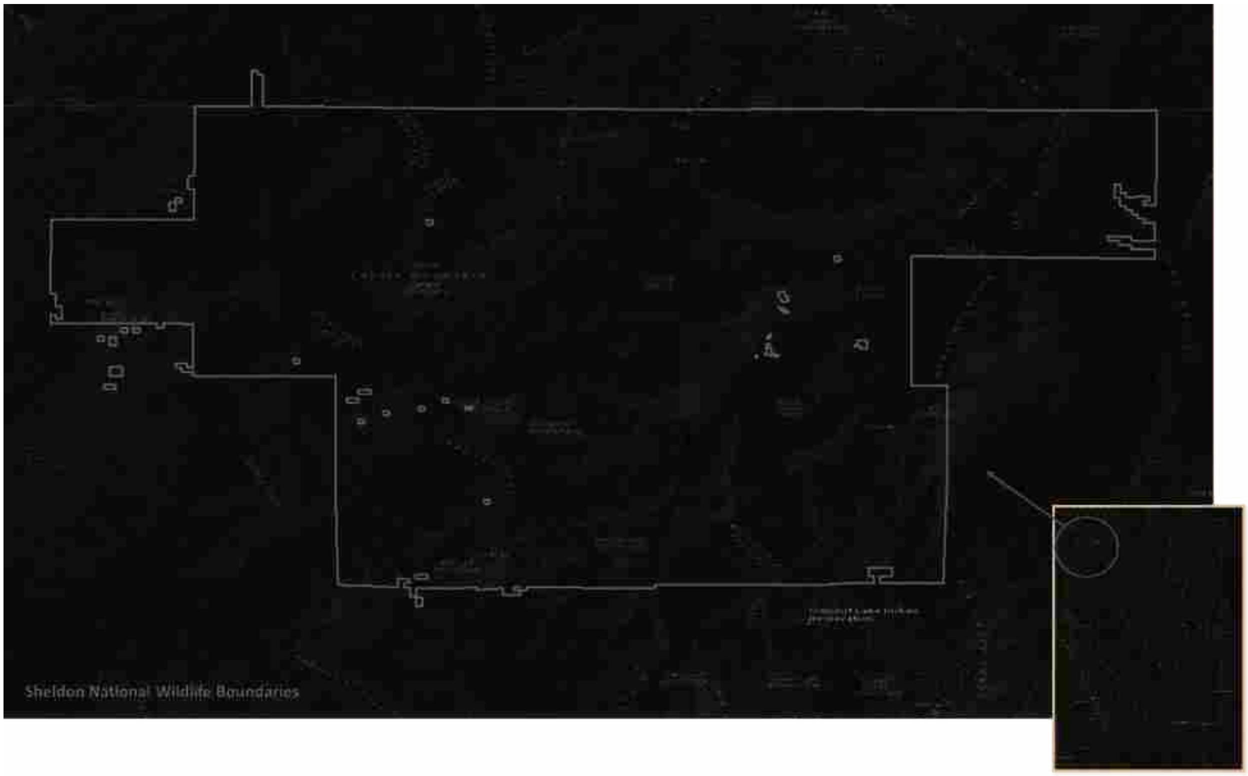


Figure 1.1. Sheldon National Wildlife Refuge is located in the northwestern corner of Nevada, USA. It was established in the 1930s as a refuge for the then endangered pronghorn (*Antilocarpa americana*). Today, the Sheldon National Wildlife Refuge contains 573,504 acres of mostly uninterrupted sagebrush-steppe ecosystem. Smaller polygons represent private inholdings within SNWR boundaries.

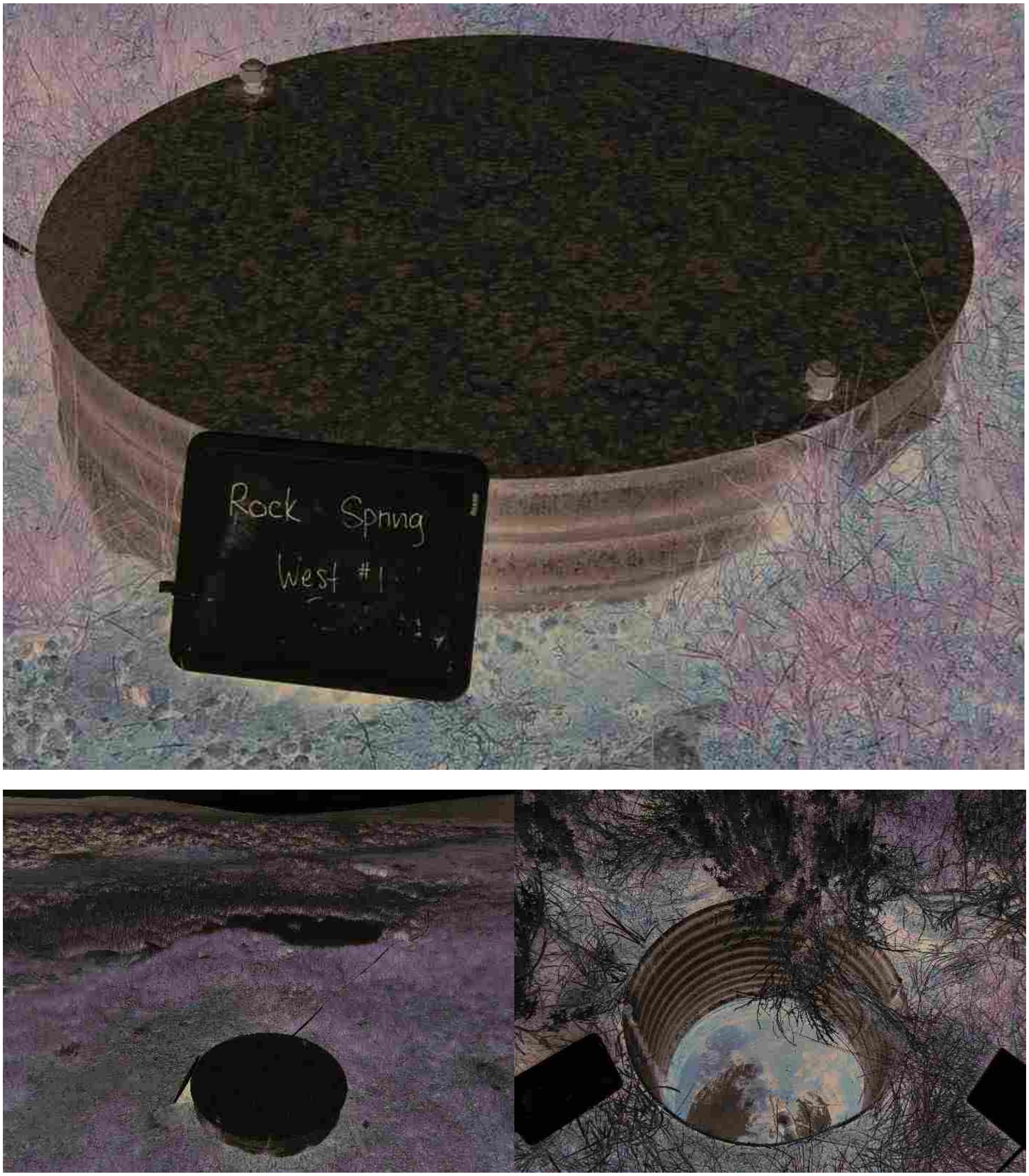


Figure 1.2. Springboxes, and other water developments, were installed on springs in the Great Basin by early settlers and land managers as water sources for livestock and wildlife. The unintended consequences of these structures was the dewatering of springs and wet meadows. Springboxes used for this study were installed around the 1960's and later by refuge managers to provide water sources for wildlife and livestock.

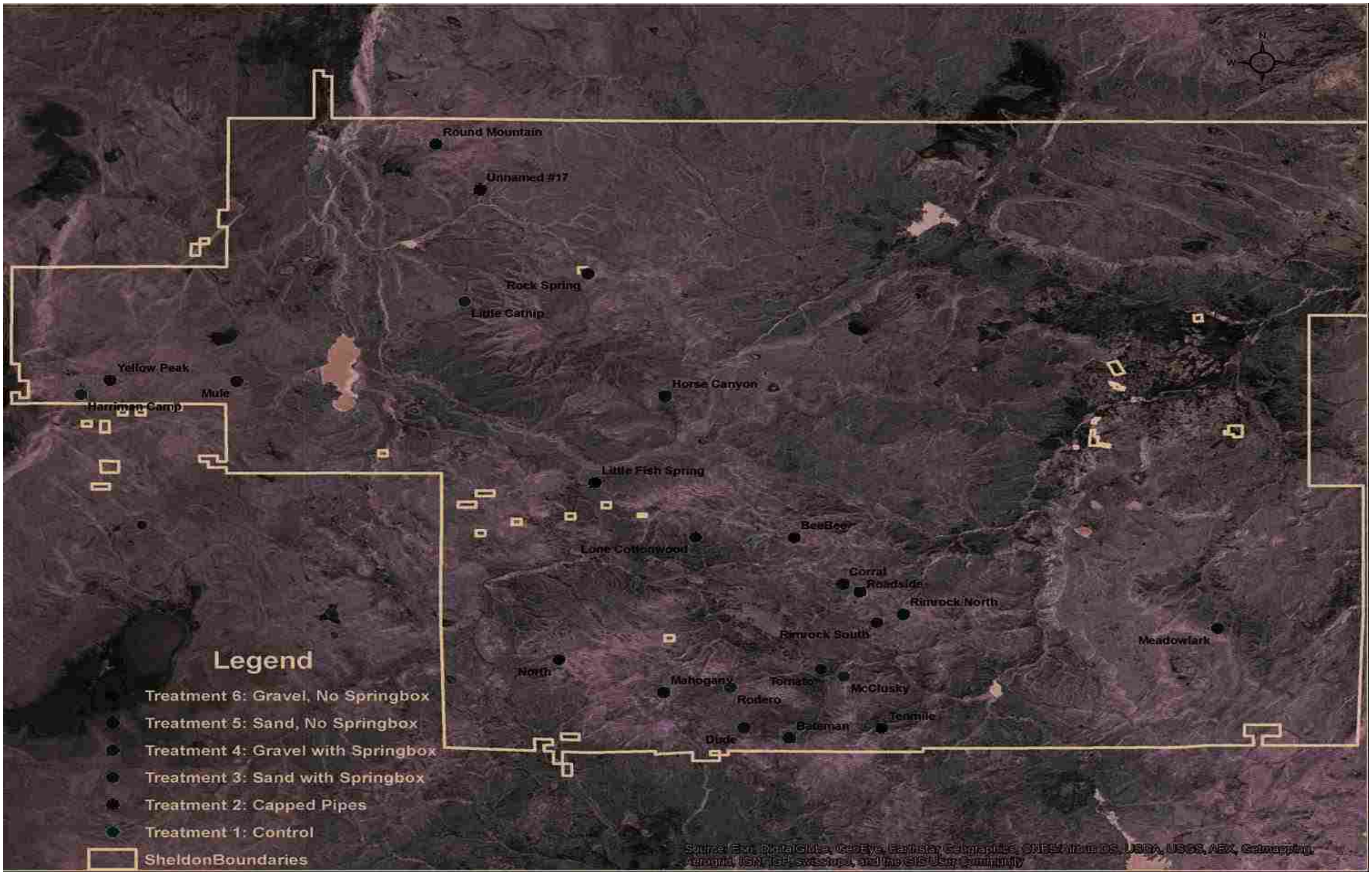
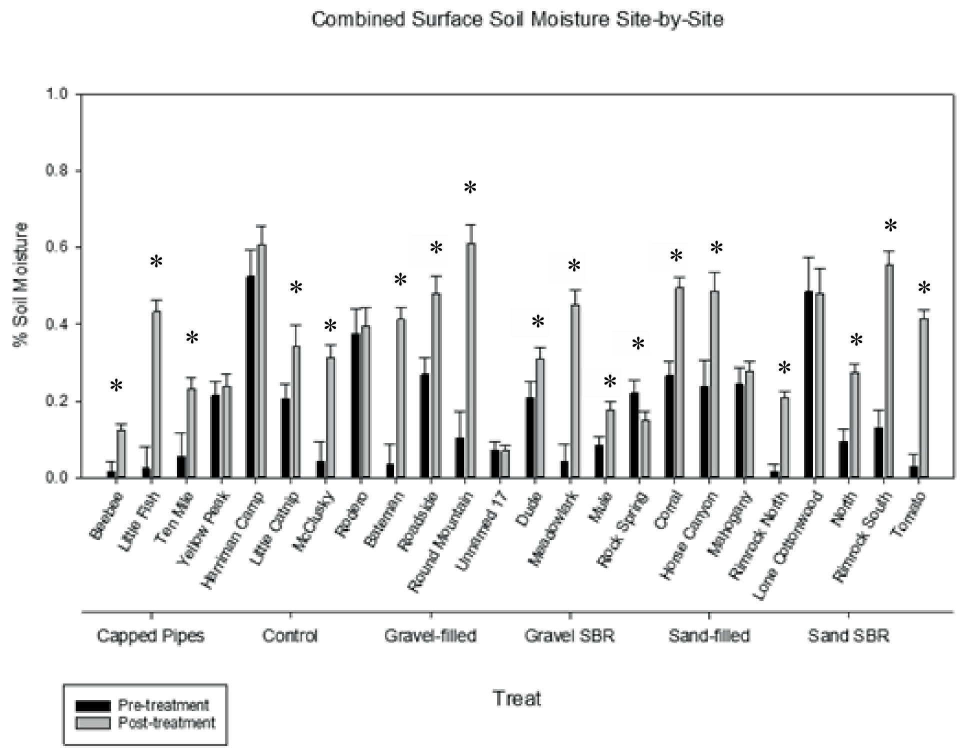


Figure 1.3. Randomly assigned spring sites in the Sheldon National Wildlife Refuge. Assigned treatments and locations are illustrated by different colored marks. Spring sites were selected based on similar elevations, accessibility, plant community and water development.

Figure 1.4. Spring-restoration treatment designs were created with cost, practicality and functionality in mind. Materials for treatments #3-6 were sourced from local sources. These treatments were designed to mimic the historic structure's functionality.

A



B

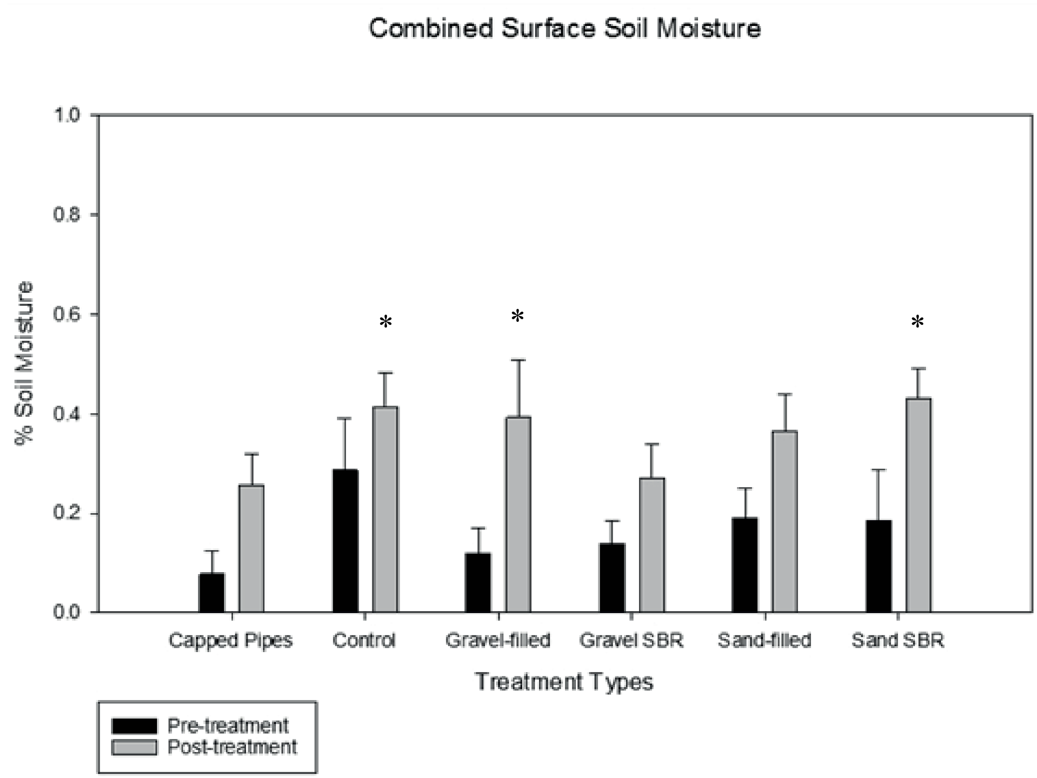
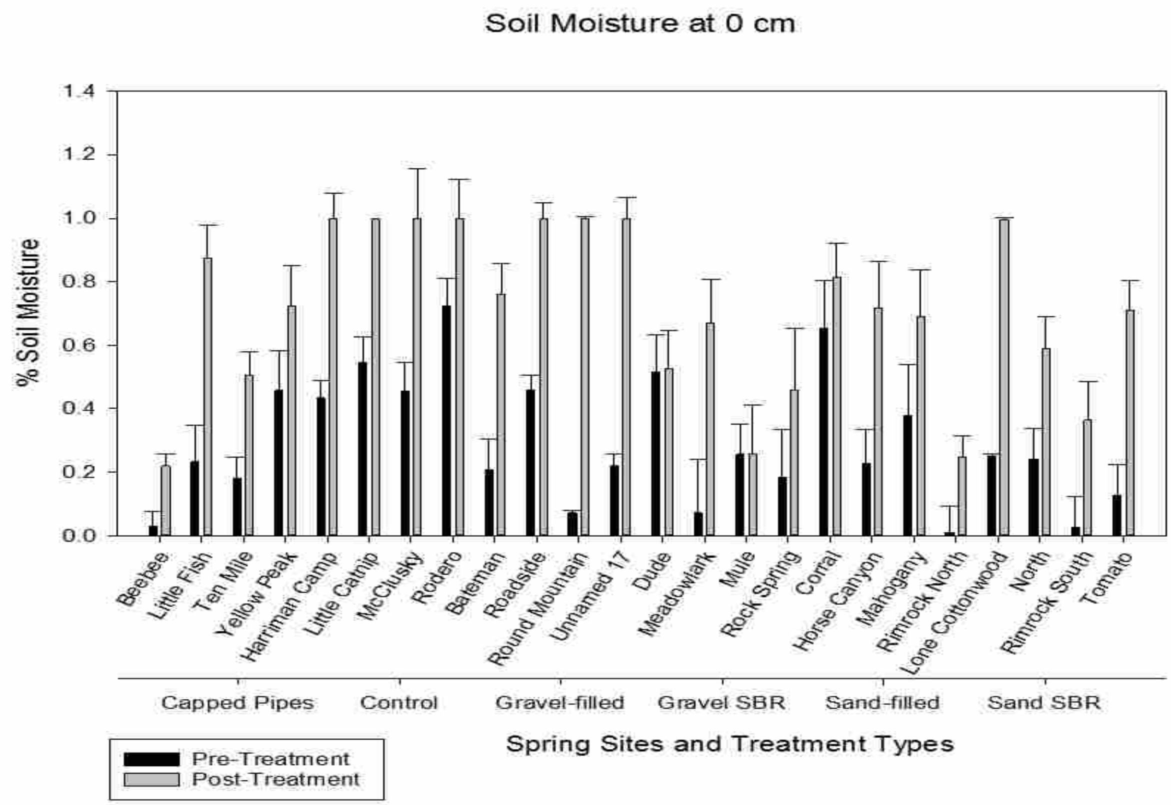
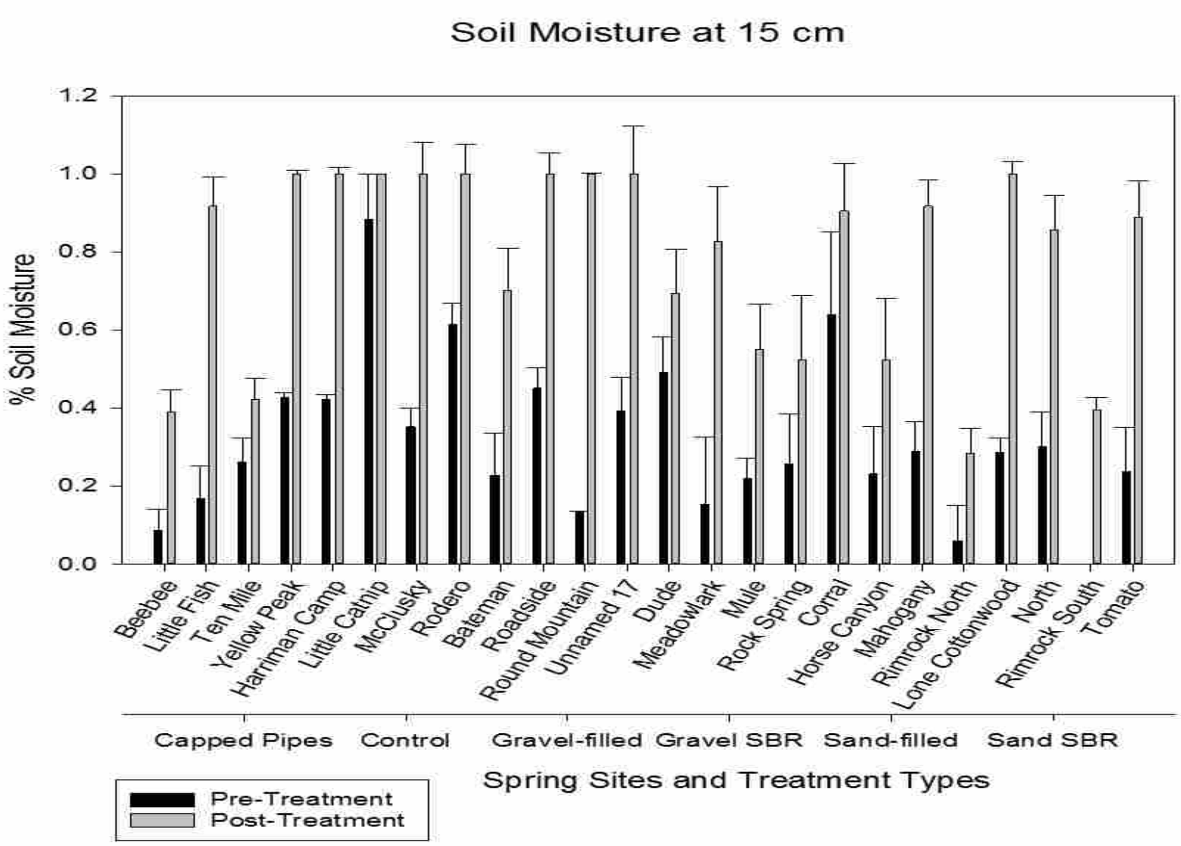
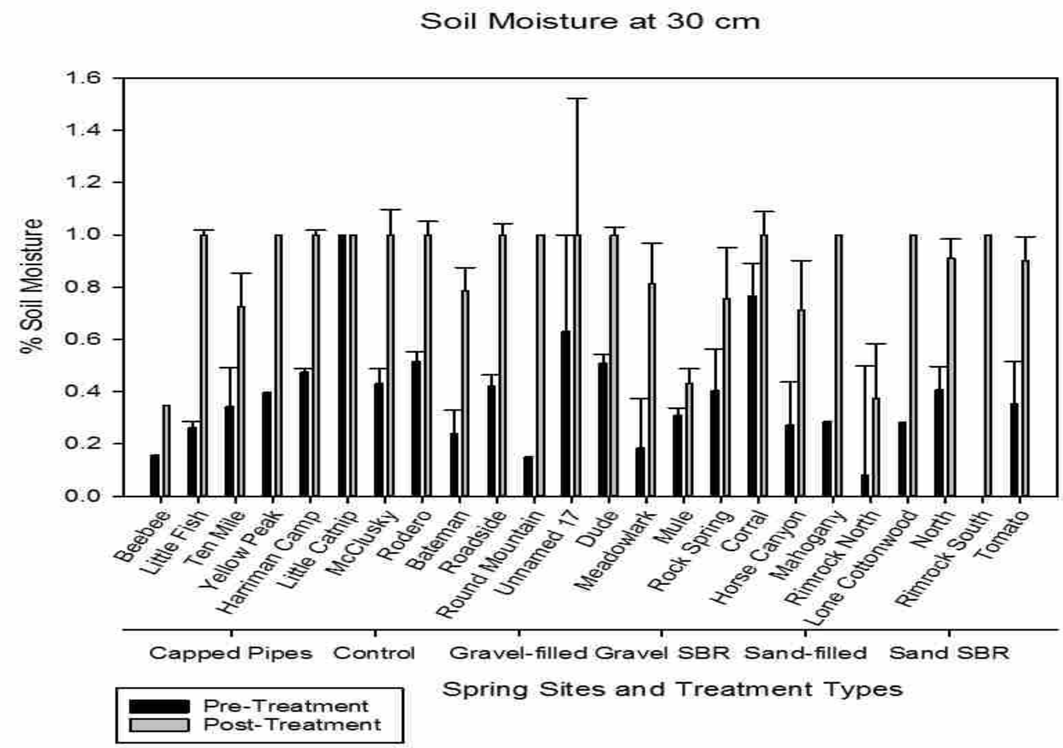


Figure 1.5. Combined (upland and riparian) surface soil moisture measurements for treatment types (A) and for individual sites (B). Sites saw an across the board increase in soil moisture, especially on an individual spring level. Significance denoted by (*).


