Water Quality and Soil Health Response to Beaver Dam Analogs in an Intermittent Stream in Southern Idaho

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Abstract

Beaver dam analogs are gaining interest in the Northwestern United States as a restoration technique to enhance riparian health to enhance resilience to increased drought because they are an accessible technology. Beaver dam analogs have been shown to provide the same benefits as natural beaver dams such as increased surface water storage, sediment aggradation, reduced stream channel incision, increased floodplain inundation, and increased hyporheic exchange. Intermittent streams make up for more than half of all streams in the United States and have further decreased flow in response to low snow melt and higher temperatures earlier in the season attributed to global warming. Beaver Dam analogs were installed in an intermittent system on rangeland in Southcentral Idaho in three treatment clusters with a reference reach upstream. Water and soil samples were taken in the summer of 2021 pre-installation of beaver dam analogs to establish baseline conditions and samples in 2022 represent the first water year within the stream with the structures. Significant drought occurred in both seasons with water flowing until May and water present in some locations until early July. Stream temperature, phosphate, nitrogen, and dissolved oxygen were monitored to assess whether beaver dam analogs could increase eutrophication risk. All constituents showed no significant change in comparison to the reference meadow, therefore beaver dam analogs did not affect eutrophication risk within this system. Soil samples were taken in two locations in each reach (reference, meadow 1, meadow 2, and meadow 3) at 2, 4, and 6 feet from the stream channel to observe lateral change in biogeochemical parameters. Carbon and nitrogen stocks had no significant changes. Microbial biomass in 2022 samples were significantly higher compared to 2021 in all locations. Soil Moisture also increased in all locations, the correlation between higher microbial biomass due to increased soil moisture is clear because of the consistency throughout the reference and treatment meadows. Our findings indicate little change within one water year but show promise in increasing soil process efficiency through wetting events if enough water is supplied to spill onto the floodplain.

Acknowledgments

I would like to acknowledge my advisor Laurel Lynch for taking me on as her first student and supporting me throughout my master's program, allowing me to grow professionally and expand my knowledge in soil and water research. I would also like to thank Michael Strickland for his help and allowing me to utilize his lab for my experiments. Julia Piaskowski helped with statistical models and analysis. Torrey Stephenson ran soil samples on the EA-IRMS. Eric Winford coordinated beaver dam analog installation and study site infrastructure as well as collected flume measurements. Cooper Moon assisted with field work and sampling. I worked with Ellen Incelli who is researching geomorphology at the same study site.

Dedication

I would like to dedicate this thesis to my parents. My dad provided housing for my field work, career advice, and field assistance. My mother helped me settle in Moscow and provided me with moral support throughout my master's program. I want to thank my brother for always believing in me. I also would like to thank my friends who have given me both personal and educational help throughout my time here at UI. Water resources beer nights every Friday gave me the community I needed when I was all alone here, I will be forever grateful to Galen Richards for organizing these and becoming a good friend of mine. Thanks to my fellow water resources buddies Mike Vogler and Reid Sutton for all the entertaining stories and pints shared. Furthermore, my friends from all around the world who brought me joy in my time here and the comic relief necessary during the tough times. Abraham Hernandez thank you for always encouraging me to go out and have fun and teaching me that I do in fact always need a jacket. Mandira Panta thanks for all the amazing meals you have fed me and the numerous times you've hosted movie nights and momo making parties. Marwa Elsayed thank you for accompanying me in the office and helping me stay focused, going to the buffet together, being my gym buddy, and belay buddy. Thanks to Kameron Richardson for always knowing when it's time to drink and his discussions on unsolved science topics like the elusive formation of the Carolina Bays. Thank you, Katie Lambert, for all your help with lab procedures that I would never have been able to get done in time without your help, all the inside jokes like our favorite gas station in Lapwai, and all the movie nights like Midsommar. Thanks to Kiran Adhikari who has helped break up the mundaneness of office life, there will be many more memories to be had. Thank you to my friends out of town Lexie Wright, Aryan Parthor, Rachael Nigro, and Ashely Bryant for the talks and visits to Moscow. Without my community of friends here in Moscow and others far away I would not be where I am today, you have all shaped me into the person I am and helped me discover what I am capable o

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Chapter 1: Literature Review

1.1 Grazing Impacts on Soil and Water

There are approximately 770 million acres of rangeland within the United States, 43% of which is managed by the federal government (About Rangeland Management). Livestock grazing is prevalent across rangeland and proper stewardship is needed to maintain ecosystem services. Land management is complicated in arid drylands with low precipitation inputs and high evaporative potential (Safriel et al., 2005). Overgrazing in these regions can greatly change the composition and quality of habitat (Conroy et al., 2016; J. H. Miller, 2001; Milchunas & Lauenroth, 1993; J. J. Miller et al., 2014; Skarpe, 1990), for example, by accelerating desertification processes that gradually replace grasses with shrubs (Browning & Archer, 2011). Land conversion from grassland to shrubland can reduce biodiversity and aboveground net primary productivity (Pierce et al., 2019), decreasing the overall economic value of the land. As grasslands currently account for approximately 30% of the global terrestrial carbon stock (White et al., 2000). Grassland management is a key factor meeting natural climate solution goals (e.g., increasing rates of soil carbon sequestration (Knapp et al., 2008; Lange et al., 2015).

Grazing can have a variety of effects on soil function. Grazing intensity can change the effect of climate change on soils. For example, overgrazing may decrease soil carbon storage and release carbon dioxide to the atmosphere, inducing a positive feedback on climate change (Ylänne et al., 2020; Zhou et al., 2017). In tundra soils, the combination of warming temperatures and heavy grazing over a long period of time decreased soil carbon stocks by reducing soil moisture and increasing microbial activity (Ylänne et al., 2020). Multiple studies have found mixed results on soil nitrogen and soil carbon storage with no found relationship to pH or soil phosphorus (Milchunas & Lauenroth, 1993), microbial biomass or inorganic nitrogen (Wienhold et al., 2001).