Soil Moisture at 0 cm
% Soil Moisture
0.0
0.2
0.4
0.6
0.8
1.0
1.2
1.4
Beebee
Little Fish
Ten Mile
Yellow Peak
Harriman Camp
Little Catnip
McClusky
Rodero
Bateman
Roadside
Round Mountain
Unnamed 17
Dude
Meadowlark
Mule
Rock Spring
Corral
Horse Canyon
Mahogany
Rimrock North
Lone Cottonwood
North
Rimrock South
Tomato
Capped Pipes
Control
Gravel-filled
Gravel SBR
Sand-filled
Sand SBR
Spring Sites and Treatment Types
Pre-Treatment
Post-Treatment


Soil Moisture at 15 cm
% Soil Moisture
0.0
0.2
0.4
0.6
0.8
1.0
1.2
Beebee
Little Fish
Ten Mile
Yellow Peak
Harriman Camp
Little Catnip
McClusky
Rodero
Bateman
Roadside
Round Mountain
Unnamed 17
Dude
Meadowlark
Mule
Rock Spring
Corral
Horse Canyon
Mahogany
Rimrock North
Lone Cottonwood
North
Rimrock South
Tomato
Capped Pipes
Control
Gravel-filled
Gravel SBR
Sand-filled
Sand SBR
Spring Sites and Treatment Types
Pre-Treatment
Post-Treatment



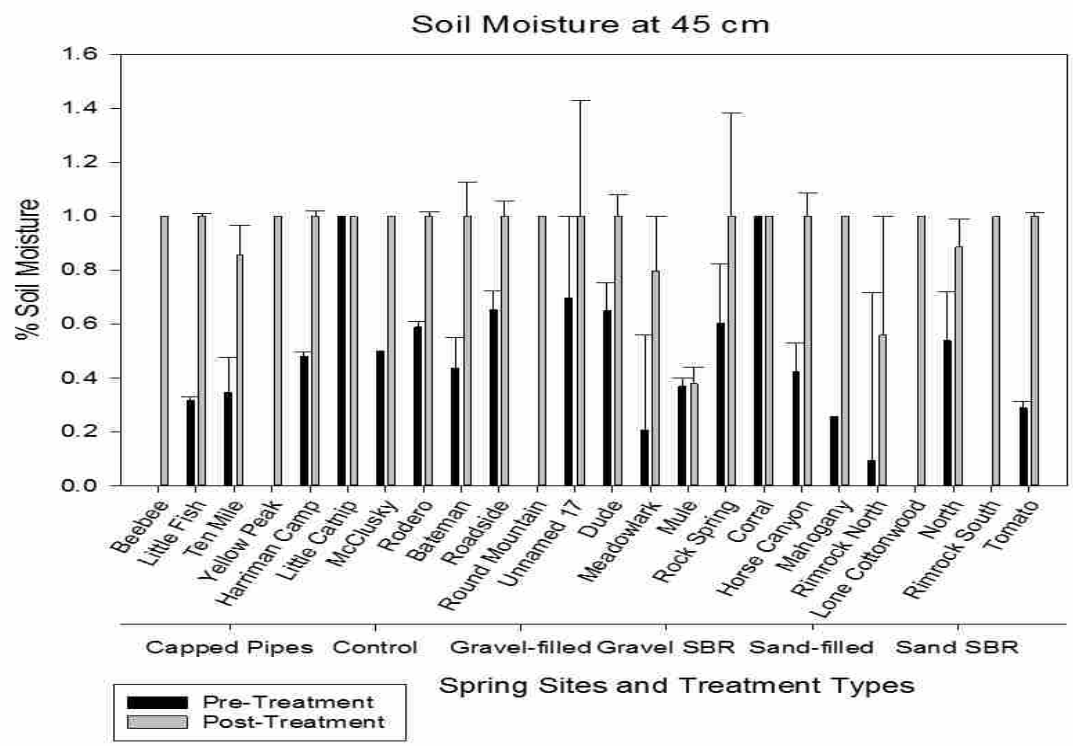


Figure 1.6. Soil moisture measurements at different depths: 0, 15, 30, and 45 cm. Wetland obligate plant communities grow when the water table is at a depth of 0-50 cm. Soil moisture measurements of 1.0 indicate that the water table was reached. Significance denoted by (*).

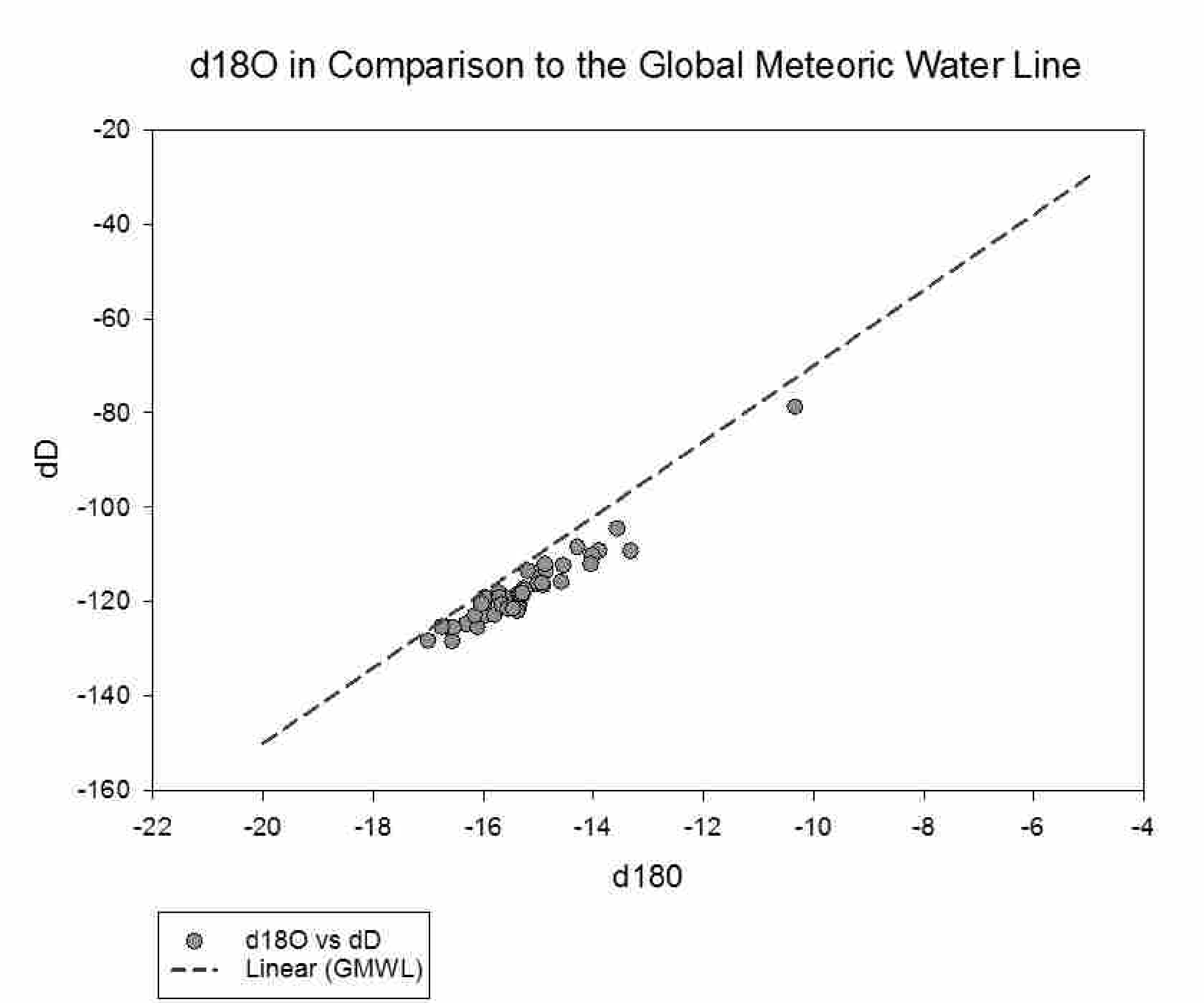
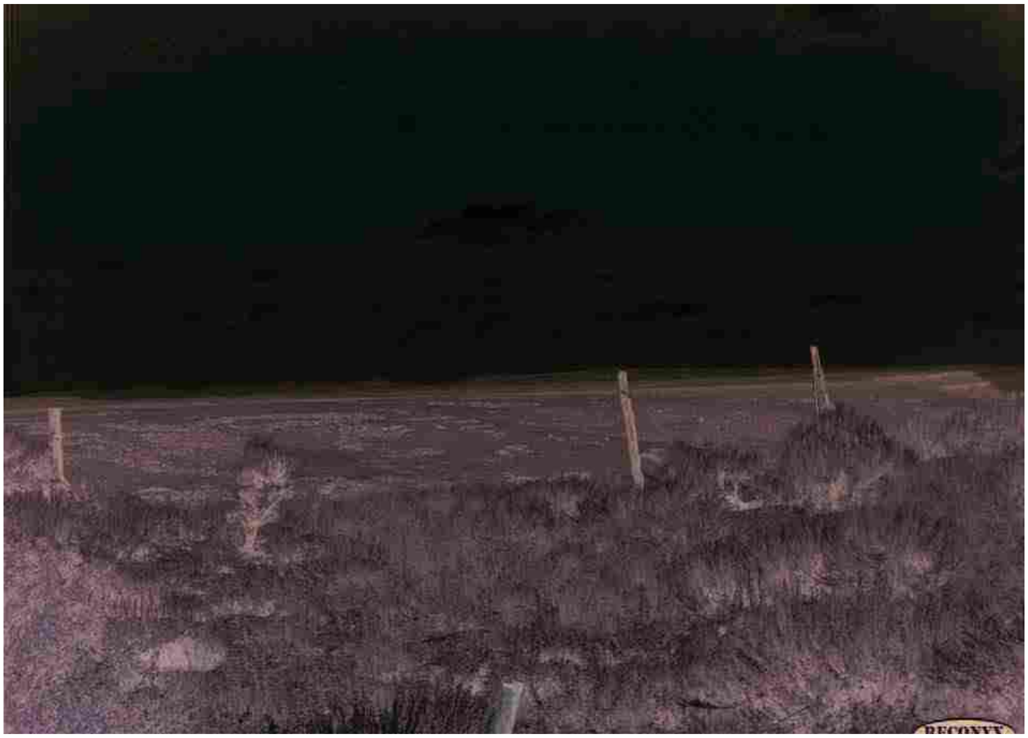


Figure 1.7. Representation of the $d^{18}O$ ratio of spring sites in comparison to the Global Meteoric Water Line (GMWL). The GMWL depicts how isotopes naturally separate in the atmosphere. The closer a ratio is to this line, the more likely it is to have originated from recent precipitation.

A



B

Figure 1.8. Round Mountain spring during pre-treatment (A) and post-treatment (B). Before treatment, Round Mountain was extremely dry around the spring source. After implementation, water had pooled at the springbox and was flowing downstream.

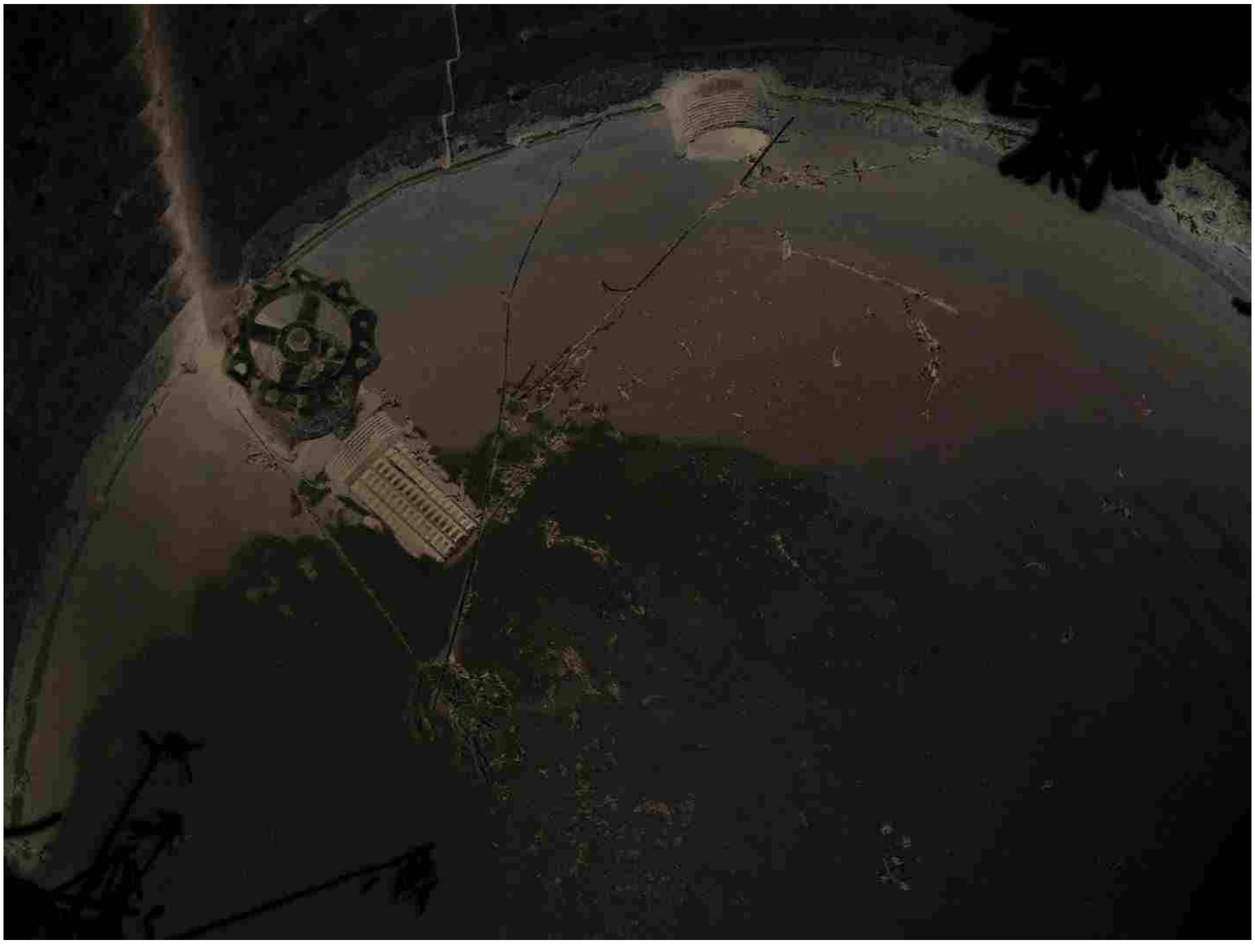
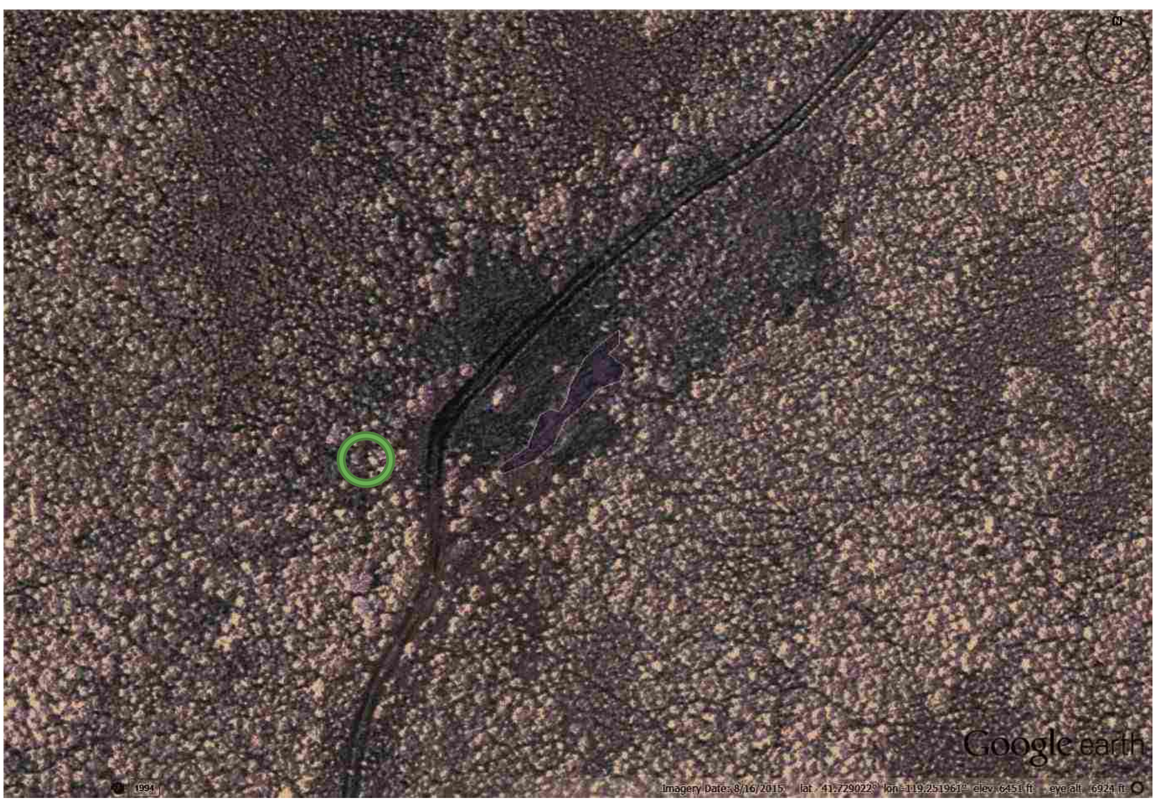
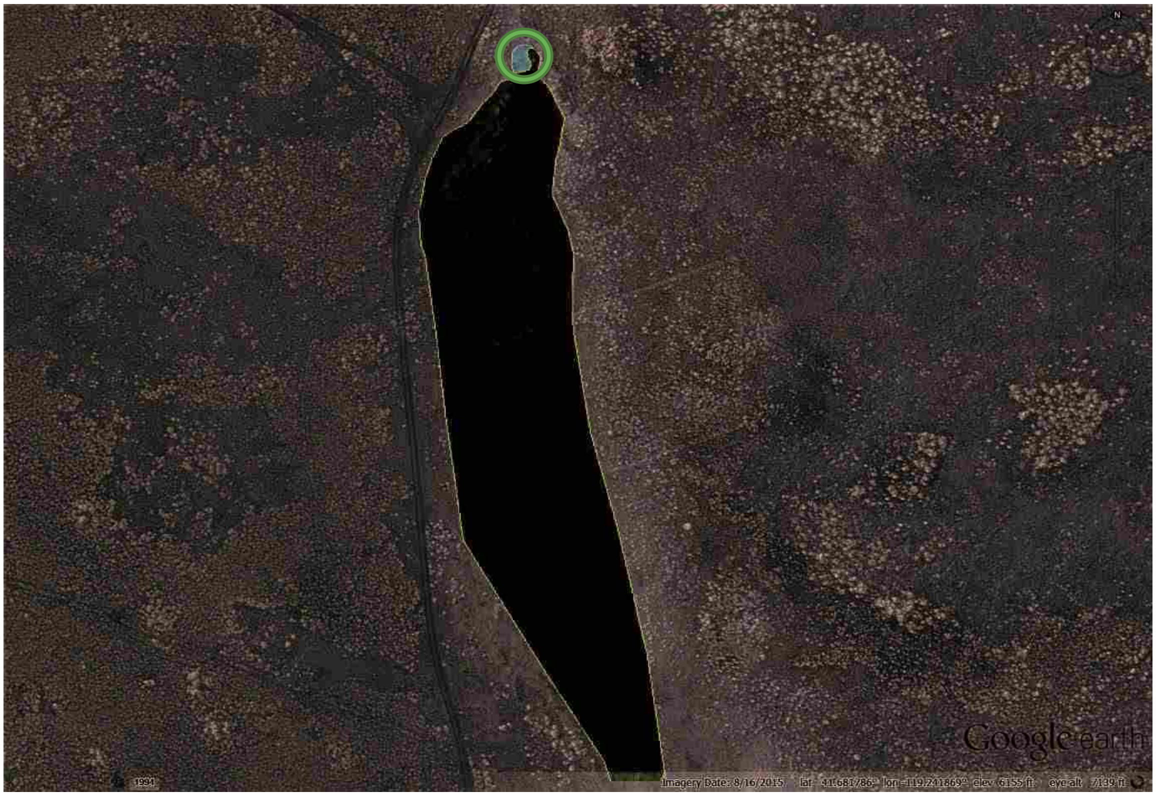


Figure 1.9. Interior of a springbox showing the pair of pipes that funnel water away from the spring source. One pipe, the outflow, diverts water to a cattle trough installed away (50-800 m) from the wet meadow. The other pipe discharges excess water out in the original wet meadow.

Bateman Spring



Beebee Spring

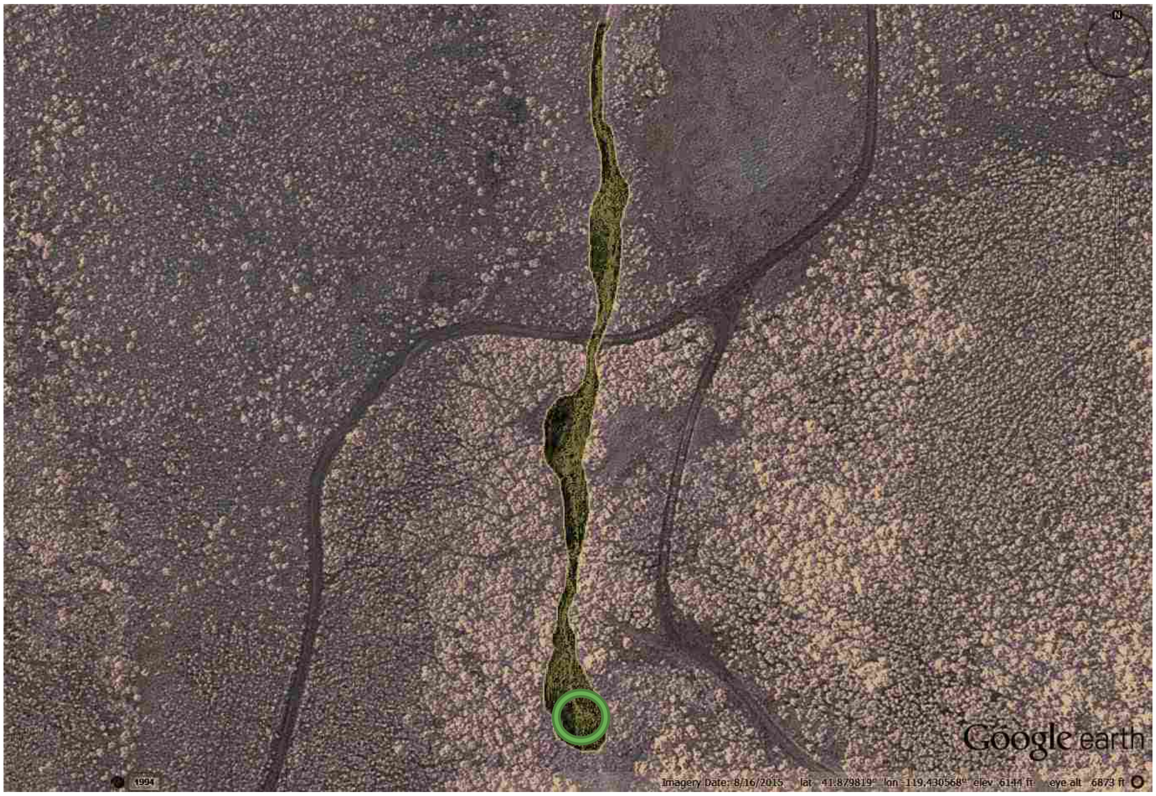


Corral Spring

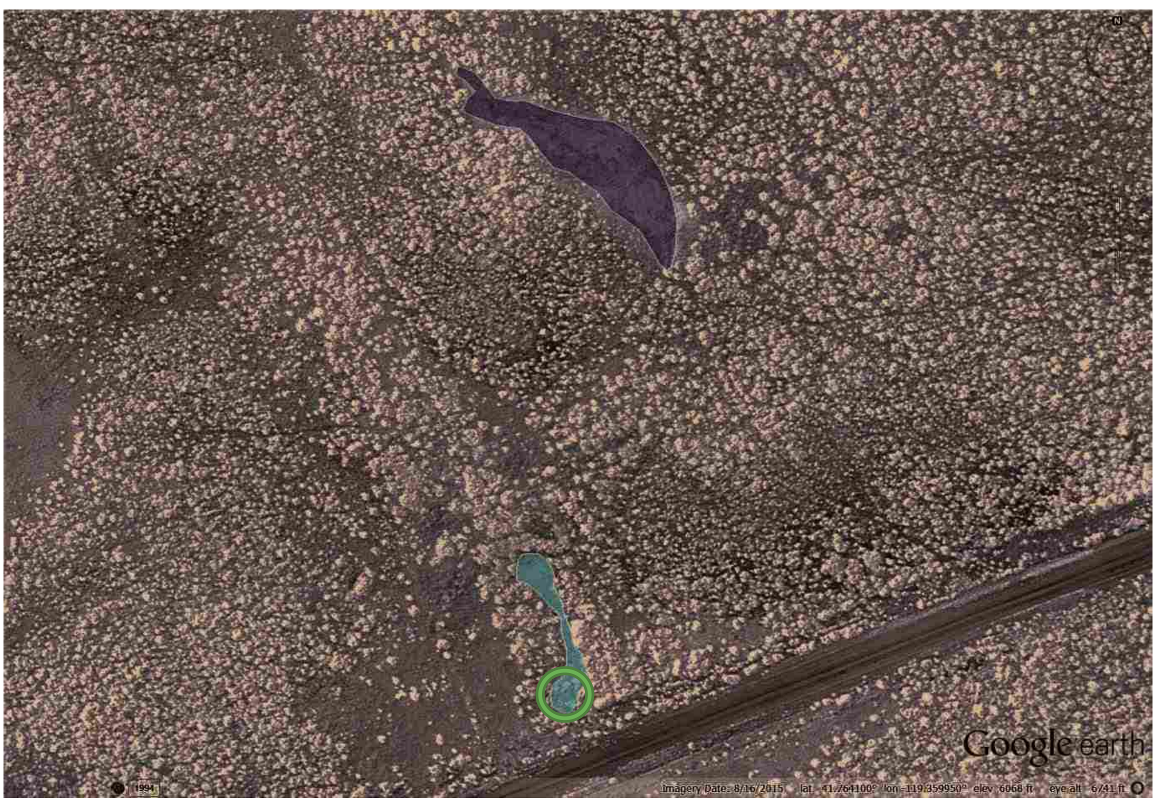
Dude Spring

Harriman Camp

Horse Canyon



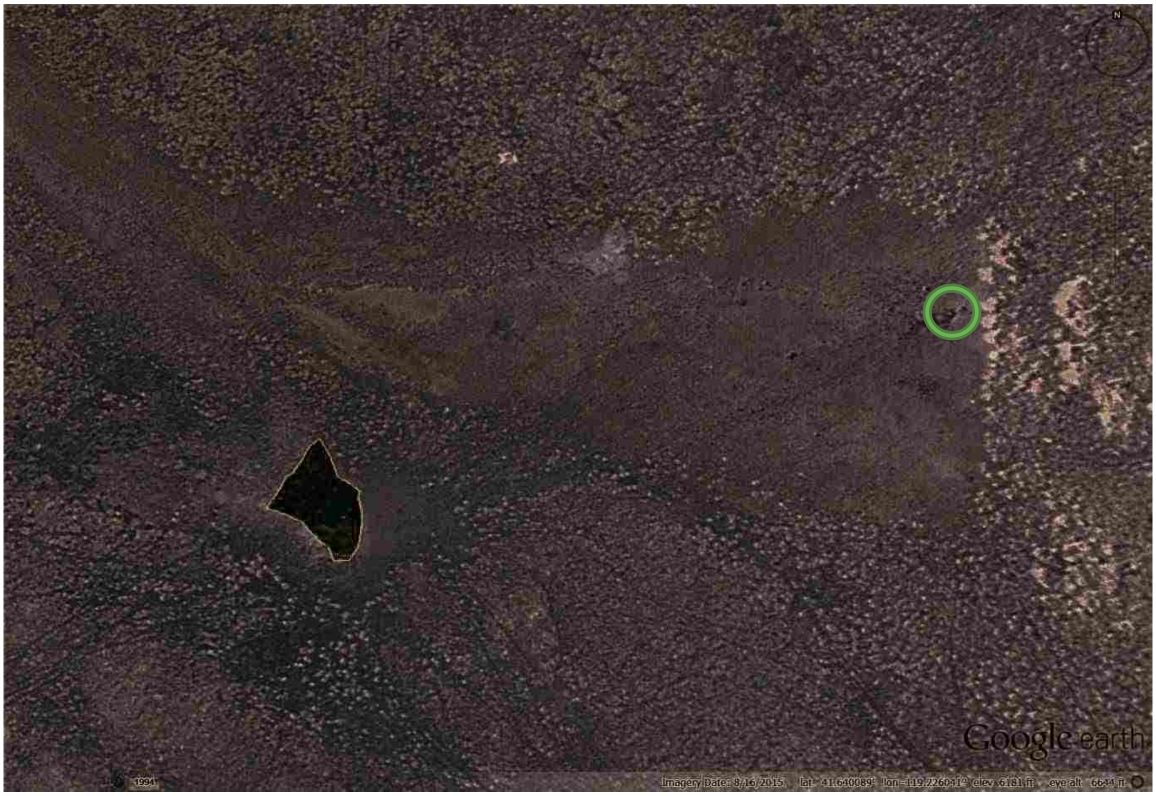
Little Catnip



Little Fish

Lone Cottonwood

Mahogany



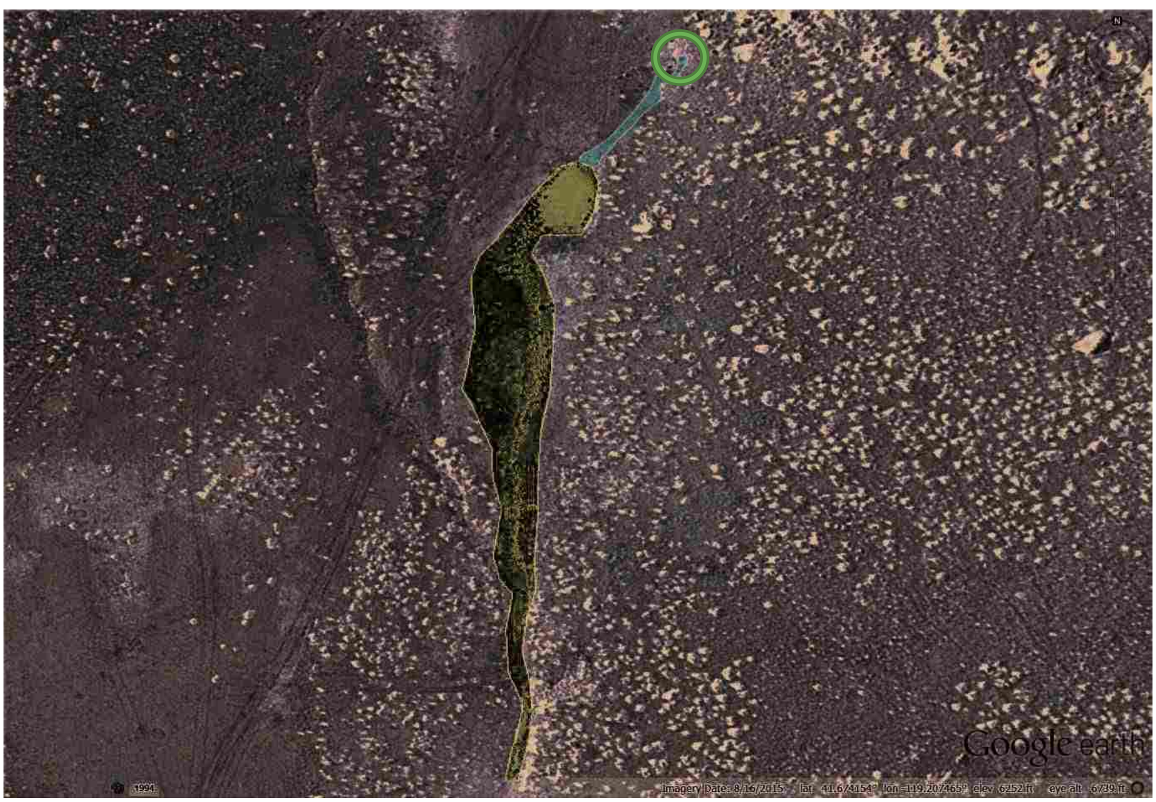
McClusky

Meadowlark


Google earth
1994
Imagery Date: 8/16/2015 lat 41.828240° lon -119.553078° elev 5931 ft eye alt 6370 ft
Mule

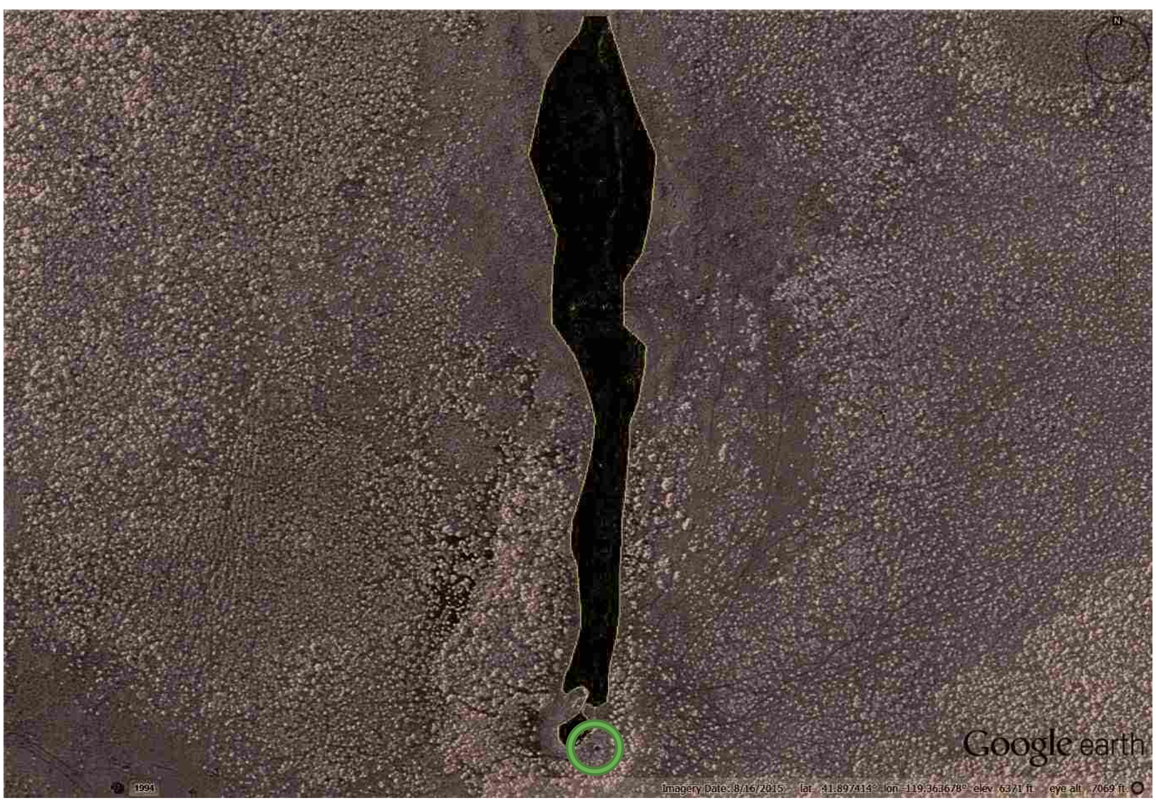

Google earth
1994
Imagery Date: 8/16/2015 lat 41.651160° lon -119.379418° elev 6439 ft eye alt 6824 ft
North

Rimrock North



Rimrock South

Roadside



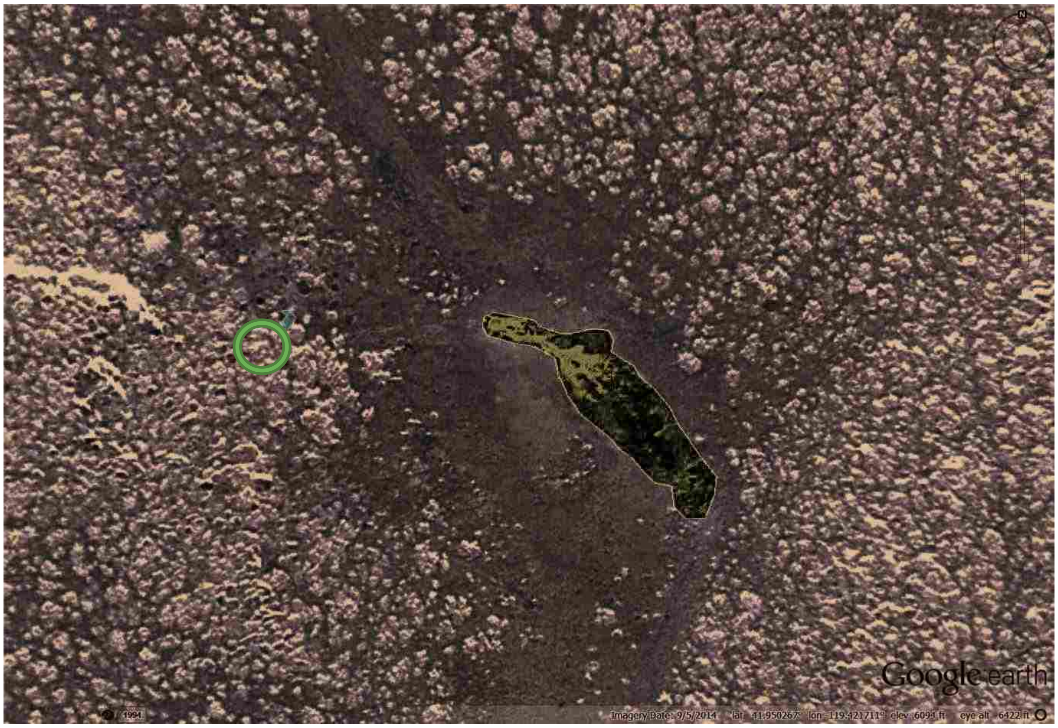
Rock Spring

Rodero

Round Mountain

Tenmile

Tomato



Unnamed 17

Yellow Peak

Figure 1.10. Overhead mapping of springs depicting where water was located during the pre- and post-treatment periods. Areas where water was found exclusively pre-treatment are outlined and colored in yellow, post-treatment in red and during both periods in blue. Springboxes are represented with a green ring.

TABLES

Table 1.1. Combined (riparian and upland) surface soil moisture measurements and p-values for treatment effects and yearly/climatic effects. Estimate column for treatment effects reports the percent change of post-treatment variables from the pre-treatment. The estimate column for yearly effects represents the percent change between the difference of the control (or other treatment) and the difference in the treatment. P-values for yearly effect were adjusted using a Tukey-Kramer adjustment. Significance denoted by (*).

Combined Surface Soil Moisture

	Treatment Effects				Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	0.2224	0.08065	0.0163*	**Control vs. Capped**	**-0.0327**	**0.1532**	**0.8343**	**0.9999**
	Single	0.1652	0.06201	0.0195*	**Control vs. Gravel SBR**	**-0.0877**	**0.1503**	**0.5694**	**0.9904**
Flow Group	High	0.2225	0.07767	0.0133*	**Control vs. Gravel-filled**	**0.0704**	**0.1233**	**0.5776**	**0.9913**
	Low	0.2389	0.06737	0.0036*	**Control vs. Sand SBR**	**0.0242**	**0.1222**	**0.8459**	**0.9999**
	Med High	0.1711	0.07078	0.031*	**Control vs. Sand-filled**	**-0.0856**	**0.1308**	**0.5245**	**0.9841**
	Med Low	0.1427	0.06884	0.0586*	Capped vs. Gravel SBR	-0.1204	0.1142	0.3111	0.8908
Treatment Type	Capped Pipes	0.1535	0.1026	0.1587	Capped vs. Gravel-filled	0.0377	0.1297	0.7757	0.9996
	Control	0.1861	0.09079	0.0611*	Capped vs. Sand SBR	-0.0085	0.1688	0.9608	1
	Gravel SBR	0.2739	0.1016	0.0184*	Capped vs. Sand-filled	-0.1182	0.1204	0.3439	0.9157
	Gravel-filled	0.1157	0.0807	0.1752	Gravel SBR vs. Gravel-filled	0.1581	0.1296	0.2442	0.82
	Sand SBR	0.1619	0.1017	0.1353	Gravel SBR vs. Sand SBR	0.1119	0.1685	0.5181	0.983
	Sand-filled	0.2717	0.08678	0.008*	Gravel SBR vs. Sand-filled	0.0022	0.1185	0.9858	1
					Gravel-filled vs. Sand SBR	-0.0462	0.1296	0.7272	0.999
					Gravel-filled vs. Sand-filled	-0.1560	0.1192	0.2133	0.7757
					Sand SBR vs. Sand-filled	-0.1098	0.1477	0.4706	0.9724

Table 1.2. Percent surface soil moisture measurements for all twenty-four spring sites. Pre-treatment and post-treatment values have been reported with the percent difference. Values reported as proportions. Significance denoted by (*).

	2016	2017	DIFFERENCE	F VALUE	PR> T
BATEMAN	0.03563	0.4129	0.37727	41.35	<.0001*
BEEBEE	0.01653	0.1241	0.10757	13.95	0.0004*
CORRAL	0.2662	0.4946	0.2284	24.03	<.0001*
DUDE	0.2099	0.3089	0.099	3.64	0.0583*
HARRIMAN CAMP	0.5247	0.6081	0.0834	0.97	0.3278
HORSE CANYON	0.2385	0.487	0.2485	9.03	0.0037*
LITTLE CATNIP	0.2052	0.343	0.1378	4.20	0.0430*
LITTLE FISH	0.02437	0.4322	0.40783	39.67	<.0001*
LONE COTTONWOOD	0.4851	0.4805	-0.0046	0.00	0.9663
MAHOGANY	0.2435	0.2763	0.0328	0.42	0.5176
MCCLUSKY	0.04174	0.3135	0.27176	19.81	<.0001*
MEADOWLARK	0.04274	0.4509	0.40816	47.93	<.0001*
MULE	0.08388	0.1765	0.09262	8.04	0.0055*
NORTH	0.09379	0.2739	0.18011	19.83	<.0001*
RIMROCK NORTH	0.01537	0.2091	0.19373	62.78	<.0001*
RIMROCK SOUTH	0.1307	0.5555	0.4248	58.39	<.0001*
ROADSIDE	0.2695	0.4801	0.2106	10.83	0.0013*
ROCK SPRING	0.2208	0.1497	-0.0711	3.16	0.0787*
RODERO	0.3762	0.3952	0.019	0.06	0.8146
ROUND MOUNTAIN	0.1034	0.6106	0.5072	37.46	<.0001*
TEN MILE	0.05605	0.2305	0.17445	6.55	0.0121*
TOMATO	0.02847	0.4147	0.38623	93.19	<.0001*
UNNAMED 17	0.07217	0.07004	-0.00213	0.01	0.9319
YELLOW PEAK	0.2143	0.2376	0.0233	0.23	0.6338

Table 1.3. Report of percent soil moisture at four depths (0, 15, 30, 45 cm) taken within a meter from the spring source. Treatment effects report significant increases in every category for every depth. However, yearly/climatic effects show that none of the treatments differs from the controls indicating that increased soil moisture is driven by precipitation. Significance denoted by (*).

Deep Soil Moisture-0 cm

	Treatment Effects				Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	0.3463	0.09299	0.0029*	**Control vs. Capped**	**-0.2367**	**0.1513**	**0.1438**	**0.6341**
	Single	0.4638	0.05719	<.0001*	**Control vs. Gravel SBR**	**-0.1116**	**0.1504**	**0.4725**	**0.9723**
Flow Group	High	0.3096	0.07748	0.0018*	**Control vs. Gravel-filled**	**0.3329**	**0.1362**	**0.0309**	**0.2156**
	Low	0.4943	0.07601	<.0001*	**Control vs. Sand SBR**	**0.1506**	**0.1388**	**0.2995**	**0.8785**
	Med High	0.5544	0.06590	<.0001*	**Control vs. Sand-filled**	**0.0768**	**0.1443**	**0.6043**	**0.9936**
	Med Low	0.2619	0.06732	0.0021*	Capped vs. Gravel SBR	-0.3483	0.1071	0.0069	0.0597*
Treatment Type	Capped Pipes	0.2826	0.1003	0.0156*	Capped vs. Gravel-filled	0.0962	0.112	0.4069	0.9493
	Control	0.5193	0.1042	0.0003*	Capped vs. Sand SBR	-0.0862	0.1699	0.6214	0.9949
	Gravel SBR	0.6309	0.1005	<.0001*	Capped vs. Sand-filled	-0.1599	0.1138	0.1854	0.724
	Gravel-filled	0.1863	0.08257	0.0435*	Gravel SBR vs. Gravel-filled	0.4445	0.112	0.0019	0.0179*
	Sand SBR	0.3687	0.1005	0.0032*	Gravel SBR vs. Sand SBR	0.2622	0.1701	0.1493	0.6473
	Sand-filled	0.4425	0.08428	0.0002*	Gravel SBR vs. Sand-filled	0.1884	0.1144	0.1255	0.5864
					Gravel-filled vs. Sand SBR	-0.1824	0.1459	0.2351	0.8049
					Gravel-filled vs. Sand-filled	-0.2561	0.1088	0.0365	0.246
					Sand SBR vs. Sand-filled	-0.0738	0.146	0.6226	0.995

Deep Soil Moisture-15 cm

	Treatment Effects				Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	0.4712	0.07594	<.0001*	**Control vs. Capped**	**-0.0523**	**0.1288**	**0.6926**	**0.9982**
	Single	0.4814	0.04932	<.0001*	**Control vs. Gravel SBR**	**-0.1274**	**0.1248**	**0.3295**	**0.9016**
Flow Group	High	0.3215	0.07202	0.001*	**Control vs. Gravel-filled**	**0.1266**	**0.1135**	**0.2883**	**0.8652**
	Low	0.6686	0.06333	<.0001*	**Control vs. Sand SBR**	**0.1343**	**0.113**	**0.2597**	**0.8336**
	Med High	0.5458	0.05478	<.0001*	**Control vs. Sand-filled**	**-0.0939**	**0.1145**	**0.4293**	**0.9575**
	Med Low	0.3694	0.05507	<.0001*	Capped vs. Gravel SBR	-0.1797	0.08983	0.0708	0.3994
Treatment Type	Capped Pipes	0.4394	0.08417	0.0003*	Capped vs. Gravel-filled	0.0743	0.09324	0.4421	0.9622
	Control	0.4916	0.08436	0.0001*	Capped vs. Sand SBR	0.0821	0.142	0.5751	0.9906
	Gravel SBR	0.619	0.08389	<.0001*	Capped vs. Sand-filled	-0.1462	0.104	0.1873	0.7234
	Gravel-filled	0.365	0.06867	0.0002*	Gravel SBR vs. Gravel-filled	0.2540	0.0933	0.0198	0.147
	Sand SBR	0.3573	0.08393	0.0013*	Gravel SBR vs. Sand SBR	0.2617	0.1424	0.0932	0.4823
	Sand-filled	0.5856	0.07758	<.0001*	Gravel SBR vs. Sand-filled	0.0335	0.1037	0.753	0.9994
					Gravel-filled vs. Sand SBR	0.0077	0.1217	0.9506	1
					Gravel-filled vs. Sand-filled	-0.2206	0.0979	0.0457	0.2888
					Sand SBR vs. Sand-filled	-0.2283	0.124	0.0927	0.4806