Previous studies have found that grazing can increase (Reeder & Schuman, 2002), decrease (Ylänne et al., 2020), or have no effect (Carey et al., 2020) on soil organic carbon. Inconsistent relationships are likely driven by variation across studies in underlying climate factors. For example, Reeder & Schuman (2002) studied grazing impacts on semi-arid land while Ylänne et al. (2020) tested how land management affected carbon cycling in tundra soils. In one meta-analysis, using 17 studies from across the US, grazing was not significantly correlated with soil organic carbon stocks, but regional climate and land use were (McSherry & Ritchie, 2013). Another series of studies in California did not reveal a significant correlation between soil organic carbon and grazing intensity, but did show that soil organic carbon were significantly correlated with regional climate (Carey et al., 2020). Meta-analyses offer a powerful approach to disentangle relationships between grazing and soil chemistry, but literature reviews uncovered contrasting results. For example, one review included 236 data sets testing the effects of grazing intensity on rangeland soils. Studies included control (ungrazed) and treatment (grazed) plots with varying histories of grazing and land management. The authors identified no significant correlations between grazing intensity and soil carbon or nitrogen stocks, which they speculated was because climate varied among sites and more strongly influenced soil chemistry than grazing. Another meta-analysis (n = 115) found that grazing decreased soil carbon and nitrogen stocks and microbial biomass (Zhou et al., 2017). Mixed results could be caused by different levels of grazing, land management practices, or environmental conditions. These results suggest that sustainable land management practices can greatly improve grassland function, but effect sizes will be constrained by climatic and land use limitations. Riparian areas within arid grasslands are of particular interest because they have high carbon sequestration potential and relatively elevated concentrations of water and nutrients (Mendez-Estrella et al., 2017; L. M. Norman et al., 2022).

Different grazing practices can also influence the physical environment, for example, by altering rates of soil erosion (Y. Li et al., 2019). Erosion can reduce the ability of soils to retain organic matter, including within the litter layer and surface soil horizons (Hancock et al., 2019). Grazing intensity also impacts soil processes. For example, light grazing was found to decrease soil respiration and soil erosion and to stimulate plant diversity and turnover (Yu et al., 2019). In contrast, heavy grazing has been shown to increase carbon dioxide output and decrease belowground root biomass (Dai et al., 2019). Greater soil compaction by livestock can reduce root biomass and water infiltration, with negative feedbacks on plant growth and forage quality (Gao et al., 2008; Veldhuis et al., 2014). Because soil respiration is already predicted to increase under global climate change (e.g., by reducing the efficiency of microbial metabolism and accelerating biological process rates) it is imperative we manage grazing intensity to facilitate CO_2 uptake via net primary productivity and promote the sequestration of new carbon inputs (Milchunas & Lauenroth, 1993; Perry et al., 2012; Silverman et al., 2019).

Approximately 80% of Western US streams have been damaged by grazing livestock (Belsky et al., 1999). Negative environmental impacts of overgrazing on riparian systems include modifying stream morphology (e.g., by disturbing soil within the streambank), compacting soils, increasing erosion and nutrient loading of soil and water systems, and decreasing plant biomass (Belsky et al., 1999; O'Callaghan et al., 2019; Owens et al., 1989; Thornton & Elledge, 2021). Some studies suggest stream connectivity governs the impact cattle-derived nutrients may have on downstream water quality (Conroy et al., 2016). For example, disconnected streams with losing reaches will not carry nutrients downstream, whereas continuous or gaining reaches within streams have the ability to transport nitrates and phosphates downstream (Covino, 2017). Riparian areas are particularly sensitive because water allows for the solubilization and export of nutrients from cattle waste. Riparian areas comprise less than 2% of the land area in the Western U.S., but provide a disproportionate amount of ecosystem services, including habitat for over half of plant and vertebrate species (Wentzel & Hull, 2021). Protecting these regions is thus a high priority.

Losing and gaining stream sections are controlled by groundwater movement. Losing streams are characterized by flow from surface water into groundwater systems while gaining streams receive groundwater seepage. Losing and gaining reaches themselves have hyporheic zones where surface and groundwater mix, but in losing streams this is limited to available surface water (Haggerty et al., 2002; Harvey & Fuller, 1998). As a result, drought-impacted streams may have reduced hyporheic exchange, which can impair ecosystem function and reduce nutrient exchange across this critical interface (Coulson et al., 2021).

Riparian systems are desirable places for cattle to congregate because they tend to have more abundant and nutritious forage and greater water availability than adjacent uplands. As a result, sites with in-stream water sources are often more impacted by cattle than those where water is supplied off-stream (Porath et al., 2002). In-stream cattle activity can also directly increase erosion rates, decreasing bank stability and the productivity of riparian vegetation (Agouridis et al., 2005). Cattle increase erosion through degradation of steam cutbanks (Schwarte et al., 2011), which increases stream sediment aggradation and widens stream channels (McIver & McInnis, 2007). In a study on cattle impact on intermittent streams though, there was no significant impact on reaches with grazing, most likely due to limited water availability leading to less intensive grazing next to and within the stream (George et al., 2002).

Nitrate and ammonium concentration in streams with cattle access are often higher than fenced areas within the same field site, with the highest concentrations of nitrogen found in test-sites nearest the stream (Davies-Colley et al., 2004; Miller et al., 2014). The direct impact of grazing on downstream function depends on grazing intensity and water quantity. Cattle waste deposited near a stream could increase nitrate, ammonium, and phosphorus loading of riparian systems (O'Callaghan et al., 2019). Well-functioning riparian systems (e.g., those with floodplain connectivity and abundant riparian vegetation) can act as a filter, removing up to 90% of nitrogen, phosphorus, and organic solids before they are exported into the stream (D. Johnson, 2019). For example, in Idaho, riparian areas isolated from grazing had higher densities of riparian vegetation, which reduced nutrient loading to streams (Dauwalter et al., 2018). Not all solutions to grazing impacts on riparian systems need be so extreme; there is room for studying the impact of other restoration practices (e.g., rotational grazing) that show promise in terms of balancing economic and ecological costs (Hulvey et al., 2021). Regardless, land management and restoration practices are often necessary to restore riparian function and increase the ability of natural systems to sequester carbon and act as natural filtration systems to increase water quality in above- and belowground systems.