Deep Soil Moisture-30 cm

Category		Treatment Effects			Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	0.4697	0.1114	0.0014*	**Control vs. Capped**	**-0.0800**	**0.1956**	**0.6902**	**0.9981**
	Single	0.5054	0.07394	<.0001*	**Control vs. Gravel SBR**	**-0.0407**	**0.188**	**0.8326**	**0.9999**
Flow Group	High	0.4118	0.1056	0.0025*	**Control vs. Gravel-filled**	**0.1398**	**0.1669**	**0.4203**	**0.9539**
	Low	0.6289	0.0964	<.0001*	**Control vs. Sand SBR**	**0.0907**	**0.1581**	**0.5774**	**0.9908**
	Med High	0.4934	0.08043	<.0001*	**Control vs. Sand-filled**	**-0.0285**	**0.1664**	**0.8671**	**1**
	Med Low	0.4159	0.08029	0.0003*	Capped vs. Gravel SBR	-0.1207	0.1318	0.3793	0.9343
Treatment Type	Capped Pipes	0.4477	0.1276	0.0049*	Capped vs. Gravel-filled	0.0597	0.1387	0.6751	0.9976
	Control	0.5278	0.1217	0.0012*	Capped vs. Sand SBR	0.0107	0.2137	0.9609	1
	Gravel SBR	0.5684	0.1237	0.0008*	Capped vs. Sand-filled	-0.1086	0.156	0.501	0.9786
	Gravel-filled	0.388	0.1011	0.0028*	Gravel SBR vs. Gravel-filled	0.1805	0.1365	0.213	0.7683
	Sand SBR	0.437	0.1235	0.0046*	Gravel SBR vs. Sand SBR	0.1314	0.2101	0.5443	0.9865
	Sand-filled	0.5563	0.1138	0.0005*	Gravel SBR vs. Sand-filled	0.0122	0.1527	0.9379	1
					Gravel-filled vs. Sand SBR	-0.0490	0.1798	0.7902	0.9997
					Gravel-filled vs. Sand-filled	-0.1683	0.1439	0.2669	0.8421
					Sand SBR vs. Sand-filled	-0.1193	0.182	0.5257	0.9835

Deep Soil Moisture-45 cm

Category		Treatment Effects			Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	0.2864	0.1408	0.0814*	**Control vs. Capped**	**-0.1633**	**0.2632**	**0.5547**	**0.9857**
	Single	0.5431	0.07179	0.0001*	**Control vs. Gravel SBR**	**0.1813**	**0.2098**	**0.4162**	**0.9441**
Flow Group	High	0.4015	0.1059	0.0068*	**Control vs. Gravel-filled**	**0.2829**	**0.1648**	**0.1297**	**0.5615**
	Low	0.4569	0.1474	0.0173*	**Control vs. Sand SBR**	**-0.0109**	**0.1491**	**0.9438**	**1**
	Med High	0.4268	0.08177	0.0012*	**Control vs. Sand-filled**	**0.1819**	**0.1963**	**0.3849**	**0.9272**
	Med Low	0.3738	0.0887	0.004*	Capped vs. Gravel SBR	0.0180	0.1809	0.9234	1
Treatment Type	Capped Pipes	0.3846	0.1995	0.0953*	Capped vs. Gravel-filled	0.1197	0.1891	0.5469	0.9844
	Control	0.5478	0.1146	0.002*	Capped vs. Sand SBR	-0.1741	0.2853	0.5608	0.9866
	Gravel SBR	0.3666	0.1499	0.0444*	Capped vs. Sand-filled	0.0187	0.2202	0.9348	1
	Gravel-filled	0.2649	0.09939	0.0322*	Gravel SBR vs. Gravel-filled	0.1017	0.1481	0.5146	0.9779
	Sand SBR	0.5587	0.1278	0.0033*	Gravel SBR vs. Sand SBR	-0.1922	0.2432	0.4553	0.9606
	Sand-filled	0.3659	0.1487	0.0434*	Gravel SBR vs. Sand-filled	0.0007	0.1767	0.9971	1
					Gravel-filled vs. Sand SBR	-0.2938	0.188	0.1621	0.6422
					Gravel-filled vs. Sand-filled	-0.1010	0.159	0.5454	0.9841
					Sand SBR vs. Sand-filled	0.1928	0.2269	0.4236	0.9476

Table 1.4. Measurements of flow for categories of spring type, flow group and treatment types. The left side of the table reports estimates and p-value for treatment effects. Yearly/climatic effects are reported on the right. Yearly effects accounts for climate by comparing the amount of change in treatments to the amount of change in the control. Significance denoted by (*).

Flow

		Treatment Effects			Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	0.0083	0.01564	0.6026	**Control vs. Capped**	**-0.0621**	**0.03075**	**0.0644**	**0.382**
	Single	0.0154	0.01202	0.2226	**Control vs. Gravel SBR**	**0.0520**	**0.02866**	**0.093**	**0.4908**
Flow Group	High	0.0303	0.02372	0.2237	**Control vs. Gravel-filled**	**0.0623**	**0.02271**	**0.0168**	**0.1317**
	Low	-0.0028	0.01516	0.8549	**Control vs. Sand SBR**	**0.0423**	**0.02291**	**0.088**	**0.4732**
	Med High	0.0054	0.01368	0.7	**Control vs. Sand-filled**	**0.0472**	**0.02722**	**0.1066**	**0.535**
	Med Low	0.0146	0.01569	0.3684	Capped vs. Gravel SBR	-0.0102	0.02322	0.6682	0.9974
Treatment Type	Capped Pipes	-0.0060	0.02128	0.7841	Capped vs. Gravel-filled	0.0002	0.02756	0.9953	1
	Control	0.0562	0.017	0.0057*	Capped vs. Sand SBR	-0.0199	0.03576	0.5878	0.9923
	Gravel SBR	0.0042	0.0196	0.8325	Capped vs. Sand-filled	-0.0149	0.02274	0.5229	0.9839
	Gravel-filled	-0.0061	0.01585	0.7056	Gravel SBR vs. Gravel-filled	0.0104	0.02528	0.6889	0.9982
	Sand SBR	0.0139	0.02068	0.5127	Gravel SBR vs. Sand SBR	-0.0097	0.03328	0.7756	0.9996
	Sand-filled	0.0090	0.01811	0.6286	Gravel SBR vs. Sand-filled	-0.0047	0.02366	0.8442	0.9999
					Gravel-filled vs. Sand SBR	-0.0200	0.02519	0.4406	0.9633
					Gravel-filled vs. Sand-filled	-0.0151	0.02505	0.5572	0.989
					Sand SBR vs. Sand-filled	0.0049	0.03155	0.8778	1

CHAPTER 2

Biologic Responses to Anthropogenically-altered Great Basin Lentic Springs and Wet Meadows

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ABSTRACT

Springs in the Great Basin account for only 1-3% of the landscape, but support a disproportionately high amount of biodiversity. They are often the only water sources in the region and are highly valued by humans and wildlife for drinking water, forage and cover. These areas are threatened by human-related activities, climate and drought. Diversions for agriculture, livestock and drinking water results in dewatering of springs and their adjoining wet meadows. As water is drained from the local aquifers that supply springs, the underlying water table drops. This results in a shift in the plant community from wetland-obligate sedges and rushes to more drought tolerant grasses and shrubs. This new community does not have the root mass needed to hold onto soils leading to erosion of the thalweg. Incision results and this drives the water table further underground. Water is no longer stored in the wet meadow vegetation and less water is available for plant and wildlife use in the hotter, drier summer months. The purpose of this study is to determine the effect of restoration techniques on riparian plant communities fed by Great Basin springs. Twenty-four spring sites were chosen on the Sheldon National Wildlife Refuge in northwestern Nevada. We implemented six different restoration treatment designs, which were randomly assigned to our spring sites. Cover, frequency, biomass, number of wildlife species and number of wildlife visits were measured. We observed no changes in cover across any of our sites. The frequency of forbs increased for four of our treatments, but the frequency of other functional groups did not. We noted an increase in the total amount of biomass for our treatments

that retained a metal casing of the springbox. We suggest that the metal casing restricted horizontal movement of water, funneling it up and onto the surface of the riparian zone. This may have resulted in greater accessibility for plant use by wetting the area around their roots. We did not see a dramatic shift in plant community for the altered communities present. However, studies in Idaho indicate that plant community recovery can occur anywhere from zero to 10 years after restoration. These sites have had several decades to degrade and change and it may take a similar amount of time to shift back. Wildlife visits and number of species decreased across many of our sites and disruption of site for treatment implementation may be to blame. Yearly effect indicates that many of these changes may be driven by precipitation as well as treatment. Additionally years of data collection are needed to determine the true influence of treatments and yearly variation on spring restoration.

INTRODUCTION

The Great Basin covers 520,000 km^2 of area, encompassing land from six states in the western US (Chambers and Miller 2004). Created from the collision of volcanic fire and receding glacial ice, the ecosystems of the Great Basin are dominated by sagebrush and salt-desert vegetation, containing a grand cast of plant and animal species (Herbst 19966, Rodgers and Tiehm 1979). In spite of the overall biodiversity within the Great Basin, a high proportion occurs primarily in resource pockets, existing as small islands surrounding surface water sources in a vast sagebrush sea (Jewett et al. 2004, Sada 2008, Naiman et al. 1993, Patten et al. 1998). Great Basin desert springs and streams cover 1-3% of sagebrush ecosystems, but have a disproportionately high biotic diversity and exhibit significant effects on surrounding ecosystems (Sada 2008, Wyman et al. 2006, Buckhouse and Elmore 1993, Barquin and Scarsbrook 2008).

Riparian ecosystems are in constant demand by livestock, wildlife and humans for drinking water. Unless carefully managed, anthropogenic related activities may degrade the springs, potentially causing depleted and altered plant communities, destabilized banks and increased erosion (Wyman et al. 2006, Lewis et al. 2003, Prichard et al. 1994, Barquin and Scarsbrook 2008, Naiman and Decamps 1997, Sada 2008). Any disturbance that changes properties of soil surface and groundwater could directly affect the distribution of plant species within the wet meadows surrounding these water sources (Chambers and Miller 2004, Perkins et al. 1984). Additionally, the change in plant community structure often affects available forage for wildlife and livestock (Naiman and Decamps 1997, Lewis et al. 2003, Wyman et al. 2006).

Riparian ecosystems are vital for sustaining terrestrial and aquatic plant and wildlife populations (USFWS 2012, Wyman et al. 2006, Sada et al. 2001). These areas, often referred to as “riparian zones”, are the transition between standing water and uplands where free water is lacking (Svejcar 1997, Lewis et al. 2003, Naiman and Decamps 1997, Patten et al. 2008). Riparian zones are a complex of soils and vegetation formed by gradients of soil moisture from groundwater flows (Svejcar 1997, Patten et al. 2008). These small patches of hydrophilic vegetation and moisture availability are particularly important resources in arid and semi-arid environments such as the sagebrush-steppe regions of the Great Basin. Within this region, riparian zones are highly productive areas that occupy a relatively small proportion of the overall landscape (Wyman et al. 2003). This production is critical to the plants and wildlife that depend of these riparian ecosystems for survival. A study in southeastern Oregon by Oakley et al. (1985) found that 80% of wildlife species in the area were directly dependent on riparian zones more than other habitats in sagebrush ecosystems (Odum 1971). Greater sage grouse (*Centrocerus urophasianus*) hens use riparian/wet meadows as late summer brooding sites. The nutrient rich riparian forbs are vital forage for growing sage grouse chicks before they transition to eating sagebrush. The greater abundance of forbs in wet meadow during the late summer is especially important as forbs begin to dry out in the uplands (Wallestad 191, Connelly et al. 1988, Savage 1969).

Riparian zones are affected by many factors: stream size, geology, hydrology, seasonal and yearly climate patterns, elevation, gradients, size of watershed, upland vegetation, prior land management and water use patterns (Svejcar 1997, Leonard et al. 1992, Welsch et al. 1995, Lewis et al. 2003, Chambers and Miller 2004, Sada 2008, Patten et al. 2008). The combination of all these elements makes each spring and riparian zone unique (Buckhouse and Elmore 1993). Subsequently, changes to a single factor may result in dynamic changes in the biological and hydrological properties of the riparian zone (Gray et al. 1992). Additionally, the size of a watershed and the elevation of a spring have a dramatic impact on the amount of flow produced. The result is that providing a "one size fits all" restoration strategy is difficult and likely ineffective (Buckhouse and Elmore 1993, Lewis et al. 2003, Patten et al. 2008).

Aside from providing resources for wildlife, riparian zones provide important ecosystem services (Wyman et al. 2006). These services are facilitated by riparian vegetation, which acts like a control valve for the entire ecosystem, influencing water quality through filtering and modified seasonal flows (Wyman et al. 2006, Lewis et al. 2003). During spring runoff, riparian vegetation dampens powerful high flows, mitigating its erosive effect (Lewis et al. 2003, Naiman and Decamps 1997). Along with slowing erosion, riparian vegetation can capture sediments, an ecosystem service that enhances site restoration, allowing degraded and eroded channels to fill in over time (Naiman and Decamps 1997, Wayman et al. 2006). The rhizomatous reproductive strategy of many wetland obligate species form dense mats of roots and stems that facilitate the development of a wet meadow acting as an organic sponge, capturing spring precipitation and runoff (Wyman et al. 2006, Micheli and Kirchner 2002, Winward 2000, Manning et al. 1989).

Moisture remains trapped in the wet meadow for a longer duration effectively extending the availability of water for vegetation use. Spring water lasts longer, extending into the drier summer months as water is slowly released from saturated riparian soils (Wyman et al. 2006, Lewis et al. 2003). Higher soil water content in the late summer typically results in prolonged high quality forage in riparian zones, especially as upland vegetation desiccates producing less palatable and less nutritious forage (Naiman and Decamps 1997, Wyman et al. 2006).

In the spring, livestock generally disperse evenly throughout the uplands because of higher quality forage and greater surface water availability. However, during the summer, upland surface water and soils dry out and water availability for plants and animals becomes concentrated in riparian areas (citation). Subsequently, livestock congregate in these areas because of the greater water and high quality forage availability. Without sufficient soil moisture to support the biomass needed to sustain grazing, wet meadows can quickly become over-grazed and defoliated (Wyman et al. 2006). Many springs in the Great Basin no longer consist of riparian vegetation due to over-grazing by large non-native ungulates (Fleischner 1994, Sada et al. 2001).

Anthropogenic effects have had a historic long-term effect on riparian systems in the Great Basin (Sada et al. 2001). Human activities over the last century (flow regulation, surface and groundwater withdrawals, agricultural activities and recreation) threaten the sustainability of riparian ecosystems (Patten 1998, Myers and Resh 20021, Barquin and Scarsbrook 2008) by changing the abundance and quality of water feeding springs and wet meadows (Grimm et al. 1997). The result is groundwater depletion and dewatering of springs and wet meadows (Sada et al. 2001, Burk et al. 2005). Consequently, many riparian systems in the western US are considered marginal or low in ecological value, no longer dampening high flows or assisting in the recharging of subsurface aquifers (Elmore and Beschta 1987). Springs can no longer support wetland-obligate vegetation that traps sediments, curtailing erosion (citation). Riparian soils lose the ability to act as a bio-sponge and slowly release water (Naiman and Decamps 1997, Prichard et al. 1994, Lewis et al. 2003)

Changes in abundance of water supplied to a spring/seep can affect the vegetation of the surrounding wet meadow (Wyman et al. 2006, Prichard et al. 1994, Chambers and Miller 2004, Perkins et al. 1984). The loss of stabilizing species, such as rushes and sedges, lead to increased soil erosion and incised stream channels (Wyman et al. 2006, Micheli and Kirchner 2002, Winward 2000). Incision disrupts the physical and hydrological characteristics of a spring system (Miller et al. 2001). When streams become incised, the overall stream surface decreases in elevation within the channel as the stream surface levels with that of the base level of the surrounding groundwater. As the stream surface lowers, the local water table will adjust to match this new level (Wyman et al. 2006, Naiman and Decamps 1997, Chambers and Miller 2004, Jewett et al. 2004). This increases the depth of the capillary fringe, pulling it out of reach of the roots of many wetland-dependent plant species (Lewis et al. 2003, Hammersmark et al. 2009, Castelli et al. 2000). In addition, the overall extent and influence of the riparian corridor is decreased which results in a progressive loss of the wet meadow complex (Sada 2008, Lewis et al. 2003, Patten 1998, Chambers and Miller 2004). This degradation is evidenced by the formation of a thalweg in lentic springs and seeps which tend to produce more overland shallow sheet flow than have a defined stream channel. This degradation results in soil loss and decreased forage (Sada 2008, Naiman and Decamps 1997). If not addressed, springs and wet meadow plant communities can alter past the point of self-repair (Wyman et al. 2006) and cross ecological thresholds. Once crossed, these thresholds are not easily reversed. Altered plant communities are very stable once established (Stringham et al. 2003) and diminished water tables may it make it impossible to return to historic conditions (Wyman et al. 2006). These altered communities are functionally different from wetland-obligate communities. Communities dominated by grasses are 6 to 10 times less effective than native rushes and sedges in holding onto riparian soils (Micheli and Kirchner 2002, Wyman et al. 2006).

Over the past century, naturally occurring springs and seeps located in the sagebrush ecosystems of western North America have provided a reliable water sources for livestock. Early pioneers and homesteaders began by modifying springs and streams to create conditions more suitable for cattle (USFWS 2012, Wyman et al. 2006, Chamberlin and Doverspike 2001, Sada et al. 2001). During these early years, many springs were developed by removing soil around the spring or seep to impede water flow and create catchment basins for use by livestock (Collins 2015, USDI 1990). Some springs were capped with springboxes to transport water away from the spring to fill nearby livestock troughs (USDI 1990). The impact these developments have had on surrounding ecosystems include lowered water table levels, reduced surface flow, decreased soil moisture availability, and altered plant community composition (Collins 2015, Wyman et al. 2006, Lewis et al. 2003, Prichard et al. 1994, Sada et al. 2001, Chambers and Miller 2004).

Grazing and other human uses have required continued development of many springs and seeps (Sada et al. 2001). These developments can result in degradation and dewatering of the springs and wet meadows (Patten 2008, Burk et al. 2005, Erman 2002). Springboxes placed at the spring source, in particular, could cause a form of single-point incision. Single-point incision acts similarly to stream incision by lowering the “stream surface” to an elevation below the surrounding water table. Just as in stream incision, this draws down the base level of the surrounding groundwater, which alters the overall water table (Chambers and Miller 2004, Hancock 2002). Drawing water away to a trough decreases the abundance of water available to the wet meadow (Sada 2008). Many springboxes have overflow pipes that release excess water into the riparian zone, often several meters away from the spring source. Additionally, the water is released from a pipe opening which encourages the formation of channels. Sheet flow is no longer possible in such systems where all the discharge pours out from a single concentrated point. This is especially problematic when that water exits the pipe with increased energy and a greater ability to move sediment, causing incisions in the wet meadow (Wyman et al. 2006).

Therefore, single-point incision initially causes a drop in the water table. This drop can be further exacerbated by an incision through the riparian corridor that drives the water table deeper underground. The wet meadow becomes dewatered and disconnected from its flood plain (Hancock 2002, Wyman et al. 2006). The degradation of these sites can be seen in the plant communities comprising the wet meadow (Chambers and Miller 2004, Prichard et al. 1994, Perkins et al. 1984). Sagebrush encroachment into the riparian zone is an indicator that the water table has dropped below a level that is accessible by obligate and facultative wetland species. Even springs dominated by *Poa pratensis* and *Juncus balticus* are evidence of dewatering of a wet meadow site (Hammersmark et al. 2009, Castelli et al. 2000).

We anticipate three signs of biologic restoration. First, returning water to the riparian zone encourages the recruitment, reestablishment and expansion of obligate wetland species (Barquin and Scarsbrook 2008, Sada et al. 2001, Lewis et al. 2003, Wyman et al. 2006). Once established, their roots can influence soil retention and deposition that can channelize thalwegs, which fill in and potentially raise the surrounding water table. Through this process, hydrologic recovery can benefit plant species by reconnecting the floodplain that gives them access to water and nutrients (Elmore and Beschta 1987, Sada 2008, Naiman and Decamps 1997, Barquin and Scarsbrook 2008, Lewis et al. 2003, Wyman et al. 2006).

Previous studies have also indicated that restoring the hydrology of a wet meadow leads to a shift in plant species from facultative wetland species, such as *J. balticus*, to more native sedges (Carex spp.) and rushes that are obligate wetland species (Hammersmark et al. 2009). Although grasses may provide a lot of cover, their root systems are ineffective in holding banks in place. On the other hand, sedges are very good at holding onto sediment and increase bank stability (Micheli and Kirchner 2002). Manning et al. (1989) showed that *Carex nebrascensis* Dewey have extremely dense root systems, with 200 cm per 3 cm of soil. This comes out to 35 km of roots in a single 30 cm x 30 cm x 40 cm block of soil. The second sign of restoration that we would anticipate is an increase in biomass as the plant community that is already established has more access to useable water (Martin and Chambers 2001, Wyman et al. 2006). A third indicator would be an increase in wildlife usage and diversity as water returns to the site, prompted by an increase in surface water availability, increased forage availability and higher forage quality (Wyman et al. 2006, Naiman and Decamps 1997).

There are 130 identified springs in the Sheldon National Wildlife Refuge (SNWR) in Nevada, and many of those sites have been developed (183 water developments) by early settlers and the U.S. Fish and Wildlife Service. Reservoirs, dugouts, stock pond, berms and springboxes were originally installed to provide water for wildlife, livestock and human use. However, as of 2016, most of these water developments have been abandoned (USFWS 2012). Overgrazing and diversion of groundwater caused by these water developments dewaters springs and surrounding wet meadows, leading to riparian systems that can no longer provide vital ecologic functions

(Wyman et al. 2006, Sada 2008). Degraded springs in SNWR are unable to store water, support healthy wetland plant communities, prevent soil erosion or provide forage and cover for wildlife (USFWS 2012, Lewis et al. 2003, Wyman et al. 2006). With most of the spring discharge being funneled to offsite cattle troughs, water flowing into the wet meadows has been reduced or even eliminated (Erman 2002). Drought tolerant grass and shrub encroachment into the riparian zone is a clear sign that the underlying water table is dropping out of reach of moisture dependent wetland sedges and rushes (Castelli et al. 2000, Hammersmark et al. 2009, USFWS 2012). Shifts in the plant community indicate that many of these springs in SNWR may have crossed an ecological threshold (Wyman et al. 2006), and altered plant communities may prove difficult to cast out once established (Stringham et al. 2003). However, reestablishing surface sheet flows allows water to soak into the soil and recharge the local aquifer. This in turn raises the water table, bringing it back to a point conducive for recolonization of natural wet meadow plant communities (Barquin and Scarsbrook 2008, Schumm 1977, Jensen et al. 1989, Lewis et al. 2003). Restoring proper ecological functions to spring systems in SNWR will benefit wildlife dependent on the quality of these areas for forage, shelter and water (Oakley et al. 1985, Stevens et al. 1997), such as pronghorn and Greater sage grouse (USFWS 2012).

The purpose of this study is to determine the impact of spring restoration on wet meadow plant communities that are fed by altered lentic spring systems and dependent wildlife species. We wanted to know if there is an immediate response in biomass, shifts in community composition or wildlife usage during the immediate post-treatment recovery, and if restoration techniques have differing effects on those treatment responses. We believe that restoring the underlying hydrology of altered springs and wet meadows will have a restorative effect on the biological aspects of the system, resulting in a return to appropriate wetland plants and increased wildlife usage.

METHODS

Site Description

The Sheldon National Wildlife Refuge (SNWR), managed by the U.S. Fish and Wildlife Service (Figure 2.1), is one of the last stretches of uninterrupted sagebrush-steppe ecosystem in the western United States (Collins 2016). It straddles the county lines of Washoe and Humboldt counties in northwestern corner of Nevada (41.806413, -119.232577). SNWR was established in the 1930s as a wildlife refuge, specifically aimed at conserving the then endangered pronghorn (*Antilocarpa americana*). Over 270 wildlife species are found in SNWR, including many that are threatened or endangered. The refuge is also a stop on the migratory path of many bird species (USFWS 2016). Salt-desert vegetation dominates the landscape in SNWR's northeastern corner sitting at 1326 m above sea level, before rising 900 meters from the desert floor to a basalt plateau (2183 m) (Collins 2016).

On top of the plateau, the landscape consists mostly of sagebrush-steppe vegetation, which is dominated by shrubs, such as big sagebrush (*Artemisia tridentate* Nutt.), low sagebrush (*Artemisia arbuscula* Nutt.), and antelope bitterbrush (*Purshia tridentate* Curran). Shrubs are the most prevalent flora followed by various grasses, forbs, rushes and sedges associated with the ecotype. In the higher elevations, particularly on ridges and slopes, curl-leaf mountain mahogany (*Cercocarpus ledifolius* S. Watson), western juniper (*Juniperus occidentalis* Hook.) and pockets of aspen (*Populus tremuloides* Michx.) can be found (Rodgers and Tiehm 1979, Collins 2016).

Springs dot the landscape, creating small islands of succulent riparian vegetation dominated by rushes, sedges and other wetland obligate species. However, with the advent of groundwater developments, many of those systems have begun transitioning toward more drought-tolerant and invasive species (Chambers and Miller 2004, Perkins et al. 1984). These springs are formed by snowmelt that seeps into the groundwater and emerges again after encountering an impermeable rock layer, similar to other contact springs (Sada et al. 2001). The refuge receives limited precipitation, averaging only 30 cm a year (Collins 2016). The western half up on the plateau is wetter (30 cm) than the east, which only receives approximately 20 cm annually (Hazeltine 1959).

Site Selection

We selected our twenty-four study sites from a list of 130 springs identified by the Fish & Wildlife Service, most of which were located in the western portion of SNWR up on the plateau (USFWS 2012). We restricted our selections to springs located within this western portion of SNWR to ensure that all of our sites would share similar elevations, vegetation types and lithology. Therefore, springs in the northeastern corner, which is 900 m lower in elevation and a salt-desert were discarded. Type of water development and accessibility to roads were also factored into our selections. Springs in SNWR had different water developments (dugouts, berms, reservoirs, springboxes, etc.) and we chose to concentrate on sites with springboxes (Figure 2.1). All the springboxes at our sites were constructed from a corrugated metal culvert roughly one meter in diameter and two meters deep. Each springbox had two outflow pipes: a main pipe siphoned water from the springbox and channeled it to a trough (which may or may not have been removed at the time of this study)and second pipe, which allowed overflow into the riparian corridor, was located at the same depth as the outflow pipe (1.5 meters below ground level). Troughs were located anywhere from 50 to 800 meters away from the riparian corridor depending on the size of the wetted area and the surrounding topography.

Study Design

After being divided into four groups, each of our twenty-four sites (Figure 2.3) were randomly assigned one of six different treatments: control, capped pipes, sand-filled, sand-filled with springbox casing removed, gravel-filled, gravel-filled with springbox casing removed. The four groups were designated based on the amount of flow produced by the site during pre-treatment: high (0.030-0.005 ft3/sec), medium-high (0.004-0.0017 ft3/sec), medium-low (0.0016-trace ft3/sec) and low (trace-0 ft3/sec).

Treatment Type Descriptions

We designed six different treatment types (Figure 2.4) with varying degrees of cost and effort needed for implementation. We wanted to know the least degree of restoration needed to achieve an acceptable level of rehabilitation for lentic springs.

1) Control: Springboxes were left completely unaltered. The outflow pipes were left open. This was to account for hydrologic variation due to climatic difference between pre-treatment and post-treatment periods.

2) Capped Pipes: Springbox casing was left intact and both outflowing pipes were capped. Leftover underground piping was removed and the ground compacted to remove air pockets. Holes were drilled in into the springbox lid to allow water to flow freely from the box. If the metal of the springbox rose above ground level, holes were also drilled around the circumference of the metal casing at evenly spaced intervals. These holes were made small to prevent rodents from climbing into the springbox and drowning (Andrew et al. 2001). This treatment was the most cost-effective, and easiest to implement. However, there is concern that the hydrologic pressure, which is greater within a solid column of water than mixed in with a substrate, could cause the spring source to collapse.

3, 5) Sand or gravel-filled: Springbox was filled with sand or gravel. Particle sizes varied and materials were sourced from the local quarry. Before filling, both outflow pipes were capped to prevent water escaping. The pipes connected to the springbox were dug out and removed. The soil was compacted to collapse any air pockets left by removing the piping. This was to prevent water from flowing down those pockets causing erosion. We hoped to encourage the water to mimic its pre-disturbance movements (Sada 2008). Any part of the metal casing that rose above the ground was also removed. Leaving any structures above the ground could impede the natural movement of the water once it reached the surface (Barquin and Scarsbrook 2008). The natural scaffolding used by the water to climb to the surface was disturbed by installation of the springbox. This treatment attempted to recreate some semblance of that scaffolding and encourage vertical movement by the water. To prevent further lateral movement, the metal casing of the springbox was left in place. Overtime that casing will degrade and allow horizontal flow. However, by that time, the natural hydrology will have been reestablished.

4, 6) Sand or gravel-filled with springbox casing removed (hereafter sand SBR and gravel SBR): This treatment was implemented in the same way as the sand or gravel-filled (see above). The difference being that the metal casing was removed along with all underground piping. Locally sourced gravel or sand was used to fill the leftover pit and then leveled to match the surrounding terrain. The treatment removes all man-made structures to allow unimpeded movement of water in all directions. With no structure to encourage vertical movement, water may be drawn down into subsurface flows eliminating all availability for use by plants and wildlife.

Biological Measurements

Foliar Cover

Vegetative cover measurements were recorded during the growing season in June 2016 and 2017 using a nested frequency quadrat frame. Pre-treatment and post-treatment measurements occurred at about the same time of the year. This measurement was intended to account for shrub presence in the riparian zone. Using the center of the springbox as a pivot point, we randomly selected a single degree value ranging between 0 and 360. Three additional points were chosen at 90 degree intervals to each subsequent transect until four perpendicular transects were chosen. Transects were laid out emanating from those degree points, lengths determined by the distance to the edge of the riparian zone. Riparian zones are the transition between water and land, characterized by wetter soils and water-dependent plant communities (Lewis et. al 2003).

We used those characteristics to determine the extent of our riparian transects. We considered the edge of the riparian-wetland area to be where soils had noticeably dried on the surface and where plant communities shifted from the dominance of wetland-obligate (Carex spp.) and even wetland-facultative species (*P. pratensis* and *J. balticus*) into clearly upland dominated communities (invasive grasses and shrubs). Transitions from vegetated ground cover to bare interspaces between shrubs were also used as an indicator of riparian extent. Once the exact distance of the riparian transect was determined, we added five additional meters to the end of the riparian transect that extended into the upland vegetation.

We chose up to five sampling units for each transect. If the riparian transect did not extend to five meters, measurements were taken at each meter point. Five measurements were always taken along the upland transect. If the riparian zone extended beyond twenty meters, the upland measurements were discarded.

This study focused on the direct influence of the springbox in question. To record cover, we used a nested quadrat frame (Smith et. al 1987). We used point method using 8 points on the frame. Bare ground was recorded if no plant material was detected and mineral soil was contacted. A summed cover value for each individual species was created by averaging observed occurrences over the entire transect. We also created a summed value for total foliar cover by averaging the cover values of all living matter across the transect. For analysis, we combined frequency measurements into functional plant community groups: grasses, grass-likes, forbs and shrubs. We also analyzed changed in frequency for the five species with the greatest overall frequency in all our sites.

Frequency

Frequency measurements were recorded from nested frequency frames (Smith et al. 1987) once in June during the pre-treatment period, and then at the same time post-treatment. We used a nested quadrat frame to indicate changes in the species abundance and distribution of plant communities between pre- and post- treatments and site-to-site (Despain et al. 1991). Theoretically, species recorded in the smallest square should have the greatest frequency in the population, with species found in larger squares proportionately less abundant in the overall population of the area (Greig-Smith 1983). When the species *J. balticus* and *P. pratensis* were located in the smallest square, the numbers of shoots were also recorded due to the rhizomatous nature of these species. The frequency of occurrence of individual species were recorded if at least fifty percent of the plant base fell in the quadrate square (Herrick et al. 2005, Smith 1987). Frequencies of individual species observed at each quadrat were averaged to create a summed frequency value for the entire transect (Smith et al. 1987). We combined frequency measurements into functional groups (grasses, grasslikes, forbs and shrubs) and selected the five most frequent species (*Poa pratensis*, *Poa secunda*, *Bromus tectorum*, *Juncus balticus* and *Carex nebrascensis*) for analysis, similar to cover measurements.

Biomass

Biomass was collected along the same transects used for frequency and cover. We measured biomass within the riparian corridor using three randomized points along the transect. At each of those points, a square (give dimensions) of plant matter was collected and transferred to paper bags. Initial weights for each bag were taken immediately after collection and recorded, providing a “wet weight”. In the lab, samples were placed in a drying oven at 65°F for forty-eight hours (Gross and Soule 1981). After samples dried, the entire sample was weighed again to get a “dry weight”. The dry weight was the total biomass without water. After collecting the total biomass, each sample was separated out into functional groups: forbs, grasses, grass-likes and shrubs. Each group was also weighed and their biomass recorded.

Wildlife Observations

Reconyx PC900 Hyperfire Professional Covert Camera Traps© were set up at each spring site. The camera was positioned behind the springbox and pointed down the riparian corridor to provide the widest angle encompassing the area where we expected wildlife use to be the most probable. Cameras were mounted on a simple T-bar approximately 1.5 m above the soil surface. Each camera was pre-programmed to take a set of three photos (one every five seconds) when motion was detected, then reset with a fifteen seconds delay before additional motion was detected. This pattern would continue until motion could no longer be detected. The motion sensor was sensitive up to twenty meters and beyond, with twenty meters being within the scope of this study. Photograph data was stored and used for analysis within a database management system (Microsoft Access©). Species and number of individuals in each picture were recorded.

Statistical Analysis

We organized and analyzed data for this study using the same procedures and methods for each variable. Data was recorded at each site. Every site was grouped into the following categories: spring type, treatment type and flow group. An example of this was Beebee spring, which was labeled: lowest flow group, capped pipes treatment and single spring type. Variables were recorded along four transects per site, and then averaged across the entire site. Multiple measurements of a variable collected at a site were then averaged over the entire field season to produce a single value per site per year. Pre-treatment variables were compared to post-treatment variables using Analysis of Variance (ANOVA) with the pre-treatment being used as the covariate. We selected a significance cutoff of $p<0.1$ due the diverse and individualistic nature of the springs used in the study. The dependent variable used in ANOVA was the change between years (pre-treatment vs. post-treatment). Our analysis aimed to identify differences between years for each category listed above.

We analyzed the individual elements of each category using a test of least square means with pre-treatment values acting as the dependent variable. We also attempted to account for yearly effects, particularly climatic variations, by testing whether the change in our treatments was significantly different from the change in the control sites. This was accomplished using a difference of least square means test. We adjusted for multiple comparisons by using a Tukey-Kramer adjustment. All analyses were conducted using SAS©.

An overall f-test analyzed the difference between pre- and post-treatment values at individual springs sites. We also conducted a least square means test to determine whether yearly measurements significantly differed from zero. A p-value of 0.1 was also used as the significance cutoff for this dataset for the same reasoning as above.

RESULTS

Cover

Between pre-treatment and post-treatment, we did not record an increase in the total vegetative cover of living plant matter; however, individual functional groups did see varying amounts of increasing and decreasing cover values (Table 2.2). Grass cover increased at spring complexes (p=0.0852) and in the low (p=0.0783) and medium-high (p=0.0672) flow groups. We observed increased grass cover with capped pipes treatments and the control group (p=0.0325 and p=0.0260) while no differences were observed with forb cover. Vegetative cover of sedges and rushes decreased by 11.26% at gravel SBR treatment sites (p=0.0149). Yearly effects show that the treatments did not differ from the controls (Table 2.2).

Frequency

For the functional groups, we saw significant change in grasses between pre- and post-treatments in both year and treatment type. For individual categories, there were no changes, except for gravel SBR sites (p=0.0106) and medium-low group (p=0.0870). These sites experienced a decrease in the frequency of grass species. However, there was no difference between this treatment type and the control. The frequency of grass-like species did not change. There was a difference in shrub frequency between years (p=0.0058); however, our analysis could not pinpoint the cause for these differences. The most dramatic change occurred in forbs which experienced a difference between pre-treatment and post-treatment (p=0.0032), and an interaction between spring type and flow group (p=0.0574). Nearly every category recorded an increase in the frequency of forbs. Every spring type, flow group and four out of six treatment experienced an increase in forb frequency (Table 2.1). Sand-filled and gravel-filled, the two treatment types did increase in forb frequency, but had p-values that were just outside of the cutoff for significance (p=0.1032 and p=0.1415, respectively). Treatment effects exhibited multiple changes, yearly effects did not show a difference between controls and treatments.

Biomass

Based on effects alone, we observed no difference in biomass between pre-treatment and post-treatment (Figure 2.5). However, we did encounter increased initial wet weight in spring complexes (p=0.0060), and all flow groups (high p=0.0029, medium-high p=0.0124, medium-low p=0.0966), except for the lowest flows. Wet weight biomass increased in the control group (p=0.0793), capped pipes (p=0.0734), gravel-filled (p=0.0779) and sand-filled (p=0.0004). It is notable that all the treatments that showed increases in wet biomass retained the metal casing of the springbox around the spring source (Table 2.4). The pattern remains for total dry weight biomass, except for capped pipes which no longer displayed differences between pre- and post-treatments. When the amount of change for the treatments was compared against the change in the controls, our analysis shows that they were not different. There was no change in the biomass of sedges and rushes. The biomass of forbs increased at springs with a medium-low flow rate ($p=0.0838$) and a sand-filled treatment ($p=0.0399$). Grass biomass increased at sites that had only one producing spring ($p=0.0359$), and across all flow groups (high $p=0.0188$, medium-high $p=0.0295$, low $p=0.0648$), except for medium-low flowing springs. The only effective treatment was the control ($p=0.0144$) and the sand-filled ($p=0.0263$) for increasing grass biomass. Sand-filled springboxes also showed increases in forb biomass post-treatment (Figure 2.5). Yearly effects indicate that changes in biomass at treatment sites did not differ significantly from changes in the biomass at the controls (Table 2.4).

Wildlife Observations

Our analysis detected a yearly effect between pre- and post-treatment periods for both species number ($p=0.0001$) and the quantity of visits ($p<0.0001$) to the springs from wildlife. Both variables decreased at all spring types and flow groups, except for the highest flowing, and most treatments. The number of observed species decreased at our control, capped pipes, sand-filled and gravel-filled sites (Table 2.5). Wildlife visits decreased at the same treatments as number of observed species with the addition of gravel SBR treatment (Table 2.6). At individual sites, we saw a decrease in the number of species appearing at seven of our spring sites. The quantity of animals frequenting our sites did not change, except for at Meadowlark. Meadowlark experienced an increase in wildlife visits from 26 visits in pre-treatment to 286 in post-treatment ($p=0.0753$). Sand SBR treatments decreased in the number of wildlife visits in comparison to the controls, independent of yearly/climatic variation. The other treatments did not differ from the control (Table 2.6).

DISCUSSION

At the beginning of our study, we identified three signs of restoration. First, we would expect a shift in plant community from facultative wetland and upland species to more obligate wetland species, such as rushes and sedges (Barquin and Scarsbrook 2008, Wyman et al. 2006). Second, we expected to see increased biomass (Martin and Chambers 2001, Wyman et al. 2006). Last, a restored functioning spring and wet meadow would attract increased usage by wildlife and more diversity of species (Stevens et al. 1977).

We did see a shift in the plant community, but not exactly in the direction we expected it would go. We hypothesized that restoring the natural hydrology of spring and wet meadow systems would shift back toward plant communities with higher concentration of sedges and rushes. Instead, we observed that grass cover increased at sites that were part of spring complexes and at control sites and where we had capped outflow pipes for treatment. Our indicator functional group, sedges and rushes, did not significantly increase in any of our categories, and in fact decreased at gravel SBR sites. While the percent cover of forb species did not change, the frequency of forbs increased over all categories. Four of our six treatments were significantly increased, and the two that were not had p-values near 0.1. One possible explanation for these changes is the disturbance caused by implementing our treatments. All treatments involving sand and gravel required heavy machinery to transport and deposit materials, and remove metal springbox casings. Shrubs, which are indicators of water table depth in riparian zones (Hammersmark et al. 2009, Castelli et al. 2000), did show that post-treatment was significantly different from pre-treatment. Unfortunately, our analysis was unable to determine the specific cause for this change. At one of our sites, Little Fish, there was extensive sagebrush encroachment in the riparian corridor and directly next to the spring source (Figure 2.6). After treatment, large decades old sagebrush growing adjacent to the springbox died, likely due to intolerance to saturated soil conditions (Castelli 2000, Hammersmark 2009). Smaller sagebrush in the newly wetted riparian zone had also died, a scenario which repeated itself across several of our other sites.

Our second indicator, biomass, did show some significant changes. Our initial wet weights increased across both springs types. Of our flow groups, only the lowest flows did not show significant increases in wet weight. It is possible that these sites simply lack sufficient initial discharge to sustain fully functional plant development and biomass (Martin and Chambers 2001). This may change as more establishment occurs due to increased soil moisture (see chapter 1). Additionally, four of our six treatments experienced increased wet weights. These four treatment all had the metal casing of the springbox left in the ground. This metal casing may have prevented horizontal movement of water in the soil profile, mimicking the natural hydrology (Sada 2008) which funnels water to the surface where it is more readily accessed by plant roots. Increased access allows them to incorporate more water into their tissues, hence the higher initial wet weights (Wyman et al. 2006). Our dry weights, which show the amount of tissue a plant can produce, follow this same pattern of increases from post-treatment. The only exception was that the changes in the dry weight biomass for capped pipes treatment were no longer significant. The biomass of forbs increased at medium-low flowing sites with a sand-filled springbox. Sand-filled sites also had increased grass biomass as well, but in every other flow group except for medium-low flowing springs. Wetland-obligate sedges and rushes did not have increased or decreased biomass.

Wildlife usage marked our final restoration indicator. We hypothesized that more water would mean more forage for wildlife (Wyman et al. 2006, Naiman and Decamps 1997) and that this would result in increased usage by wildlife, increasing total species diversity at that site (Stevens et al. 1997). However, we only recorded decreases in wildlife usage and species diversity, despite an apparent increase in biomass. This was especially apparent in regards to the number of species visiting our spring sites, particularly those with treatments that left the metal casing of the springbox intact. A difference of one of two species may not seem important from a management perspective. However, when that one or two consists of threatened or endangered species then the change carries more weight. For example, greater sage grouse hens have been observed using some of our sites. Sage grouse hens use these wet meadows in the late summer to a brooding ground their chicks making these spring systems important habitats for recruitment of a threatened species (Wallestad 191, Connelly et al. 1988, Savage 1969).

The disturbance to the area caused by implementing our treatments may have encouraged warier species to seek out other wet meadows. Therefore, as the area more fully recovers, species diversity will increase but that determination would require additional research and is outside the scope of this study. The highest flow groups did not see a decrease in species diversity or number of visits. Ample water and an already robust plant community may be less disrupted by restoration efforts (Carothers 1997); therefore, not affecting the established foraging behaviors of dependent wildlife.

One of our sites, Meadowlark, saw a significant increase in the number of wildlife visits in post-treatment. Located in the southeastern corner of SNWR, Meadowlark was the most distance to other springs. In pre-treatment, Meadowlark produced little to no water. It is clear that wildlife aware of the water availability, such as images of coyotes (*Canis latrans*) sniffing at the springbox's metal casing as if they could smell the water but not access it. Post-treatment in post-treatment, Meadowlark experienced a dramatic increase in water flow, producing one, if not the only, sizeable water source in this corner of the refuge. This increased flow and water availability may account for the increased wildlife usage at this site (Sada 2008, Wymen et al. 2006). It was particularly important in late summer as other spring runoff sources dried up.

Our findings seem contradictory to other studies on wildlife use of restored sites. Brawley et al. (1998) found that both abundance and species richness of wetland birds increased at restored marshes in the Barn Island Wildlife Management Area. Wildlife use of restored drained agricultural land enrolled in the USDA's Wetlands Reserve Program was greater than expected (Rewa 2005). Many of these studies compare restored to unrestored sites in the same physical space. Our study investigates the relationship of the same site restored and unrestored, a temporal shift and in a semi-arid environment.

It may take time for the effects of hydrologic restoration on biologic aspects of springs and wet meadows to be fully understood. Wet meadows are highly dynamic and can change quickly with small change to the underlying hydrology of springs (Gray et al. 1992, Lewis et al. 2003). The native plant communities of these areas are hydrophilic sedges and rushes that must be in contact with the water table for persistence (Lewis et al. 2003). Dropping water tables and over-grazing can rapidly remove these species from the system, allowing them to be replaced by an altered plant community (Fleischner 1994, Chambers and Miller 2004, Prichard et al. 1994, Perkins et al. 1984). Without sedges and rushes to trap sediments in place (Micheli and Kirchner, Winward 2000, Manning et al. 1989), increased erosion will lead to a cycle of self-perpetuating degradation.

Even with the disturbance removed, the established alternate plant community may be difficult for wetland-obligates to outcompete (Stringham et al 2003, Wyman et al. 2006). Proper hydrologic conditions must be reinstated before wetland obligates can recolonize (Schumm 1977, Jensen et al. 1989) and it may take years for soils to rehydrate and channels to fill in with sediments (Cowley 1995, Wyman et al. 2006). Many water developments in SNWR were constructed in the 1960s and earlier (Hazeltine 1959, USFWS 2012). Seedbanks for sedges and rushes may have become depleted in that time and other species will establish on a first come, first serve basis. If sedges and rushes are already present at restoration, their return may happen relatively quickly (Wyman et al. 2006). If not, intervention with planting and seeding may be needed to encourage them to recolonize. Even then, studies have shown that it can take five to ten years for riparian plant communities to fully recover a site (Cowley 1995).

Similar to hydrologic variables (see chapter 1), changes in biomass, cover, frequency and wildlife usage reflect our treatment effects. We also accounted for yearly variations by comparing the amount of change in each treatment to the amount of change in the control. When we consider these yearly/climatic effects, the treatment effects become insignificant. This variation in yearly effect is in part due to increased precipitation between the pre- and post-treatment periods. According to NRCS Soil Climate Analysis Data (SCAN), the Sheldon National Wildlife Refuge had similar precipitation amounts for 2016 and 2017 (25.1 cm and 24.94 cm, respectively) during the water year (USDA 2017). However, spring precipitation increased 26% in May 2017 in comparison to May 2016. With more water available during the growing season, biomass and other plant community measurements could be impacted, particularly since 2016 came on the tail end of a stretch of dry years. This increase in precipitation appears to negate the significant effects of the treatments. However, we believe that precipitation cannot completely account for all the change seen at our sites. As seen in Chapter 1, individual sites changed too dramatically to be solely attributed to climatic variation. This incongruity emphasizes the need for more data to confidently draw out the treatment effect.

Management Implications

For land managers it is important to understand that plant communities in restored springs and seeps will not quickly shift from degraded seral stages to fully functional obligate wetland species (Stringham et al. 2003). It can take upwards of a decade for herbaceous and woody species to recover (Cowley 1997). We saw many species that indicate riparian degradation, such as *A. tridentata* (USFWS 2012), at our sites signifying that they have been extensively degraded for an extended period. Some weed control and seeding may be needed to encourage the recruitment of obligate wetland species (Stringham et al. 2003) as they may have been removed from the community for such a long time that there is no longer a viable seedbank to facilitate their return (Sada et al. 2001). The increase in biomass of what is present in the community leads us to believe that once established obligate wetland species have a good chance of flourishing in these restored systems (Barquin and Scarsbrook 2008, Sada 2008). For those mangers concerned about wildlife, it is important to note that restoration does cause wildlife to avoid these springs. If there are other undisturbed springs in the area, wildlife usage may just shift to those sites. Restoring springs one at a time will allow wildlife to have access to familiar unsullied water sources as the restored sites recover. This is especially important for sensitive and wary species, such as Greater sage grouse, that may actively avoid restored sites. Restoring high flowing springs first, which did not decrease in wildlife visits or species diversity, provides a robust site for wildlife to depend on while less productive sites recover.