1.2 Intermittent Streams

Intermittent and ephemeral streams account for nearly half of all streams on a global scale (Shumilova et al., 2019) and more than 59% of streams in the US (Levick et al., 2008) making them critical study systems. Intermittent streams are categorized as streams that flow 20-80% of the year (Datry et al., 2017). In desert environments, intermittent streams develop in regions with low rates of precipitation and high rates of evapotranspiration. Intermittent streams are highly affected by human activity and take a long time to recover from impacts. Despite their short flow seasons, intermittent streams are critical suppliers of water and nutrients in arid and semiarid environments (Levick et al., 2008).

Because there is a higher ratio of water surface area to sediment in intermittent streams, phosphorus and nitrogen runoff may not affect downstream water quality as much as a permanent stream. For example, intermittent streams often exhibit high levels of phosphorus adsorption (Jarvie et al., 2002). The lower flow velocity of intermittent streams relative to perennial rivers may increase nutrient uptake by facilitating greater interactions between nutrients and adsorbing sediments (Wollheim et al., 2006). While nitrates can easily be dissolved by water, the slower movement of soil pore waters to receiving stream channels may increase the uptake of plant-assimilable nutrients, reducing nutrient loading in streams and downstream export (Roley et al., 2012). Also of note, the lower velocities of small intermittent streams may increase the retention of litter-derived carbon within the stream channel (Webster et al., 1999) and facilitate sediment accumulation (Sutfin et al., 2016). Incised streams that have been restored by legacy sediment removal to increase floodplain connectivity showed a decrease in sediment and nutrient loading downstream (McMahon et al., 2021). Low flow rates can increase eutrophication risk as smaller quantities of water are unable to dilute salts and nutrients. In many cases, eutrophication occurs when excessive nutrients and warm, stagnant waters permit a bloom of plants and/or algae; when those organisms die, microbial decomposition can deplete oxygen availability and create anoxic zones (Smith et al., 1999). Harmful algal blooms have been shown to increase with climate change as rising temperatures create an environment for excessive growth in water systems (O'Neil et al., 2012).

Climate change has also significantly modified flow regimes. In general for snowpack dependent systems like the Western United States, smaller snowpacks melt earlier in the year, shifting peak flows earlier, which could deplete water supply later in the summer when crop and animal demand is highest (Perry et al., 2012). Climate change has also been linked to an increase in flooding events. Changes in water availability throughout the Western US require restoration efforts that target wetland and riparian zones associated with intermittent streams. Beaver Dam Analogs (Beaver dam analogs) are a potential restoration solution to increase and expand the duration of flow within drylands and mitigate negative impacts like flooding and erosion.

1.3 Beaver Dams and Beaver Dam Analogs as a Restoration Technique

Natural beaver dams have shown promise in gradually converting degraded rangeland to more productive wetlands (Law et al., 2017; L. M. Norman et al., 2022). While ecosystem responses may take multiple years to manifest, some of the more immediate effects of natural beaver dams can include improving bank stability and reducing erosion (Curran & Cannatelli, 2014) by increasing sediment retention and decreasing stream velocity (Meentemeyer & Butler, 2013). In contrast, beaver dams can increase bank erosion causing sediment deposition and reduction in stream channel incision causing an increase in floodplain connectivity (Pollock et al., 2014). Beaver dams may also mitigate pollution by increasing sediment storage and altering in-stream cycling of anthropogenically-sourced carbon, nitrogen, and phosphorus (Andersen & Shafroth, 2010). Increases in water storage both above- and belowground may also help increase riparian biomass and soil water content, improving stream function. Figure 1 shows a conceptual model of the ecosystem impact of beaver dam analog installation.