Conclusions

The biologic response to restoration of lentic springs and seeps does not occur as quickly as hydrologic restoration (Knighton et al. (Unpublished results)). One of our indicators, biomass, showed that restoration had occurred. However, we did not see the shift in plant communities toward obligate wetland species, or an increase in wildlife usage. This does not necessarily mean that our restoration attempts failed. The biological aspects of these wet meadows may simply take longer to show signs of recovery (Cowley 1997). The transition to a degraded state took decades to occur, and it may take a similar amount of time to shift back to something resembling its historic condition (Stringham et al. 2003). It is important to understand that these sites may have crossed an ecological threshold. They may never return to what they were, but may in time settle on a new state. Hopefully, this new state fulfills all the roles of a properly functioning Great Basin wet meadow (Lewis et al. 2003, Wyman et al. 2006, Buckhouse and Elmore 1993, Cowley 1997). It is important to recognize the importance of precipitation in the restoration of biological components of springs and wet meadows in the Great Basin. Increased precipitation naturally provides more water for plant and wildlife use. The apparent conflict between treatment effects and yearly/climatic effects simply highlights the need for further data collection at these sites, preferably with multiple years of varying precipitation. With this information, the full impact of restoration on springs will be more clearly understood.

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FIGURES

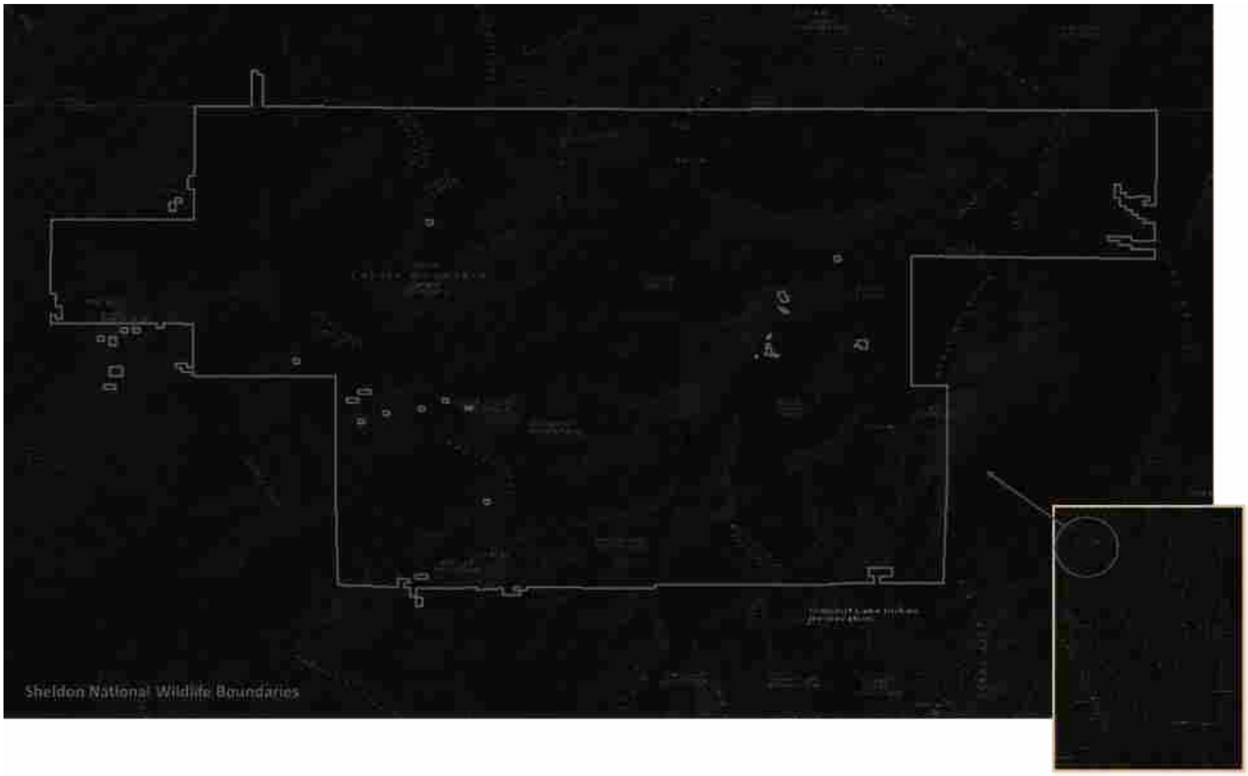


Figure 2.1. Sheldon National Wildlife Refuge is located in the northwestern corner of Nevada, USA. It was established in the 1930s as a refuge for the then endangered pronghorn (*Antilocarpa americana*). Today, SNWR is 573,504 acres of uninterrupted sagebrush-steppe ecosystem.

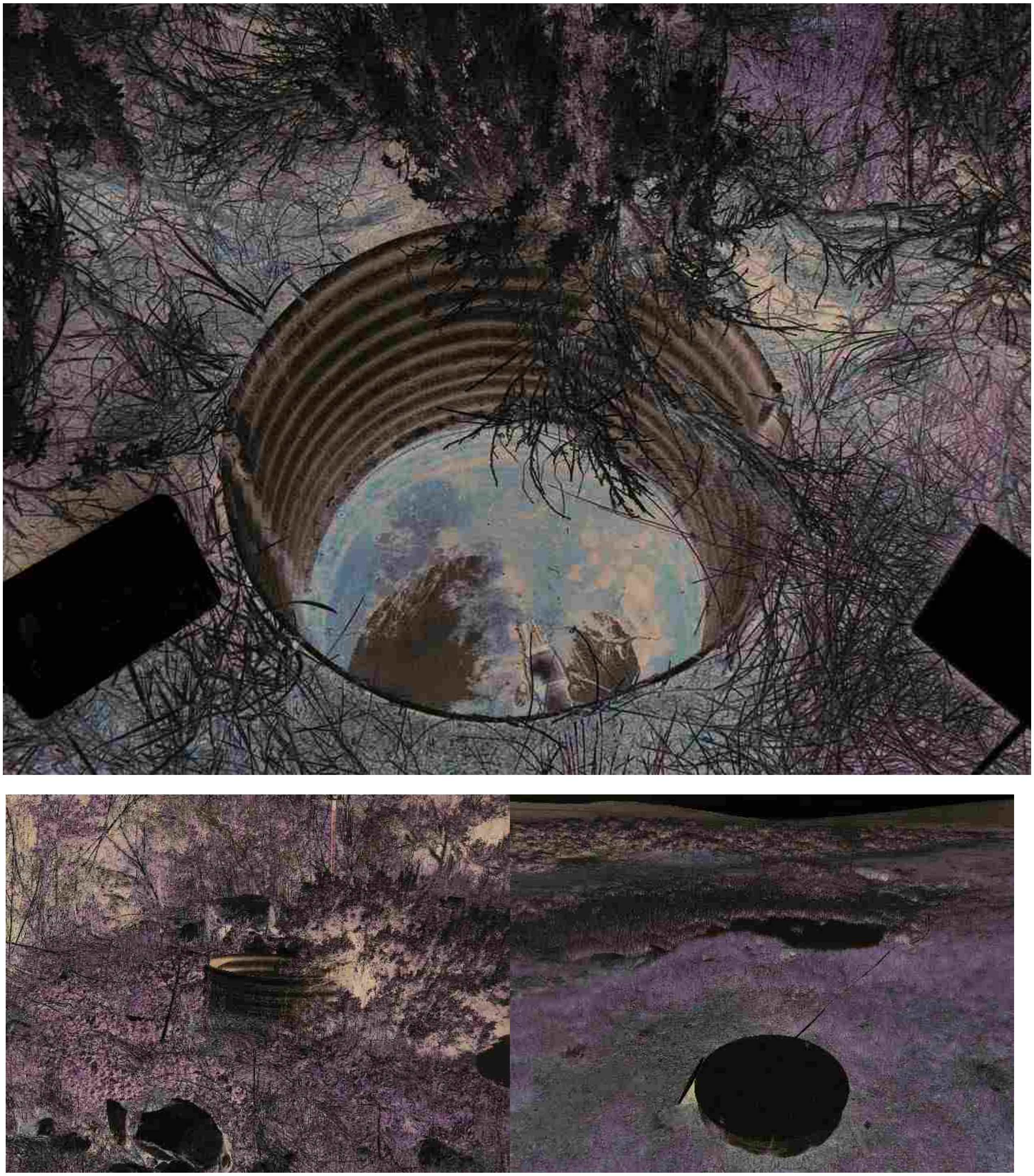


Figure 2.2. Springboxes, and other water developments, were installed on springs in the Great Basin by early settlers and land managers as water sources for livestock and wildlife. The unintended consequences of these structures was the dewatering of springs and wet meadows.

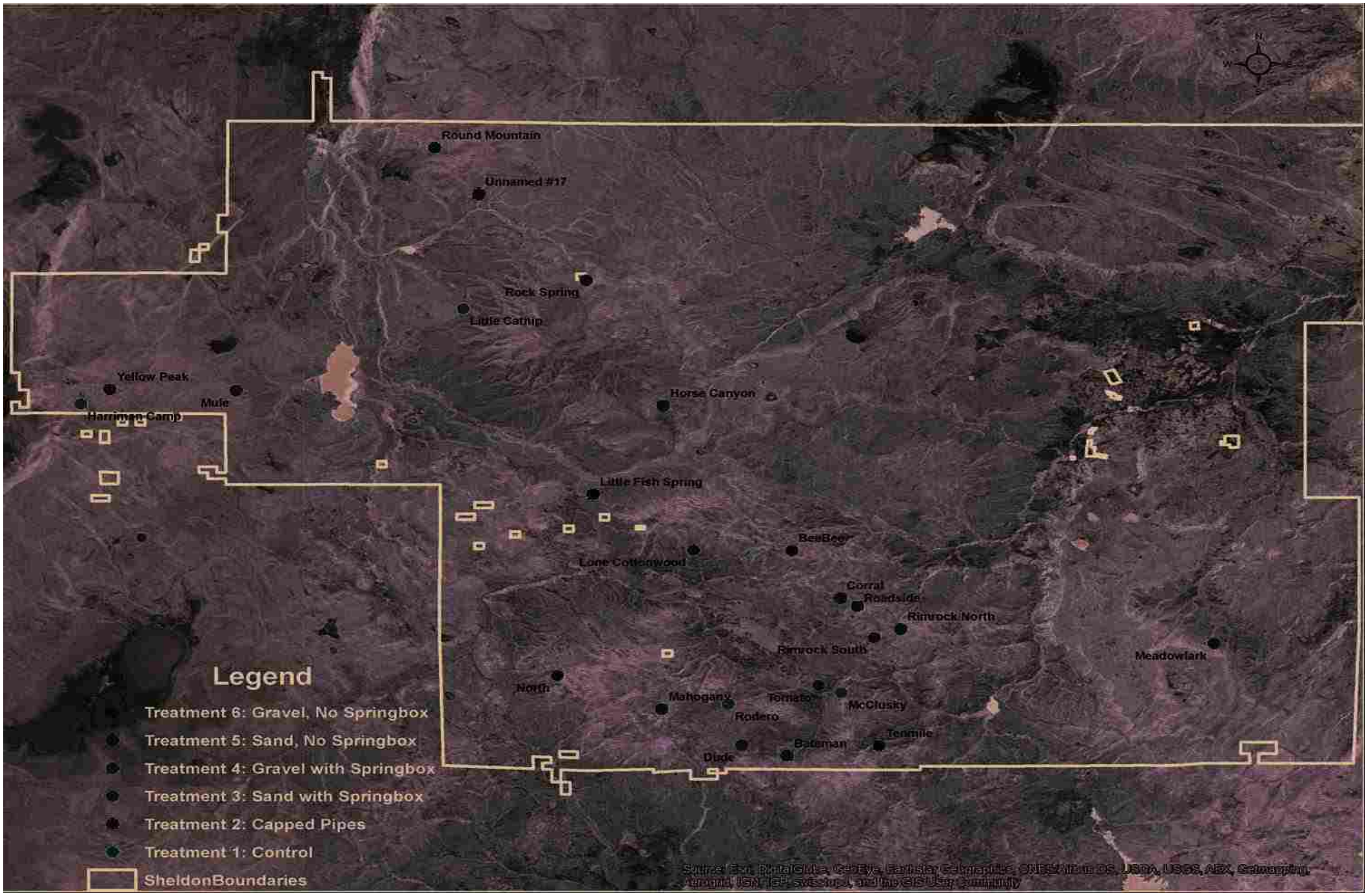
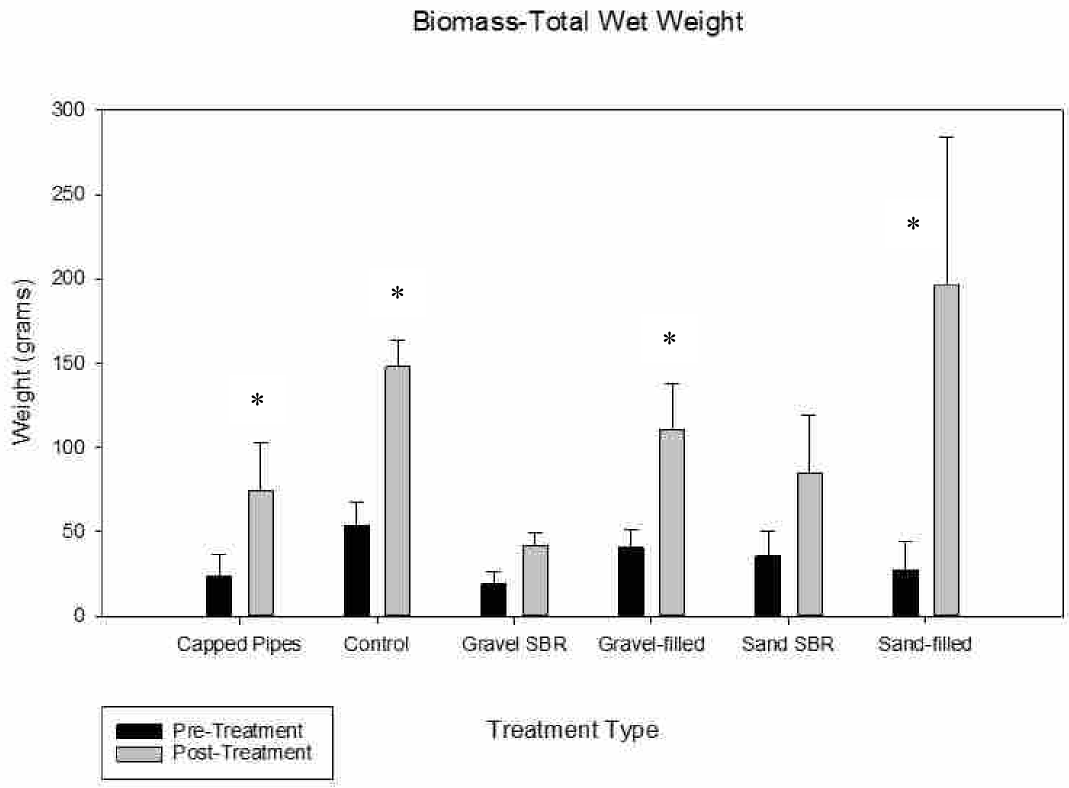
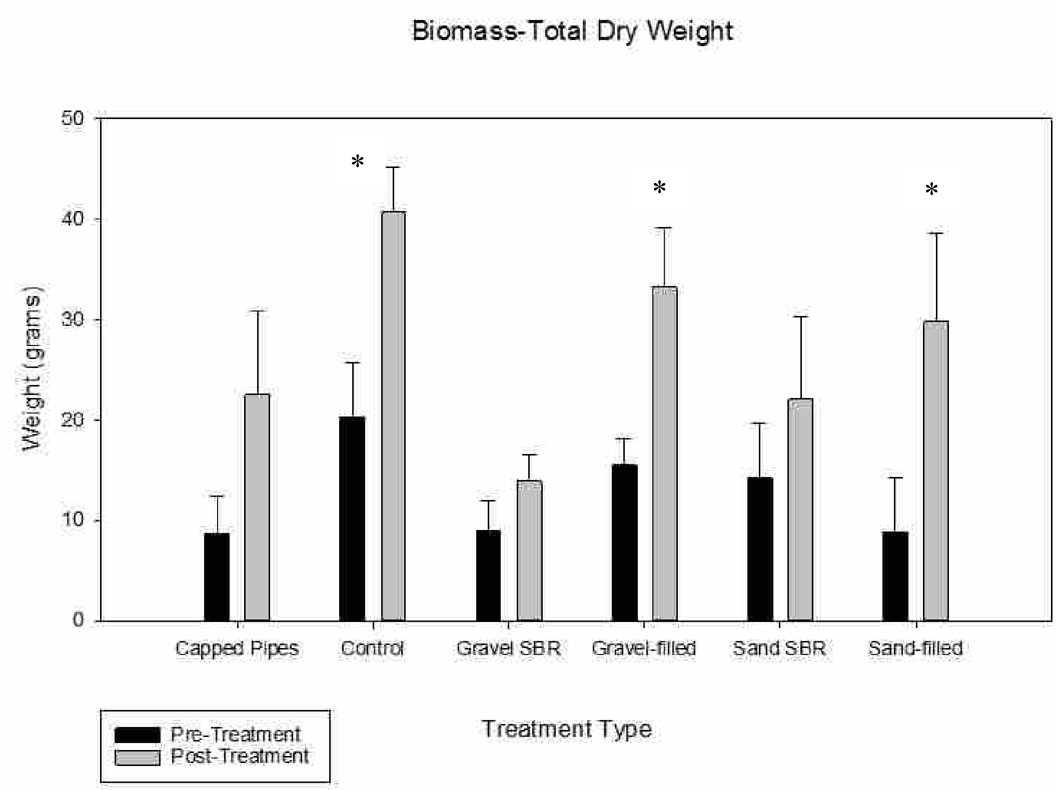
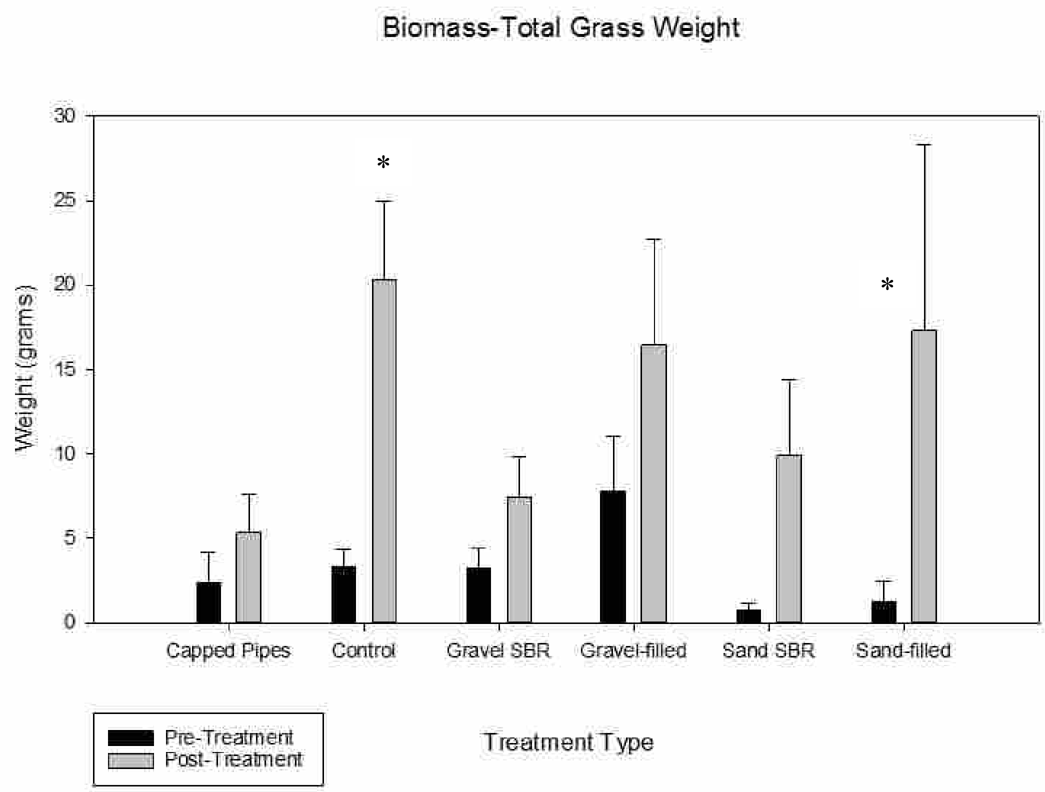


Figure 2.3. Randomly assigned spring sites in the Sheldon National Wildlife Refuge. Assigned treatments and locations are illustrated by different colored marks. Spring site were selected based on similar elevations, accessibility, plant community and water development.

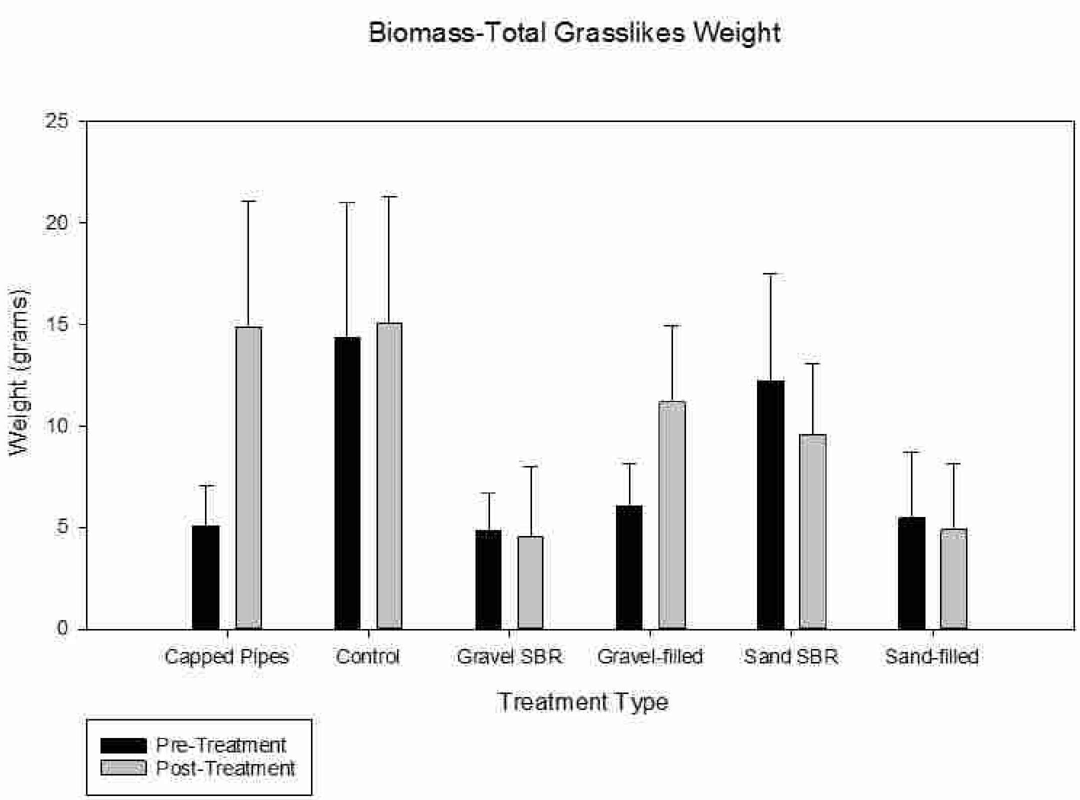
Figure 2.4. Spring-restoration treatment designs were created with cost, practicality and functionality in mind. Materials for treatments #3-6 were sourced from local sources. These treatments were designed to mimic the historic structure's functionality.