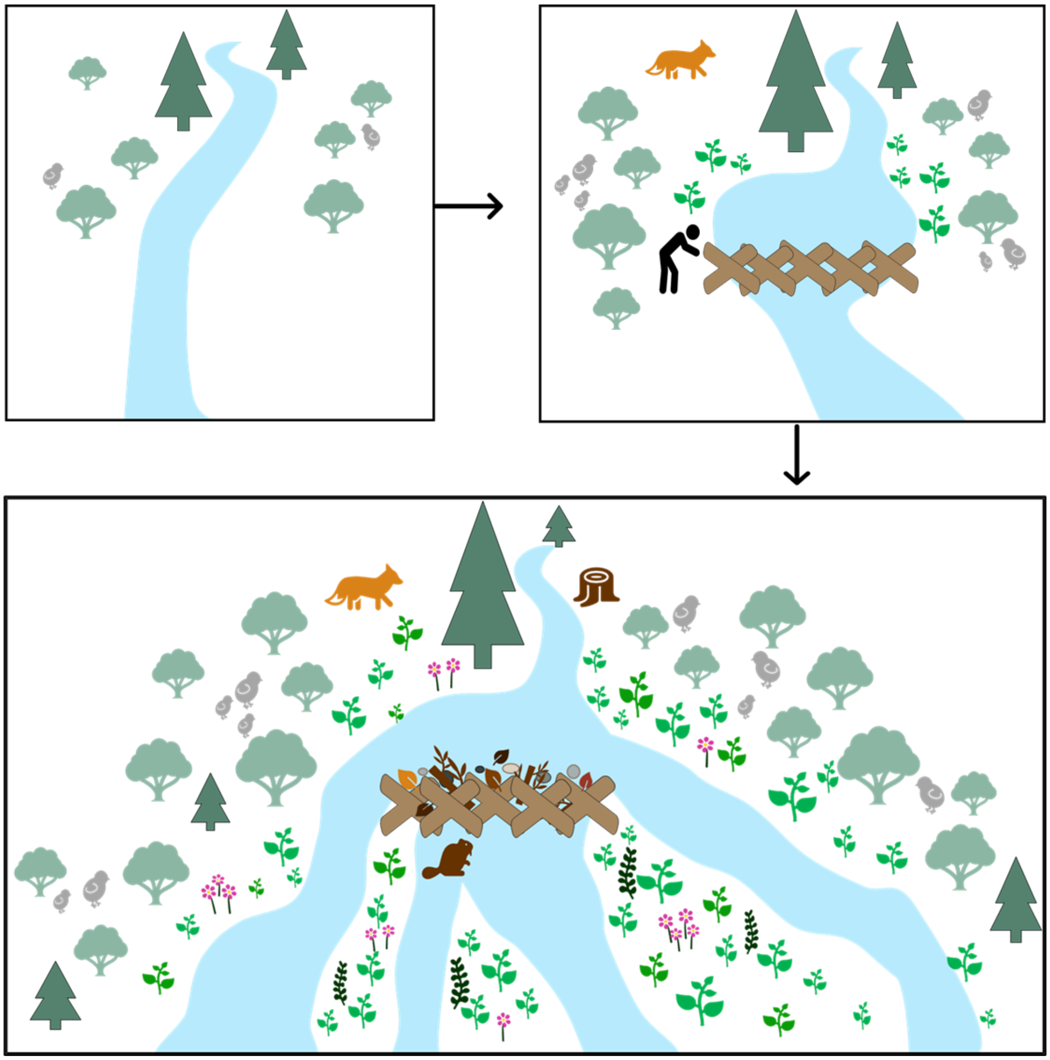


Figure 1. Conceptual model of potential BDA impact on a riparian ecosystem. Increase in surface area of water reconnects stream to floodplain leading to increases in water and nutrient availability for vegetation, specifically forbs and riparian obligates. Increase in plant biomass and diversity creates a thriving ecosystem.

Historic removal of beaver dams has greatly altered geomorphic settings within the US. The removal of beaver dams can convert multithreaded streams to those with a single channel of flow, which have relatively higher water velocity and lower sediment retention (Green & Westbrook, 2009). Beaver dam analogs have the potential to reverse stream channelization because they create conditions that mimic natural beaver meadows (Bouwes et al., 2016; Orr et al., 2020). Beaver dam analogs are of particular interest because they are relatively cheap and easy to install; if materials are sourced on or near the restoration site (e.g., willow boughs), the only major cost is labor to install and maintain them (Wheaton et al., 2019).

Chapter 2: Testing the Effect of Beaver Dam Analogs on Soil Health and Water quality

2.1 Introduction

Beaver dam analogs are being installed across the Western US in an attempt to recreate the ecosystem services provided by natural beaver meadows/complexes (Pollock et al., 2014). Beaver meadows are stream sections that have multiple beaver dams creating a low gradient multichannel stream (Polvi & Wohl, 2012). Beaver dam analogs and other structures that mimic natural beaver dams have been shown to store additional water within a stream. By gradually elevating the water table, these dams can increase soil moisture and plant-available water (Silverman et al., 2019). Dams, both natural and artificial, can also increase channel-floodplain connectivity, which in turn increases fine sediment deposit and nutrient storage (Polvi & Wohl, 2013). Beaver dam analogs slow the movement of water, which may increase surface water availability for cattle and wildlife, promote groundwater recharge, and reduce the loading of anthropogenically-sourced pollutants by stimulating microbial metabolism (Andersen & Shafroth, 2010). Beaver dam analogs may also reduce wildfire risk, which is becoming increasingly prevalent in the western US (Fairfax & Whittle, 2020). There is promise that dryland systems with intermittent streams could be transitioned into systems with year-round water supply if beaver dam analogs are properly installed (Pollock et al., 2003).

Beaver dam analogs can increase stream-floodplain connectivity by providing additional surface water storage and pooling water and nutrients above dam structures (Demmer & Beschta, 2008). Beaver dam analogs have been shown to create pooling within just one year of installation (Scamardo & Wohl, 2020), which could increase soil water content and nutrient exchange among rewetted soil pores (Morillas et al., 2015). Rapid increases in plant abundance and diversity suggest beaver dam analogs could also improve forage quality (Silverman et al., 2019) but minor changes in groundwater storage suggests it will take far longer to create extensive ecosystem change.

Beaver dams may increase hyporheic flow and the residence time of water within a stream system (Larsen et al., 2020, Norman, 2020, & Janzen & Westbrook, 2011). Hyporheic flow occurs when groundwater and surface water mix through streambed sediments and is linked with stream morphology (Orghidan, 1959). Differences in streambed geomorphology are a controlling factor for hyporheic exchange, for example streams that are incised have less surface area for groundwater and surface water mixing (Daniele Tonina & Buffington, 2009). Additionally, finer grain sizes in the stream channel and decreased surface water availability reduce the amount of hyporheic exchange (Dole-Olivier et al., 2022). Both beaver dams and beaver dam analogs have been found to increase hyporheic exchange through replenishing surface water and increasing surface water residence times (Janzen & Westbrook, 2011; Newbold et al., 1983; Wade et al., 2020). Biogeochemical processes (e.g., cycling of dissolved organic carbon and nutrients) are facilitated in hyporheic zones because the pooling of water and sediment aggradation allows for dissolved solutes and riparian microbial communities to interact (Wagner & Beisser, 2005). Similarly, beaver dams can trap organic matter in accumulating sediment deposits, which could increase watershed carbon storage and increase the availability of nutrients for uptake by riparian vegetation and microbial communities (Sutfin et al., 2016).

Figure 2 shows a conceptual model of the impact on peak flow and water storage after beaver dam analog installation. Natural beaver dams attenuate flows, which can reduce peak discharge following spring snowmelt and storm events and increase water storage both above- and belowground (Puttock et al., 2021). Additionally, groups of dams (and beaver meadows) appear to impact channel morphology more than individual structures (Green & Westbrook, 2009). Similar impacts have been found with beaver dam analogs, where the installation of multiple structures (e.g., 2 or 3) led to greater water table rise and surface-water pooling than sites with a single dam (Munir & Westbrook, 2021). Beaver dam analogs have the potential to increase water storage and mitigate peak flows in streams, but it is also important to note that beaver dam analogs are not a panacea for all dryland systems.

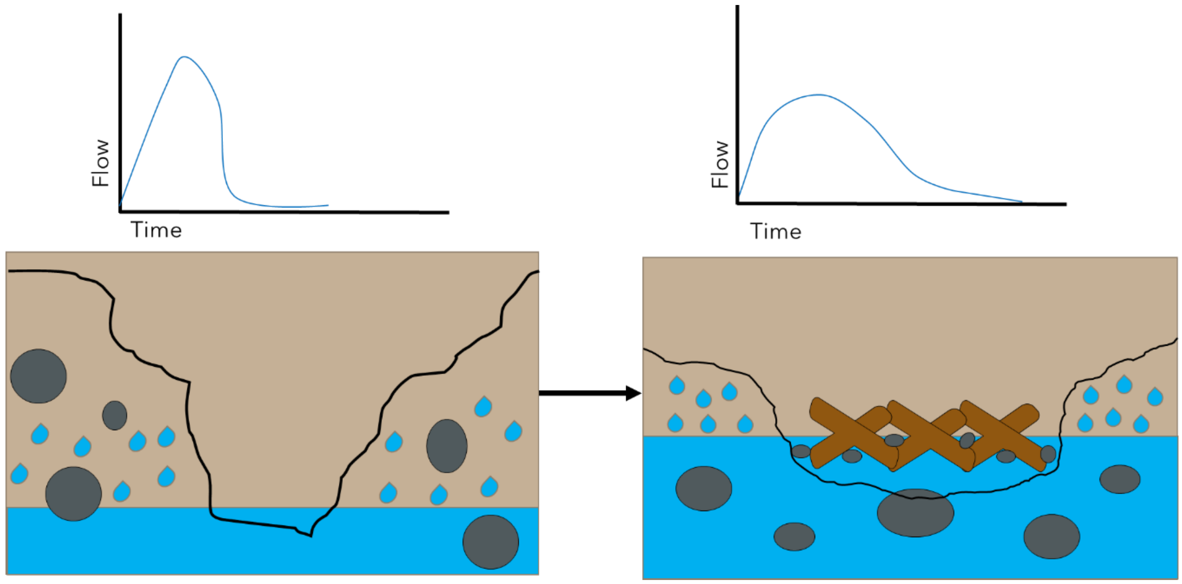


Figure 2. Conceptual model of BDA impact from a cross sectional point of view, in-stream structures slow water allowing for increases in surface water availability, reducing peak flow, and decreasing depth to groundwater. Water availability and soil moisture increases support above and below ground biomass. Water droplets represent the amount of water present in soil, soil moisture increases with groundwater and stream water increase.

Studies examining the effect of beaver dam analogs on stream function are mixed, which is important to consider as stream alteration can have varying impacts. For example, some studies have found that beaver dams decrease surface water temperatures (Dittbrenner et al., 2022) while others have documented significant increases (Majerova et al., 2015). Regional climate conditions likely play a significant role in mediating this effect. For example, both studies were conducted in mountainous regions, but significant precipitation inputs could reduce surface water temperatures, as observed by Dittbrenner et al., (2022) in Washington, while anomalously high temperatures could increase surface water temperatures, particularly in stagnant pools, as observed by Majerova et al., (2015) in Utah. Increases in surface water temperatures (Majerova et al., 2015) elevate the risk of eutrophication. Especially in a rangeland setting with high nutrient inputs near and at riparian sites, the combination of high temperatures and nutrient loading could be of particular concern. Eutrophication can be toxic to wildlife and aquatic organisms. The lack of oxygen within the stream can kill aquatic organisms while toxins released by cyanobacteria can kill large mammals (Le Moal et al., 2019). There is not a clear correlation between beaver dams and eutrophication. For example, some studies have shown that beaver dams can improve water quality and nutrient filtration (Maret et al., 1987) while others have found increases in nutrient loading and downstream export (Hammerson, 1994). Overall, beaver dam analogs seem to have less impact on surface water temperatures than natural beaver meadows (Orr et al., 2020; Pearce et al., 2021a), particularly when considering low-gradient channels located in semi-arid drylands (conditions similar to our study sites at Rinker Rock Creek Ranch). Research on beaver dam analogs is still relatively limited, which may be another reason there are limited reports on how they influence surface water temperatures.

Table 1 articulates potential positive and negative impacts BDA installation may have on arid lands. Along with uncertainty in channel temperatures, beaver dam analogs could alter channel morphology and critical habitat. For example, many fish species rely on specific habitat conditions, including the size and depth of streambed sediment, stream velocity, temperature, and water quantity. Interestingly, beaver dam analogs have been used to restore fish habitats because they can increase surface water storage without impeding fish navigation through and around the structures (Davee et al., 2019). It is still important to consider how changes in sediment size distribution, water temperatures, or other environmental shifts will influence sensitive fish populations. For example, pooling water around beaver dam analogs could increase local water storage, but reduce downstream flows, unless groundwater storage is recharged. However, in arid land streams in Montana, beaver dam analogs attenuated surface water flows by increasing groundwater storage, successfully mimicking natural beaver dams in extending surface water supply through the dry season (E. G. Norman, 2020). This should be seen as a great benefit to natural systems that lack year-round water supply. In areas where irrigation occurs, timing of water supply could be an issue to fulfill water rights. Changing in-stream structures which manipulate this timing could therefore pose legal risk for any streams where water is allocated through prior appropriation (Cobourn et al., 2022; Xu et al., 2014).