Biomass-Total Wet Weight
Weight (grams)
300
250
200
150
100
50
0
Capped Pipes
Control
Gravel SBR
Gravel-filled
Sand SBR
Sand-filled
Pre-Treatment
Post-Treatment
Treatment Type


Biomass-Total Dry Weight
Weight (grams)
50
40
30
20
10
0
Capped Pipes
Control
Gravel SBR
Gravel-filled
Sand SBR
Sand-filled
Pre-Treatment
Post-Treatment
Treatment Type



Biomass-Total Grass Weight
Weight (grams)
30
25
20
15
10
5
0
*
*
Capped Pipes
Control
Gravel SBR
Gravel-filled
Sand SBR
Sand-filled
Treatment Type
Pre-Treatment
Post-Treatment



Biomass-Total Grasslikes Weight
Weight (grams)
25
20
15
10
5
0
Capped Pipes
Control
Gravel SBR
Gravel-filled
Sand SBR
Sand-filled
Treatment Type
Pre-Treatment
Post-Treatment

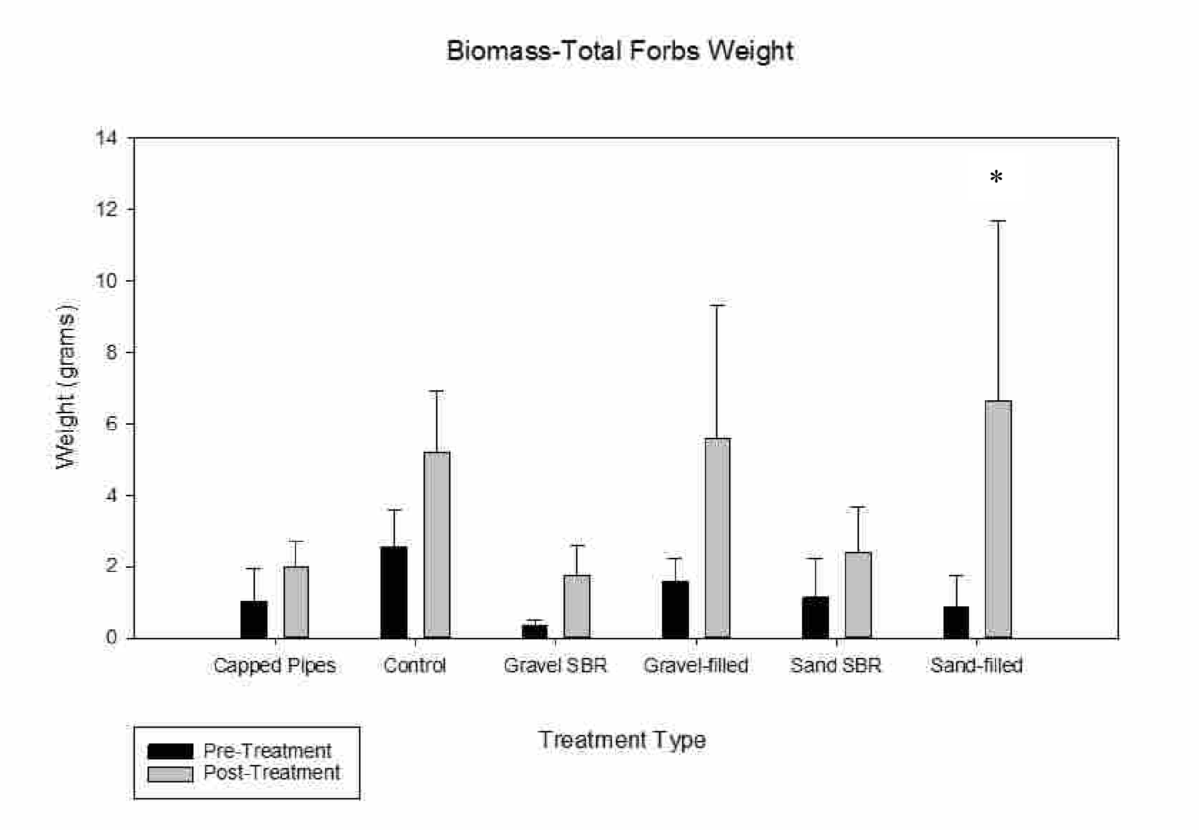
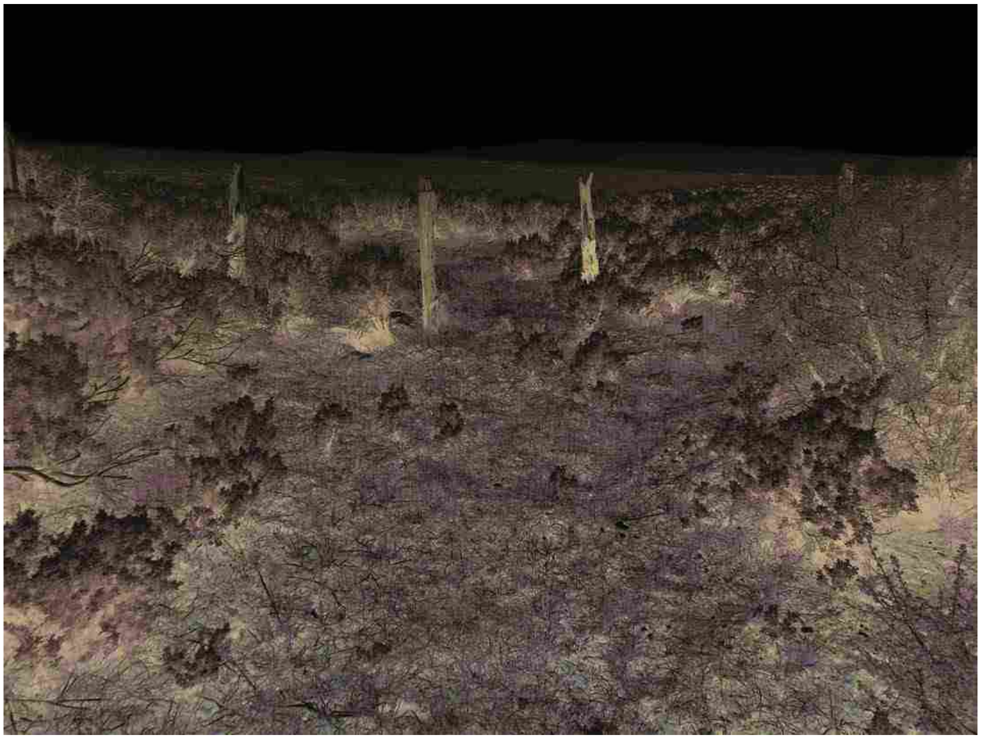


Figure 2.5. Changes in biomass for different functional groups and total overall biomass (wet and dry). Shrubs were not included, as they did not appear in riparian areas enough to be measureable. Significant changes are denoted by a (*).



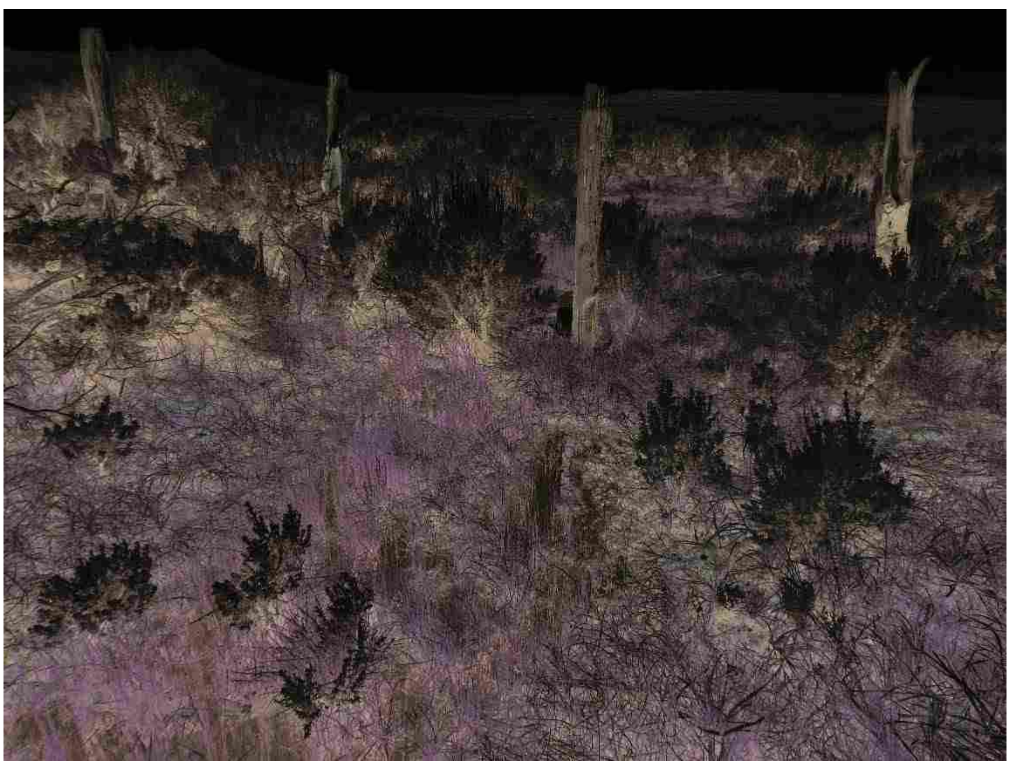


Figure 2.6. Little Fish spring pre- (top) and post-treatment (bottom). The small sagebrush in the center of the wet meadow in pre-treatment period was an indication of water table depletion. Those same sagebrush in post-treatment have drowned and standing water is in the thalweg.

TABLES

Table 2.1. Frequency of functional groups (grasses, grasslikes, shrubs and forbs) at our spring sites representing treatment and yearly/climatic effects. Nearly every sub-category of the variable experienced increases in forb frequency based on treatment effect. The two treatments that did not have significant increases had p-values that were just outside the p=0.1 cutoff. However, yearly effect showed no deviation from the control, suggesting that increases in forb frequency were driven by higher precipitation post-treatment. Significance denoted by (*).

Total Grass Frequency

Category		Treatment Effects		
		Estimate	*SE*	*P-value*
Spring Type	Complex	-0.0379	0.04046	0.3664
	Single	-0.0492	0.03094	0.1355
Flow Group	High	-0.0658	0.03794	0.1065
	Low	-0.0064	0.03157	0.8431
	Med High	-0.0408	0.03317	0.2411
	Med Low	-0.0613	0.03311	0.087*
Treatment Type	Capped Pipes	-0.0551	0.04942	0.2847
	Control	-0.0565	0.0415	0.1968
	Gravel SBR	-0.1460	0.04892	0.0106*
	Gravel-filled	0.0215	0.03979	0.5985
	Sand SBR	-0.0526	0.05011	0.3128
	Sand-filled	0.0274	0.04175	0.5237

Yearly/Climatic Effects	*Estimate*	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**0.0013**	**0.0714**	**0.9855**	**1**
Control vs. Gravel SBR	**0.0895**	**0.07088**	**0.2289**	**0.7992**
Control vs. Gravel-filled	**-0.0779**	**0.05792**	**0.2015**	**0.7563**
Control vs. Sand SBR	**-0.0038**	**0.05727**	**0.9475**	**1**
Control vs. Sand-filled	**-0.0838**	**0.06277**	**0.2047**	**0.7617**
Capped vs. Gravel SBR	0.0908	0.05483	0.1216	0.5797
Capped vs. Gravel-filled	-0.0766	0.06231	0.2406	0.8154
Capped vs. Sand SBR	-0.0025	0.08313	0.9763	1
Capped vs. Sand-filled	-0.0825	0.05667	0.1692	0.6955
Gravel SBR vs. Gravel-filled	-0.1674	0.06265	0.0192	0.1476
Gravel SBR vs. Sand SBR	-0.0933	0.08219	0.2766	0.8581
Gravel SBR vs. Sand-filled	-0.1733	0.05669	0.0092	0.0781*
Gravel-filled vs. Sand SBR	0.0741	0.06564	0.2794	0.861
Gravel-filled vs. Sand-filled	-0.0059	0.05679	0.919	1
Sand SBR vs. Sand-filled	-0.0800	0.07257	0.2904	0.8721

Total Grasslikes Frequency

Category		Treatment Effects		
		Estimate	*SE*	*P-value*
Spring Type	Complex	-0.0274	0.04094	0.5156
	Single	-0.0513	0.0315	0.1273
Flow Group	High	-0.0652	0.04041	0.1308
	Low	-0.0307	0.03618	0.4116
	Med High	-0.0374	0.03724	0.3339
	Med Low	-0.0241	0.03635	0.5192
Treatment Type	Capped Pipes	-0.0306	0.05196	0.5664
	Control	-0.0254	0.04417	0.5754
	Gravel SBR	-0.0333	0.05224	0.5353
	Gravel-filled	-0.0183	0.04236	0.6725
	Sand SBR	-0.0560	0.05202	0.3015
	Sand-filled	-0.0725	0.04398	0.1233

Yearly/Climatic Effects	*Estimate*	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**-0.0052**	**0.07505**	**0.9459**	**1**
Control vs. Gravel SBR	**0.0079**	**0.07577**	**0.9187**	**1**
Control vs. Gravel-filled	**-0.0071**	**0.06193**	**0.911**	**1**
Control vs. Sand SBR	**0.0306**	**0.06012**	**0.6192**	**0.9949**
Control vs. Sand-filled	**0.0471**	**0.06618**	**0.4893**	**0.9771**
Capped vs. Gravel SBR	0.0027	0.05877	0.9641	1
Capped vs. Gravel-filled	-0.0123	0.06756	0.8589	1
Capped vs. Sand SBR	0.0254	0.086	0.7723	0.9996
Capped vs. Sand-filled	0.0419	0.06015	0.4983	0.9791
Gravel SBR vs. Gravel-filled	-0.0150	0.06625	0.825	0.9999
Gravel SBR vs. Sand SBR	0.0227	0.08663	0.7973	0.9998
Gravel SBR vs. Sand-filled	0.0392	0.06058	0.5288	0.9848
Gravel-filled vs. Sand SBR	0.0377	0.06774	0.5877	0.9923
Gravel-filled vs. Sand-filled	0.0542	0.06127	0.3928	0.9439
Sand SBR vs. Sand-filled	0.0165	0.0751	0.8296	0.9999

Total Shrubs Frequency

Treatment Effects

Category		*Estimate*	*SE*	*P-value*
Spring Type	Complex	0.0205	0.04263	0.6386
	Single	-0.0110	0.03271	0.7424
Flow Group	High	0.0529	0.04041	0.2129
	Low	-0.0183	0.036	0.6198
	Med High	-0.0227	0.03777	0.5581
	Med Low	0.0071	0.03603	0.8469
Treatment Type	Capped Pipes	0.0016	0.05476	0.9773
	Control	-0.0344	0.04606	0.4687
	Gravel SBR	0.0758	0.05293	0.1755
	Gravel-filled	-0.0024	0.0452	0.9592
	Sand SBR	-0.0461	0.05567	0.4227
	Sand-filled	0.0340	0.04485	0.4624

Yearly/Climatic Effects

	Estimate	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**0.0360**	**0.08046**	**0.6622**	**0.9972**
Control vs. Gravel SBR	**-0.1102**	**0.07769**	**0.1795**	**0.7163**
Control vs. Gravel-filled	**-0.0320**	**0.06733**	**0.6423**	**0.9963**
Control vs. Sand SBR	**0.0117**	**0.06164**	**0.8523**	**1**
Control vs. Sand-filled	**-0.0683**	**0.06857**	**0.3372**	**0.911**
Capped vs. Gravel SBR	-0.0743	0.06016	0.239	0.8131
Capped vs. Gravel-filled	0.0039	0.06742	0.9542	1
Capped vs. Sand SBR	0.0477	0.09326	0.6178	0.9948
Capped vs. Sand-filled	-0.0324	0.06246	0.613	0.9944
Gravel SBR vs. Gravel-filled	0.0782	0.06884	0.2765	0.858
Gravel SBR vs. Sand SBR	0.1219	0.09001	0.1987	0.7515
Gravel SBR vs. Sand-filled	0.0419	0.06126	0.5063	0.9807
Gravel-filled vs. Sand SBR	0.0437	0.07581	0.5739	0.9909
Gravel-filled vs. Sand-filled	-0.0363	0.06309	0.5747	0.991
Sand SBR vs. Sand-filled	-0.0801	0.07895	0.3292	0.9052

Total Forbs Frequency

Treatment Effects

Category		*Estimate*	*SE*	*P-value*
Spring Type	Complex	0.1244	0.04109	0.0097*
	Single	0.0907	0.03153	0.0129*
Flow Group	High	0.1224	0.03905	0.0079*
	Low	0.1003	0.03399	0.0112*
	Med High	0.0874	0.03496	0.0266*
	Med Low	0.1202	0.03494	0.0044*
Treatment Type	Capped Pipes	0.1550	0.05119	0.0097*
	Control	0.0922	0.04351	0.054*
	Gravel SBR	0.1655	0.05099	0.0064*
	Gravel-filled	0.0659	0.04213	0.1415
	Sand SBR	0.0908	0.05099	0.0984*
	Sand-filled	0.0761	0.04343	0.1032

Yearly/Climatic Effects

	Estimate	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**0.0628**	**0.07439**	**0.4141**	**0.9534**
Control vs. Gravel SBR	**-0.0733**	**0.07399**	**0.3401**	**0.9131**
Control vs. Gravel-filled	**0.0262**	**0.05954**	**0.6667**	**0.9974**
Control vs. Sand SBR	**0.0014**	**0.05929**	**0.9814**	**1**
Control vs. Sand-filled	**0.0161**	**0.06565**	**0.8106**	**0.9998**
Capped vs. Gravel SBR	-0.0105	0.05714	0.8569	1
Capped vs. Gravel-filled	0.0890	0.06714	0.2077	0.7667
Capped vs. Sand SBR	0.0642	0.08469	0.4621	0.9701
Capped vs. Sand-filled	0.0788	0.0591	0.2052	0.7626
Gravel SBR vs. Gravel-filled	0.0995	0.06626	0.157	0.6691
Gravel SBR vs. Sand SBR	0.0747	0.08456	0.3931	0.944
Gravel SBR vs. Sand-filled	0.0893	0.05921	0.1553	0.6653
Gravel-filled vs. Sand SBR	-0.0248	0.06608	0.7132	0.9988
Gravel-filled vs. Sand-filled	-0.0102	0.06136	0.8707	1
Sand SBR vs. Sand-filled	0.0147	0.07392	0.846	0.9999

Table 2.2. Cover measurements for spring sites. Tables represent total living cover and cover of functional groups. Treatment effect varied over all functional groups. However, yearly effect shows that no treatment changed significantly more or less than the control for any of the functional groups. This suggests that changes in treatment effect are driven by climatic changes. Significance denoted by (*).

Total Living Cover

Treatment Effects

Category		*Estimate*	*SE*	*P-value*
Spring Type	Complex	0.0460	0.06781	0.5099
	Single	0.0249	0.05177	0.6381
Flow Group	High	0.0290	0.08471	0.7378
	Low	0.0781	0.07352	0.3076
	Med High	0.0550	0.05674	0.3497
	Med Low	-0.0203	0.05912	0.7366
Treatment Type	Capped Pipes	0.1029	0.08096	0.2258
	Control	0.0921	0.07392	0.2349
	Gravel SBR	-0.0602	0.0864	0.4986
	Gravel-filled	0.0836	0.08672	0.3527
	Sand SBR	-0.0231	0.08171	0.7823
	Sand-filled	0.0173	0.06966	0.8083

Yearly/Climatic Effects

	Estimate	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**0.0109**	**0.1208**	**0.9297**	**1**
Control vs. Gravel SBR	**0.1522**	**0.1171**	**0.2163**	**0.7804**
Control vs. Gravel-filled	**0.0085**	**0.1273**	**0.9478**	**1**
Control vs. Sand SBR	**0.1151**	**0.09522**	**0.2481**	**0.8251**
Control vs. Sand-filled	**0.0748**	**0.1043**	**0.4859**	**0.9763**
Capped vs. Gravel SBR	0.1631	0.09606	0.1133	0.5557
Capped vs. Gravel-filled	0.0194	0.1176	0.8718	1
Capped vs. Sand SBR	0.1260	0.1349	0.3672	0.9303
Capped vs. Sand-filled	0.0857	0.09486	0.3827	0.9388
Gravel SBR vs. Gravel-filled	-0.1437	0.1361	0.3102	0.89
Gravel SBR vs. Sand SBR	-0.0371	0.1356	0.7886	0.9997
Gravel SBR vs. Sand-filled	-0.0774	0.09557	0.4326	0.9605
Gravel-filled vs. Sand SBR	0.1066	0.1246	0.4074	0.9506
Gravel-filled vs. Sand-filled	0.0663	0.1174	0.5816	0.9917
Sand SBR vs. Sand-filled	-0.0403	0.1171	0.7362	0.9992

Total Grass Cover

Treatment Effects

Category		*Estimate*	*SE*	*P-value*
Spring Type	Complex	0.0805	0.04319	0.0852*
	Single	0.0443	0.03324	0.2059
Flow Group	High	0.0635	0.04183	0.1532
	Low	0.0677	0.0354	0.0783*
	Med High	0.0745	0.03732	0.0672*
	Med Low	0.0438	0.03747	0.2636
Treatment Type	Capped Pipes	0.1371	0.05728	0.0325*
	Control	0.1174	0.04674	0.026*
	Gravel SBR	0.0647	0.05497	0.2601
	Gravel-filled	0.0174	0.04348	0.6954
	Sand SBR	-0.0046	0.05601	0.9361
	Sand-filled	0.0421	0.04658	0.3829

Yearly/Climatic Effects

	Estimate	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**0.0197**	**0.08229**	**0.8147**	**0.9999**
Control vs. Gravel SBR	**0.0527**	**0.07988**	**0.521**	**0.9835**
Control vs. Gravel-filled	**0.1000**	**0.0635**	**0.1393**	**0.627**
Control vs. Sand SBR	**0.1220**	**0.0638**	**0.0782**	**0.4371**
Control vs. Sand-filled	**0.0754**	**0.0704**	**0.3039**	**0.8846**
Capped vs. Gravel SBR	0.0724	0.06242	0.2671	0.8477
Capped vs. Gravel-filled	0.1197	0.07284	0.1243	0.5873
Capped vs. Sand SBR	0.1417	0.0952	0.1605	0.677
Capped vs. Sand-filled	0.0950	0.06483	0.1664	0.6897
Gravel SBR vs. Gravel-filled	0.0473	0.07035	0.513	0.9821
Gravel SBR vs. Sand SBR	0.0693	0.09229	0.4661	0.9712
Gravel SBR vs. Sand-filled	0.0227	0.06349	0.727	0.999
Gravel-filled vs. Sand SBR	0.0220	0.07024	0.7593	0.9995
Gravel-filled vs. Sand-filled	-0.0247	0.06395	0.706	0.9986
Sand SBR vs. Sand-filled	-0.0466	0.08073	0.5733	0.9909

Total Grasslikes Cover

Treatment Effects

Category		*Estimate*	*SE*	*P-value*
Spring Type	Complex	-0.0197	0.03139	0.542
	Single	-0.0105	0.02417	0.6716
Flow Group	High	-0.0238	0.03078	0.4539
	Low	-0.0008	0.02615	0.9753
	Med High	-0.0011	0.02703	0.9685
	Med Low	-0.0346	0.02704	0.223
Treatment Type	Capped Pipes	0.0057	0.04014	0.8888
	Control	0.0233	0.0342	0.5073
	Gravel SBR	-0.1126	0.04015	0.0149*
	Gravel-filled	0.0004	0.03162	0.9901
	Sand SBR	0.0096	0.04002	0.8151
	Sand-filled	-0.0168	0.03603	0.6481

Yearly/Climatic Effects

	Estimate	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**-0.0176**	**0.05775**	**0.7654**	**0.9996**
Control vs. Gravel SBR	**0.1359**	**0.05775**	**0.035**	**0.2412**
Control vs. Gravel-filled	**0.0229**	**0.04681**	**0.6325**	**0.9957**
Control vs. Sand SBR	**0.0138**	**0.04631**	**0.7708**	**0.9996**
Control vs. Sand-filled	**0.0402**	**0.05376**	**0.4684**	**0.9718**
Capped vs. Gravel SBR	0.1183	0.04462	0.02	0.1526
Capped vs. Gravel-filled	0.0053	0.05128	0.9189	1
Capped vs. Sand SBR	-0.0038	0.0662	0.9548	1
Capped vs. Sand-filled	0.0226	0.04918	0.6541	0.9968
Gravel SBR vs. Gravel-filled	-0.1130	0.05129	0.0463	0.2992
Gravel SBR vs. Sand SBR	-0.1221	0.0662	0.0879	0.473
Gravel SBR vs. Sand-filled	-0.0958	0.04921	0.0736	0.4194
Gravel-filled vs. Sand SBR	-0.0092	0.05113	0.8607	1
Gravel-filled vs. Sand-filled	0.0172	0.04739	0.722	0.999
Sand SBR vs. Sand-filled	0.0264	0.05975	0.6661	0.9974

Total Shrub Cover

Treatment Effects

Category		*Estimate*	*SE*	*P-value*
Spring Type	Complex	-0.0226	0.0391	0.5738
	Single	-0.0222	0.03011	0.4735
Flow Group	High	-0.0072	0.03819	0.8541
	Low	-0.0370	0.03232	0.2731
	Med High	-0.0380	0.03454	0.2914
	Med Low	-0.0075	0.03605	0.8395
Treatment Type	Capped Pipes	-0.0142	0.05318	0.794
	Control	-0.0369	0.04398	0.417
	Gravel SBR	-0.0646	0.05035	0.2222
	Gravel-filled	0.0187	0.04085	0.6545
	Sand SBR	-0.0186	0.05391	0.7351
	Sand-filled	-0.0189	0.04577	0.687