Table 1. Comparison of Beaver Dam Analog impacts on a local stream system. Information gathered from Pilliod et al., 2018.

Potential pros	Potential Cons
• Low cost	• Stream temperature increase
• Easy Installation	• Fish habitat alteration
• Forage quality increase	• Local nutrient loading increase
• Water filtration	• Reduction in downstream water availability
• Increased hyporheic exchange	• Water Rights - Legal Risks
• Microbial biomass increase	
• Carbon storage increase	

Beaver dam analogs could serve as a low-technology, cost-effective means of increasing carbon stocks and sediment storage and improving water quality. There is thus a clear need for more extensive research in rangeland systems to understand how beaver dam analogs could impact ecosystem services and grazing outcomes. For example, while natural beaver dams can increase carbon and sediment storage on a watershed scale (Scamardo & Wohl, 2020) it remains unclear whether beaver dam analogs will similarly impact riparian biogeochemistry.

For my MS research, I studied how BDA installation on an intermittent stream influences water quality and soil biogeochemistry. The purpose of my project was to assess the environmental impact of beaver dam analogs on dryland productivity. The study took place at Rinker Rock Creek Ranch, a University of Idaho research facility. Rinker Rock Creek Ranch has four tributaries within actively managed rangeland, including Guy Canyon, which was selected for BDA-restoration in 2022. I collected baseline measurements in 2021 (pre-installation) and continued monitoring and assessment activities in 2022 (one-year post-restoration). Unlike many BDA studies, my research captured both pre- and post-restoration conditions, allowing me to disentangle potential effects of BDA installation on ecosystem function. Specifically, I used this study to answer two important and interrelated questions. (1) Does BDA installation alter water quality? (2) Does BDA installation increase soil carbon sequestration? Based on these research questions, I constructed three hypotheses:

Hypothesis 1 Beaver dam analogs increase organic carbon and nutrient concentrations by decreasing stream velocity and re-establishing stream-floodplain connectivity.

Hypothesis 2 Beaver dam analogs trap water and sediment in pools upstream of each structure, which increases surface water temperatures and decreases oxygen availability.

Hypothesis 3 Beaver dam analogs increase soil moisture by pushing surface water and entrained nutrients laterally into the adjacent floodplain. More favorable growing conditions increase soil microbial biomass, which may eventually lead to greater soil organic carbon sequestration.

To address these questions, I monitored monthly changes in dissolved oxygen, temperature, pH, and total dissolved solid concentrations at multiple locations within the stream channel, capturing changes in flow velocity from spring snowmelt to summer low flow conditions. I also collected paired surface water samples to quantify nitrate, ammonium, and phosphate concentrations. Finally, I collected soil samples annually at three distances from the stream channel and measured changes in soil organic carbon and nitrogen stocks, dissolved organic carbon and nitrogen pools, microbial biomass, and nutrient availability.

2.2 Methods

2.2.1 Site Description

Guy Canyon stream flows through the University of Idaho Rinker Rock Creek Ranch in central Idaho. Rinker Rock Creek Ranch is an active rangeland site managed with rotational grazing to prevent excessive damage to vegetation and soils (Milchunas & Lauenroth, 1993; Teague et al., 2011). For this study, I divided the upper reach of Guy Canyon into four study meadows, with one reference control meadow located upstream of three BDA treatment meadows. Baseline data were collected for one year prior to BDA installation (July 2021). During baseline and post-installation measurements, I tracked stream temperature, pH, total dissolved solids (TDS), and nutrient concentrations (e.g., nitrate, phosphate, orthophosphate, carbon) in surface water samples. I also tested how potential floodplain inundation influenced soil moisture, microbial growth, and soil nutrient availability.

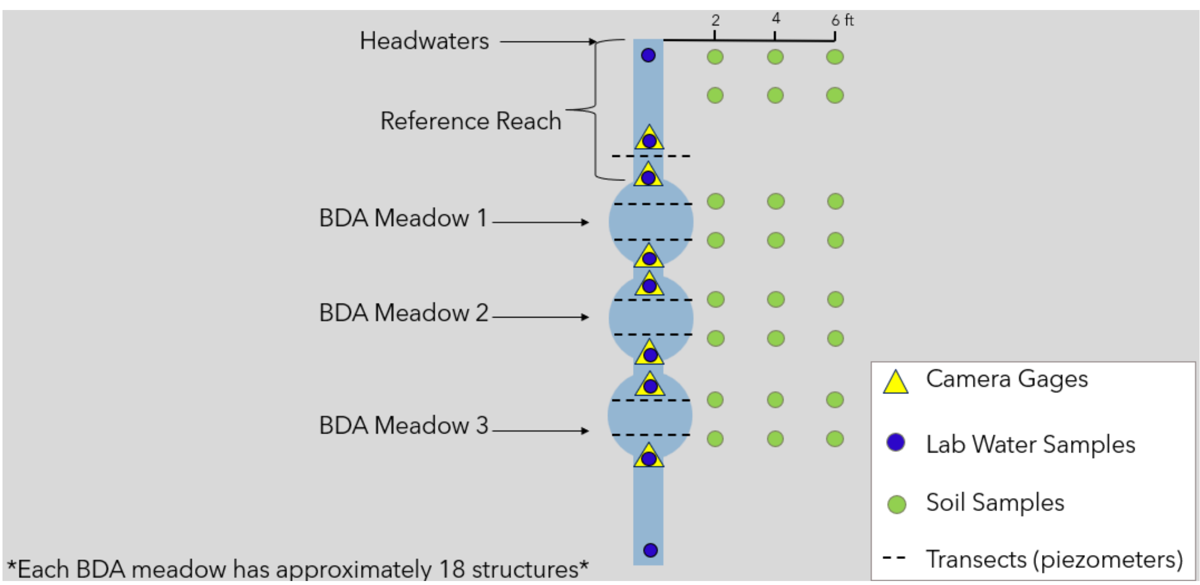


Figure 3. Experimental design. A reference control meadow was established upstream of three treatment meadows, where 18, 20, & 21 beaver dam analogs (respectively in meadows 1, 2, &3) were deployed in tandem within each meadow. The location of camera gages (yellow triangles), surface-water grab samples for nutrient analysis (blue circles) and soil samples (green samples) are depicted. Piezometer transects (dashed lines) were used to monitor groundwater levels.