Yearly/Climatic Effects

	Estimate	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**0.0227**	**0.07837**	**0.7767**	**0.9997**
Control vs. Gravel SBR	**0.0277**	**0.07463**	**0.7167**	**0.9988**
Control vs. Gravel-filled	**-0.0556**	**0.05783**	**0.3541**	**0.9224**
Control vs. Sand SBR	**-0.0182**	**0.05841**	**0.7599**	**0.9995**
Control vs. Sand-filled	**-0.0180**	**0.07014**	**0.8014**	**0.9998**
Capped vs. Gravel SBR	0.0504	0.05694	0.3924	0.9437
Capped vs. Gravel-filled	-0.0329	0.06993	0.646	0.9965
Capped vs. Sand SBR	0.0045	0.0914	0.9618	1
Capped vs. Sand-filled	0.0047	0.05782	0.9367	1
Gravel SBR vs. Gravel-filled	-0.0833	0.06597	0.2291	0.7994
Gravel SBR vs. Sand SBR	-0.0459	0.08712	0.6071	0.994
Gravel SBR vs. Sand-filled	-0.0457	0.05878	0.4509	0.9667
Gravel-filled vs. Sand SBR	0.0373	0.06432	0.5714	0.9907
Gravel-filled vs. Sand-filled	0.0376	0.06433	0.5692	0.9904
Sand SBR vs. Sand-filled	0.0002	0.08156	0.9979	1

Total Forb Cover

Treatment Effects

Category		*Estimate*	*SE*	*P-value*
Spring Type	Complex	0.0302	0.05186	0.5702
	Single	0.0004	0.0398	0.9924
Flow Group	High	0.0207	0.05201	0.6968
	Low	0.0319	0.04182	0.4588
	Med High	-0.0023	0.04365	0.9594
	Med Low	0.0108	0.04398	0.8098
Treatment Type	Capped Pipes	0.0308	0.06567	0.6464
	Control	-0.0275	0.05649	0.6351
	Gravel SBR	0.0939	0.06456	0.1696
	Gravel-filled	0.0086	0.05144	0.8699
	Sand SBR	-0.0562	0.0665	0.4134
	Sand-filled	0.0421	0.05471	0.4553

Yearly/Climatic Effects

	Estimate	*SE*	*P-value*	*Adj. P*
Control vs. Capped	**0.0583**	**0.09705**	**0.5584**	**0.9891**
Control vs. Gravel SBR	**-0.1213**	**0.09495**	**0.2236**	**0.7915**
Control vs. Gravel-filled	**-0.0360**	**0.07769**	**0.6504**	**0.9967**
Control vs. Sand SBR	**0.0288**	**0.07477**	**0.7068**	**0.9986**
Control vs. Sand-filled	**-0.0696**	**0.08306**	**0.4175**	**0.9548**
Capped vs. Gravel SBR	-0.0631	0.07265	0.4012	0.9478
Capped vs. Gravel-filled	0.0223	0.08237	0.7913	0.9998
Capped vs. Sand SBR	0.0870	0.1107	0.4459	0.9651
Capped vs. Sand-filled	-0.0113	0.07618	0.8847	1
Gravel SBR vs. Gravel-filled	0.0853	0.08226	0.3187	0.897
Gravel SBR vs. Sand SBR	0.1501	0.1087	0.1907	0.7374
Gravel SBR vs. Sand-filled	0.0518	0.07495	0.5018	0.9798
Gravel-filled vs. Sand SBR	0.0648	0.08542	0.4617	0.97
Gravel-filled vs. Sand-filled	-0.0335	0.07531	0.6636	0.9973
Sand SBR vs. Sand-filled	-0.0983	0.09434	0.3164	0.8952

Table 2.3. Changes in cover of *J. balticus* and *A. tridentata*. Cover of *A. tridentata* did not change between the pre- and post-treatment periods. In treatment effects, *J. balticus* decreased at gravel SBR sites. However, yearly effects show no difference between the change in gravel SBR and the change in the control, so this decrease was driven by yearly changes not treatment. Significance denoted by (*).

Total JUBA Cover

	Treatment Effects				Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	-0.0018	0.0314	0.9549	**Control vs. Capped**	**-0.0427**	**0.06384**	**0.5191**	**0.9817**
	Single	0.0007	0.02419	0.9772	**Control vs. Gravel SBR**	**0.1940**	**0.06324**	**0.0119**	**0.0923**
Flow Group	High	0.0180	0.03249	0.5911	**Control vs. Gravel-filled**	**-0.0160**	**0.04507**	**0.7306**	**0.999**
	Low	-0.0318	0.02953	0.3067	**Control vs. Sand SBR**	**-0.0240**	**0.04504**	**0.6059**	**0.9933**
	Med High	-0.0062	0.02769	0.8286	**Control vs. Sand-filled**	**0.0143**	**0.05454**	**0.7992**	**0.9998**
	Med Low	0.0177	0.02989	0.5668	Capped vs. Gravel SBR	0.1514	0.0521	0.0157	0.1177
Treatment Type	Capped Pipes	-0.0081	0.04759	0.869	Capped vs. Gravel-filled	-0.0586	0.05856	0.3404	0.9074
	Control	0.0346	0.03307	0.3199	Capped vs. Sand SBR	-0.0667	0.07214	0.3772	0.9313
	Gravel SBR	-0.1594	0.0474	0.0072*	Capped vs. Sand-filled	-0.0284	0.06238	0.6584	0.9968
	Gravel-filled	0.0506	0.03119	0.136	Gravel SBR vs. Gravel-filled	-0.2100	0.05877	0.0051	0.0429*
	Sand SBR	0.0586	0.03923	0.1661	Gravel SBR vs. Sand SBR	-0.2180	0.07114	0.0119	0.0928*
	Sand-filled	0.0204	0.04106	0.6306	Gravel SBR vs. Sand-filled	-0.1798	0.06379	0.0182	0.1337
					Gravel-filled vs. Sand SBR	-0.0080	0.04954	0.8744	1
					Gravel-filled vs. Sand-filled	0.0302	0.0499	0.5585	0.9882
					Sand SBR vs. Sand-filled	0.0382	0.06039	0.5408	0.9856

Total ARTR Cover

	Treatment Effects				Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	0.0234	0.08831	0.7984	**Control vs. Capped**	**-0.0505**	**0.1382**	**0.7255**	**0.9987**
	Single	-0.0055	0.06409	0.9337	**Control vs. Gravel SBR**	**0.1260**	**0.1378**	**0.3907**	**0.9306**
Flow Group	High	0.0158	0.08906	0.8639	**Control vs. Gravel-filled**	**0.0138**	**0.1311**	**0.9189**	**1**
	Low	0.1019	0.07716	0.2281	**Control vs. Sand SBR**	**-0.1075**	**0.1899**	**0.5889**	**0.9904**
	Med High	-0.0009	0.08998	0.9923	**Control vs. Sand-filled**	**-0.0846**	**0.1325**	**0.5433**	**0.9837**
	Med Low	-0.0811	0.1009	0.4483	Capped vs. Gravel SBR	0.0755	0.09715	0.4625	0.9631
Treatment Type	Capped Pipes	-0.0419	0.09655	0.6776	Capped vs. Gravel-filled	-0.0367	0.1362	0.7956	0.9997
	Control	0.0087	0.09073	0.9267	Capped vs. Sand SBR	-0.1581	0.2175	0.4909	0.9719
	Gravel SBR	-0.1174	0.09444	0.2539	Capped vs. Sand-filled	-0.1352	0.1101	0.2595	0.8125
	Gravel-filled	-0.0052	0.08916	0.9552	Gravel SBR vs. Gravel-filled	-0.1122	0.1333	0.4279	0.9495
	Sand SBR	0.1162	0.1677	0.5108	Gravel SBR vs. Sand SBR	-0.2336	0.2147	0.3127	0.872
	Sand-filled	0.0933	0.09389	0.3535	Gravel SBR vs. Sand-filled	-0.2107	0.1102	0.0975	0.465
					Gravel-filled vs. Sand SBR	-0.1214	0.1779	0.5171	0.9785
					Gravel-filled vs. Sand-filled	-0.0985	0.1374	0.4968	0.9736
					Sand SBR vs. Sand-filled	0.0229	0.2087	0.9157	1

Table 2.4. Biomass measurements for three functional groups (grasses, grasslikes and forbs) and total wet and dry weights. Treatment effects show both wet and dry weights increasing in several categories. Grasses also increase in most categories. The biomass of forbs only increases for medium-low flow groups and sand SBR treatments. Grasslike biomass does not change. Yearly effect shows that none of the treatment increased or decreased significantly from the control. This suggests that biomass change was driven by increased precipitation during the post-treatment period. Significance denoted by (*).

Biomass-Total Wet Weight

		Treatment Effects			Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	128.5700	39.2442	0.006*	**Control vs. Capped**	**13.0930**	**68.8769**	**0.8522**	**1**
	Single	38.6277	29.9982	0.2203	**Control vs. Gravel SBR**	**16.6256**	**69.4267**	**0.8145**	**0.9999**
Flow Group	High	133.3400	36.5128	0.0029*	**Control vs. Gravel-filled**	**6.7836**	**55.0691**	**0.9038**	**1**
	Low	46.4285	32.0783	0.1715	**Control vs. Sand SBR**	**73.9189**	**58.5164**	**0.2287**	**0.7989**
	Med High	97.1579	33.5089	0.0124*	**Control vs. Sand-filled**	**-110.3600**	**61.2497**	**0.0948**	**0.4968**
	Med Low	57.4573	32.0808	0.0966*	Capped vs. Gravel SBR	29.7186	53.0215	0.5847	0.992
Treatment Type	Capped Pipes	92.3351	47.415	0.0734*	Capped vs. Gravel-filled	19.8767	60.346	0.7471	0.9994
	Control	79.2420	41.6198	0.0793*	Capped vs. Sand SBR	87.0119	79.4465	0.2933	0.8748
	Gravel SBR	62.6164	47.2982	0.2084	Capped vs. Sand-filled	-97.2714	54.9074	0.0999	0.5138
	Gravel-filled	72.4584	37.8644	0.0779*	Gravel SBR vs. Gravel-filled	-9.8420	60.6401	0.8736	1
	Sand SBR	5.3232	48.2005	0.9137	Gravel SBR vs. Sand SBR	57.2933	78.9182	0.4807	0.975
	Sand-filled	189.6100	40.098	0.0004*	Gravel SBR vs. Sand-filled	-126.9900	54.8319	0.0375	0.2547
					Gravel-filled vs. Sand SBR	67.1352	62.1941	0.3	0.8811
					Gravel-filled vs. Sand-filled	-117.1500	55.1118	0.0533	0.3326
					Sand SBR vs. Sand-filled	-184.2800	69.1127	0.0194	0.149

Biomass-Total Dry Weight

		Treatment Effects			Yearly/Climatic Effects				
Category		*Estimate*	*SE*	*P-value*		*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	13.0777	7.1148	0.089*	**Control vs. Capped**	**-13.6171**	**12.4524**	**0.294**	**0.8755**
	Single	15.1491	5.4345	0.0154*	**Control vs. Gravel SBR**	**22.2948**	**12.4429**	**0.0965**	**0.5024**
Flow Group	High	16.2182	6.7304	0.0315*	**Control vs. Gravel-filled**	**5.3964**	**9.9293**	**0.596**	**0.9931**
	Low	12.7410	5.8341	0.0479*	**Control vs. Sand SBR**	**14.8590**	**10.5192**	**0.1813**	**0.7197**
	Med High	19.6792	5.8323	0.005*	**Control vs. Sand-filled**	**5.8585**	**11.1996**	**0.6097**	**0.9942**
	Med Low	7.8153	5.8232	0.2025	Capped vs. Gravel SBR	8.6777	9.5299	0.3791	0.9369
Treatment Type	Capped Pipes	10.8340	8.5334	0.2265	Capped vs. Gravel-filled	-8.2207	10.8902	0.4638	0.9706
	Control	24.4510	7.5329	0.0064*	Capped vs. Sand SBR	1.2419	14.2608	0.9319	1
	Gravel SBR	2.1563	8.5371	0.8045	Capped vs. Sand-filled	-7.7586	9.9308	0.4486	0.966
	Gravel-filled	19.0547	6.835	0.0154*	Gravel SBR vs. Gravel-filled	-16.8984	10.8857	0.1446	0.64
	Sand SBR	9.5920	8.6514	0.2876	Gravel SBR vs. Sand SBR	-7.4358	14.2704	0.6111	0.9943
	Sand-filled	18.5926	7.2501	0.0235*	Gravel SBR vs. Sand-filled	-16.4363	9.9382	0.1221	0.5811
					Gravel-filled vs. Sand SBR	9.4626	11.1784	0.4126	0.9528
					Gravel-filled vs. Sand-filled	0.4621	10.0313	0.964	1
					Sand SBR vs. Sand-filled	-9.0005	12.3658	0.4796	0.9748

Biomass-Total Grass Weight

Category		Treatment Effects Estimate	SE	P-value	Yearly/Climatic Effects	Estimate	SE	P-value	Adj. P
Spring Type	Complex	8.8769	5.6893	0.1427	**Control vs. Capped**	**-14.8791**	**10.5575**	**0.1822**	**0.7215**
	Single	10.2453	4.3777	0.0359*	**Control vs. Gravel SBR**	**13.7738**	**10.4749**	**0.2113**	**0.7725**
Flow Group	High	14.9674	5.579	0.0188*	**Control vs. Gravel-filled**	**9.2382**	**8.9852**	**0.3226**	**0.9002**
	Low	9.4571	4.6879	0.0648*	**Control vs. Sand SBR**	**7.1531**	**8.5499**	**0.4179**	**0.955**
	Med High	12.2811	5.0244	0.0295*	**Control vs. Sand-filled**	**1.3175**	**9.4683**	**0.8915**	**1**
	Med Low	1.5387	4.8797	0.7575	Capped vs. Gravel SBR	-1.1053	8.0839	0.8933	1
Treatment Type	Capped Pipes	2.4089	7.3012	0.7467	Capped vs. Gravel-filled	-5.6409	10.4397	0.5981	0.9933
	Control	17.2880	6.1284	0.0144*	Capped vs. Sand SBR	-7.7260	11.9519	0.5293	0.9849
	Gravel SBR	3.5142	7.2249	0.6348	Capped vs. Sand-filled	-13.5616	8.3785	0.1295	0.6016
	Gravel-filled	8.0498	6.8414	0.2604	Gravel SBR vs. Gravel-filled	-4.5356	10.1442	0.6622	0.9972
	Sand SBR	10.1349	7.3303	0.1901	Gravel SBR vs. Sand SBR	-6.6207	11.9792	0.5899	0.9925
	Sand-filled	15.9706	6.3732	0.0263*	Gravel SBR vs. Sand-filled	-12.4564	8.4542	0.1644	0.6855
					Gravel-filled vs. Sand SBR	-2.0851	10.5214	0.846	0.9999
					Gravel-filled vs. Sand-filled	-7.9208	10.058	0.4451	0.9649
					Sand SBR vs. Sand-filled	-5.8356	10.4384	0.5856	0.9921

Biomass-Total Grasslikes Weight

Category		Treatment Effects Estimate	SE	P-value	Yearly/Climatic Effects	Estimate	SE	P-value	Adj. P
Spring Type	Complex	-0.7594	3.9435	0.8503	**Control vs. Capped**	**4.8171**	**6.7547**	**0.4884**	**0.9769**
	Single	4.0147	3.007	0.2047	**Control vs. Gravel SBR**	**5.2568**	**6.7589**	**0.4506**	**0.9666**
Flow Group	High	-2.8534	3.7924	0.4652	**Control vs. Gravel-filled**	**-2.5747**	**5.5882**	**0.6526**	**0.9968**
	Low	4.4017	3.1231	0.1822	**Control vs. Sand SBR**	**2.3378**	**5.5257**	**0.6792**	**0.9978**
	Med High	3.5413	3.1733	0.2846	**Control vs. Sand-filled**	**4.3530**	**6.0152**	**0.4821**	**0.9754**
	Med Low	1.4209	3.2365	0.6678	Capped vs. Gravel SBR	10.0739	5.21	0.0752	0.4258
Treatment Type	Capped Pipes	7.2041	4.691	0.1486	Capped vs. Gravel-filled	2.2424	6.0275	0.7159	0.9988
	Control	2.3870	4.063	0.5669	Capped vs. Sand SBR	7.1549	7.7641	0.3736	0.9339
	Gravel SBR	-2.8699	4.6841	0.5507	Capped vs. Sand-filled	9.1701	5.4284	0.115	0.5607
	Gravel-filled	4.9617	3.7152	0.2046	Gravel SBR vs. Gravel-filled	-7.8316	6.017	0.2157	0.7794
	Sand SBR	0.0492	4.6646	0.9918	Gravel SBR vs. Sand SBR	-2.9190	7.7581	0.7128	0.9988
	Sand-filled	-1.9660	3.9504	0.627	Gravel SBR vs. Sand-filled	-0.9038	5.4224	0.8702	1
					Gravel-filled vs. Sand SBR	4.9125	5.9459	0.4236	0.9572
					Gravel-filled vs. Sand-filled	6.9277	5.4231	0.2238	0.7917
					Sand SBR vs. Sand-filled	2.0152	6.7458	0.7699	0.9996

Biomass-Total Forbs Weight

Category		Treatment Effects Estimate	SE	P-value	Yearly/Climatic Effects	Estimate	SE	P-value	Adj. P
Spring Type	Complex	4.3261	2.7028	0.1335	**Control vs. Capped**	**1.3650**	**5.2968**	**0.8007**	**0.9998**
	Single	1.4906	2.0797	0.4862	**Control vs. Gravel SBR**	**-2.2618**	**5.5615**	**0.6909**	**0.9982**
Flow Group	High	4.0595	2.6948	0.1559	**Control vs. Gravel-filled**	**-2.6908**	**4.1096**	**0.524**	**0.984**
	Low	0.6355	2.2175	0.779	**Control vs. Sand SBR**	**1.2113**	**4.1519**	**0.7751**	**0.9996**
	Med High	2.3501	2.7355	0.4059	**Control vs. Sand-filled**	**-5.6031**	**4.7579**	**0.2601**	**0.8396**
	Med Low	4.5883	2.4503	0.0838*	Capped vs. Gravel SBR	-0.8968	3.8821	0.8209	0.9999
Treatment Type	Capped Pipes	2.4884	3.4579	0.4845	Capped vs. Gravel-filled	-1.3259	4.4374	0.7698	0.9996
	Control	1.1234	3.2162	0.7325	Capped vs. Sand SBR	2.5763	5.7139	0.6595	0.9971
	Gravel SBR	3.3852	3.6087	0.3653	Capped vs. Sand-filled	-4.2381	3.9679	0.3049	0.8854
	Gravel-filled	3.8143	2.7245	0.1849	Gravel SBR vs. Gravel-filled	-0.4290	4.601	0.9271	1
	Sand SBR	-0.0879	3.4268	0.9799	Gravel SBR vs. Sand SBR	3.4731	5.8248	0.5612	0.9895
	Sand-filled	6.7265	2.9464	0.0399*	Gravel SBR vs. Sand-filled	-3.3413	4.018	0.4207	0.9561
					Gravel-filled vs. Sand SBR	3.9021	4.366	0.3877	0.9414
					Gravel-filled vs. Sand-filled	-2.9122	4.0531	0.4851	0.9761
					Sand SBR vs. Sand-filled	-6.8144	4.9986	0.1959	0.7468

Table 2.5. Number of wildlife species recorded on trap cameras at our spring sites. All categories that experienced a significant difference in species number saw a decrease. None of the treatments were significantly different from the controls suggesting that changes were driven by a wetter year post-treatment. Significance denoted by (*).

Species Number

Category	Treatment Effects	*Estimate*	*SE*	*P-value*	Yearly/Climatic Effects	*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	-1.3014	0.515	0.0281*	**Control vs. Capped**	**0.6725**	**0.9894**	**0.5107**	**0.9807**
	Single	-1.4316	0.4239	0.0062*	**Control vs. Gravel SBR**	**-1.3771**	**0.9185**	**0.1619**	**0.6717**
Flow Group	High	-0.1201	0.5305	0.8251	**Control vs. Gravel-filled**	**-1.1495**	**0.721**	**0.1392**	**0.6179**
	Low	-1.5538	0.4127	0.0031*	**Control vs. Sand SBR**	**-2.4642**	**0.8425**	**0.0138**	**0.1078**
	Med High	-1.5234	0.5142	0.0129*	**Control vs. Sand-filled**	**0.6578**	**0.8574**	**0.4591**	**0.9677**
	Med Low	-2.2687	0.4551	0.0004*	Capped vs. Gravel SBR	-0.7046	0.8232	0.4103	0.9496
Treatment Type	Capped Pipes	-1.5282	0.7361	0.0621*	Capped vs. Gravel-filled	-0.4770	0.8771	0.5974	0.9928
	Control	-2.2007	0.5281	0.0016*	Capped vs. Sand SBR	-1.7917	1.3339	0.2063	0.7572
	Gravel SBR	-0.8236	0.6642	0.2408	Capped vs. Sand-filled	1.3303	0.876	0.1571	0.6608
	Gravel-filled	-1.0513	0.4906	0.0554*	Gravel SBR vs. Gravel-filled	0.2276	0.8241	0.7875	0.9997
	Sand SBR	0.2635	0.768	0.738	Gravel SBR vs. Sand SBR	-1.0871	1.1599	0.3688	0.9283
	Sand-filled	-2.8585	0.6145	0.0007*	Gravel SBR vs. Sand-filled	2.0349	0.8407	0.034	0.2289
					Gravel-filled vs. Sand SBR	-1.3147	0.9197	0.1806	0.7106
					Gravel-filled vs. Sand-filled	1.8073	0.7865	0.0422	0.2716
					Sand SBR vs. Sand-filled	3.1220	1.0577	0.0132	0.1034

Table 2.6. The number of wildlife visits to spring sites. The treatment effects estimate column reports the difference between pre- and post-treatment periods. Overall, average abundance of wildlife visits our sites decreased after treatment implementation. Yearly/climatic effects appear to be the driving force behind the changes seen in the treatment effects. One treatment, Sand SBR, reported decreased wildlife visits in comparison to the controls, independent of precipitation changes. Significance denoted by (*).

Wildlife Visits

Category	Treatment Effects	*Estimate*	*SE*	*P-value*	Yearly/Climatic Effects	*Estimate*	*SE*	*P-value*	*Adj. P*
Spring Type	Complex	-275.3900	69.1945	0.0022*	**Control vs. Capped**	**-125.8200**	**127.76**	**0.3459**	**0.9136**
	Single	-147.4200	58.4258	0.0283*	**Control vs. Gravel SBR**	**70.8716**	**126.31**	**0.586**	**0.9917**
Flow Group	High	-109.6900	70.7211	0.1492	**Control vs. Gravel-filled**	**-110.1100**	**98.0315**	**0.2853**	**0.8621**
	Low	-148.9400	63.6423	0.0392*	**Control vs. Sand SBR**	**-365.5600**	**112.59**	**0.0078**	**0.0649***
	Med High	-264.4100	61.3232	0.0012*	**Control vs. Sand-filled**	**52.5341**	**124.87**	**0.6821**	**0.9978**
	Med Low	-322.5900	59.3524	0.0002*	Capped vs. Gravel SBR	-54.9494	94.3393	0.572	0.9902
Treatment Type	Capped Pipes	-374.9700	90.5317	0.0016*	Capped vs. Gravel-filled	-235.9300	111.01	0.057	0.3416
	Control	-249.1500	71.9074	0.0053*	Capped vs. Sand SBR	-491.3800	166.3	0.0131	0.1029
	Gravel SBR	-320.0200	88.3815	0.004*	Capped vs. Sand-filled	-73.2869	128.95	0.5812	0.9912
	Gravel-filled	-139.0400	66.7154	0.0613*	Gravel SBR vs. Gravel-filled	-180.9800	109.7	0.1272	0.5864
	Sand SBR	116.4100	102.03	0.2781	Gravel SBR vs. Sand SBR	-436.4300	162.43	0.0211	0.1551
	Sand-filled	-301.6800	100.8	0.0122*	Gravel SBR vs. Sand-filled	-18.3375	128.74	0.8893	1
					Gravel-filled vs. Sand SBR	-255.4500	124.5	0.0648	0.3748
					Gravel-filled vs. Sand-filled	162.6400	119.89	0.2021	0.7501
					Sand SBR vs. Sand-filled	418.1000	152.96	0.0195	0.1446