Figure 3 shows a general experimental design for sampling and meadow locations. Cameras were placed at the beginning and end of each meadow and in the reference stream. Cameras captured flow measurements on meter sticks four times per day to account for the amount of water flowing into and out of each meadow complex. Traditional flow meters could not be used because the stream channel was too narrow and water levels were too shallow (e.g., for most of the spring and summer, flow rates were ≤ 1 cubic foot per second). We also tried using salt-tracer methods (e.g., monitoring changes in stream water conductivity with an in=-stream probe and calculating flow using breakthrough curve analysis), but this method requires enough water for the probe to be in the stream and a straight section of ~20 feet of stream channel. These conditions were met at meadow 2 once in 2021, but were not met in meadow 1, meadow 3, or the reference reach, so this was not an optimal approach to collect flow data in each location. Cameras allowed for more accurate and comprehensive data collection throughout the summer. These stage values were correlated to direct flume measurements taken once each month.

Two shallow groundwater piezometer transects were also deployed in the reference meadow and each treatment meadow. Piezometer transects were used to estimate seasonal changes in groundwater level for another study. For our study, we used the transects as a marker for collecting soil samples in consistent locations each year at one upstream and downstream location within each meadow complex. Water samples were collected at the beginning and end of each meadow (at the camera locations) and at each piezometer transect (when water was present).

The upper drainage area of Guy Canyon is characterized as semi-arid rangeland (Figure 4). Big Sagebrush (*Artemisia tridentata*) dominates the upland vegetation community. Riparian areas are primarily colonized by willows, grasses (especially Sandberg bluegrass/*Poa secunda sandbergii*, bluebunch wheatgrass/*Pseudoroegneria spicata*, Idaho fescue/*Festuca idahoensis,* and cheatgrass/*Bromus tectorum*), and sedges (especially elk sedge/C*arex garberi*, Nebraska sedge/C*arex nebrascensis* , expressway sedge/C*arex nebrascensis*, and Douglas sedge/C*arex douglasii*), interspersed with horsetail sage wort/A*rtemisia frigida*, wild geranium/*Geranium maculatum*, wild rose/R*osa acicularis*, and mountain mahogany/C*ercocarpus ledifolium* (Hankins, 2001; Hironaka et al., 1983). The basin elevation ranges from 5,000 ft to 5,200 ft. Summers are hot and dry, with most of the precipitation falling as snow in the winter (mean annual temperature = 43ºF, mean annual precipitation = 15.9 inches, mean snow = 80 inches) (*Hailey, ID Climate Summary*, n.d.). Stream water is used to supply water for cattle on the Rinker Rock Creek Ranch. The stream ranges from vertical incision of 5 ft to a depth of a half-foot in other locations with the most incised sections located at the beginning of meadow 1, this was measured through ArcGIS imagery. Stream width also varied from ~2 to 5 ft wide. The stream only consisted of one main channel but had seepage and some additional inputs in meadow 3. Figure 4 shows stream variability with some sections entirely incised and disconnected from the floodplain with no flow, while others have flowing water during the same time and are surrounded by abundant vegetation.

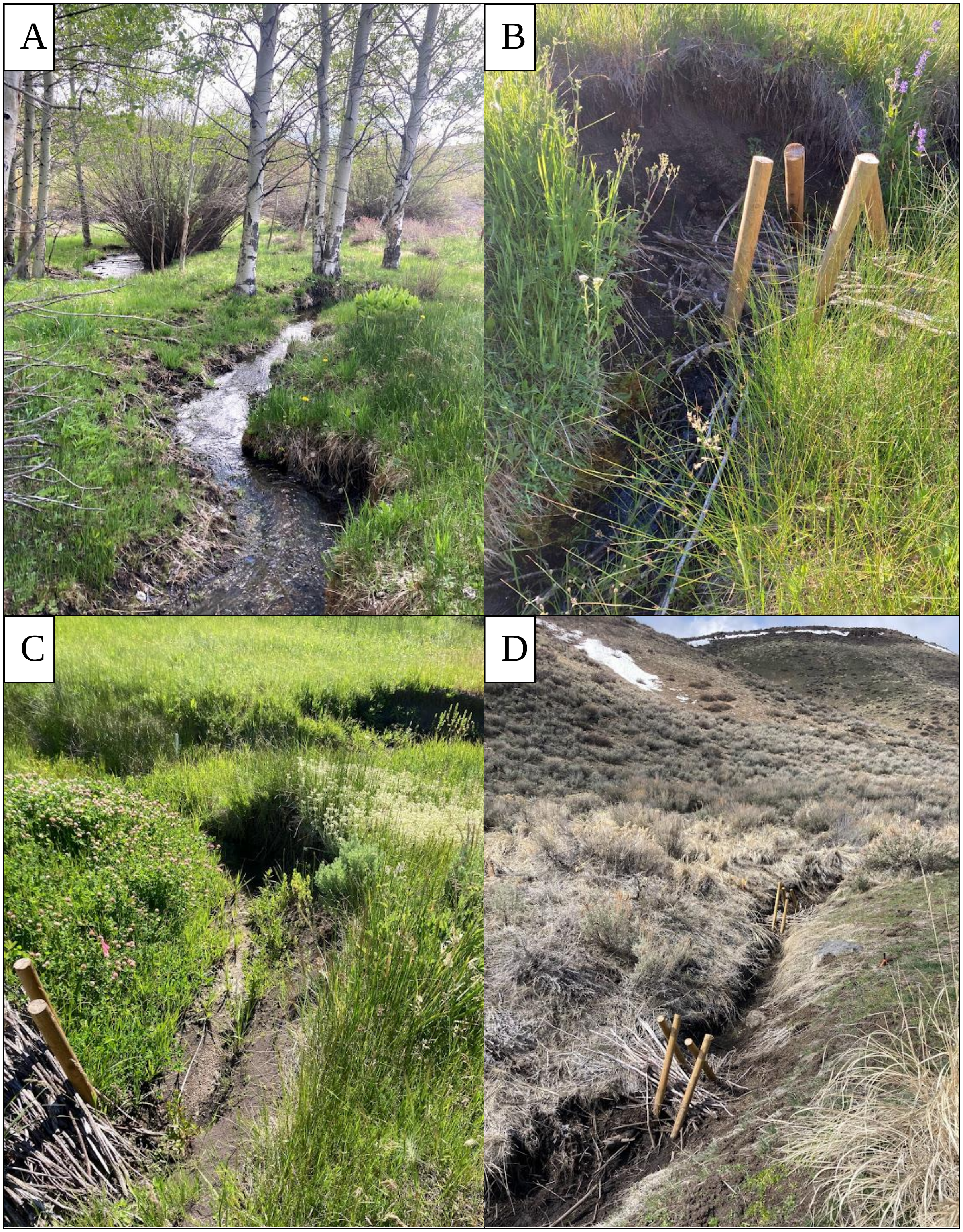
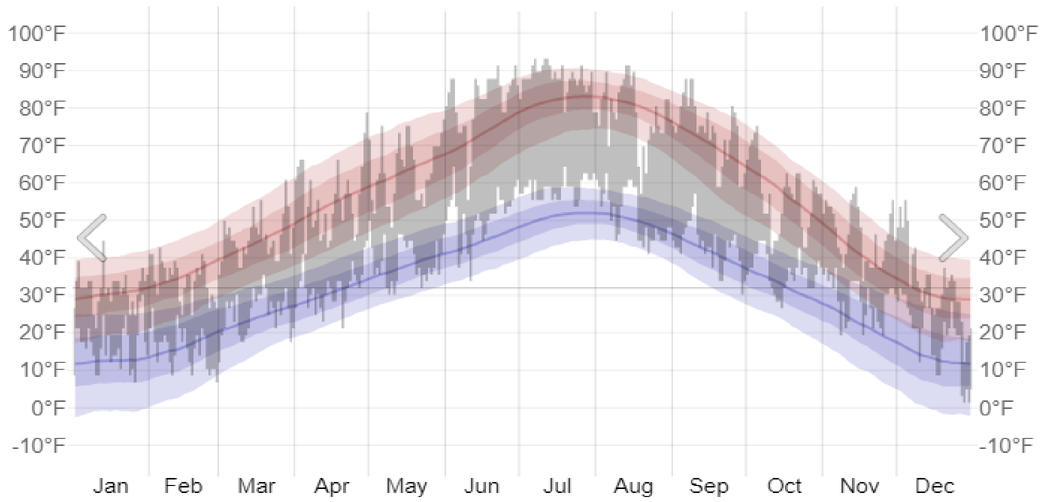


Figure 4. Representative photos of Rinker Rock treatment meadows, including Meadow 2 in May 2021 (A), Meadow 1 July 2021 (B), Meadow 3 July 2021 (C), and Meadow 1 in April 2022 (D).

Hailey, Idaho Temperatures 2021



Hailey, Idaho Temperatures 2022

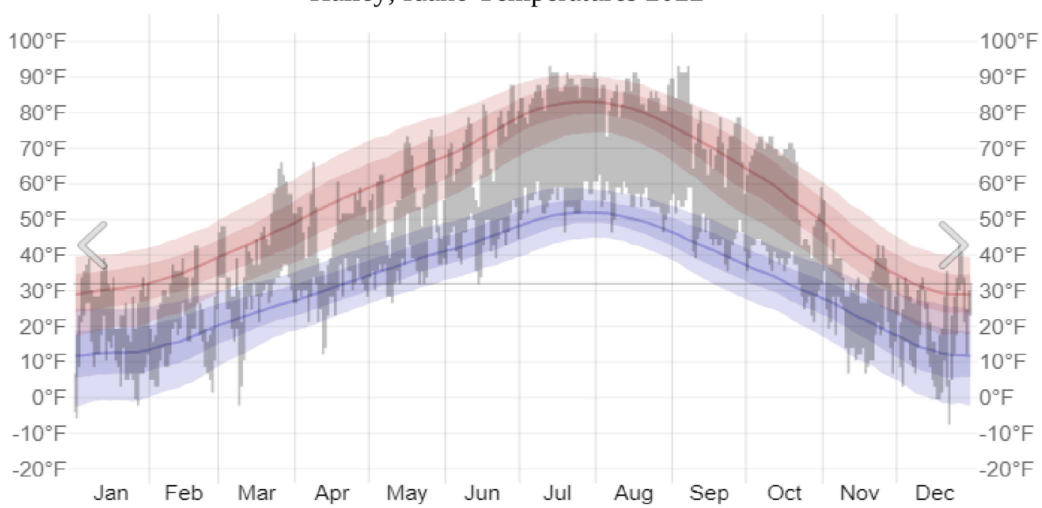


Figure 5. Hailey, Idaho Temperatures in Fahrenheit in 2021 and 2022 from https://weatherspark.com

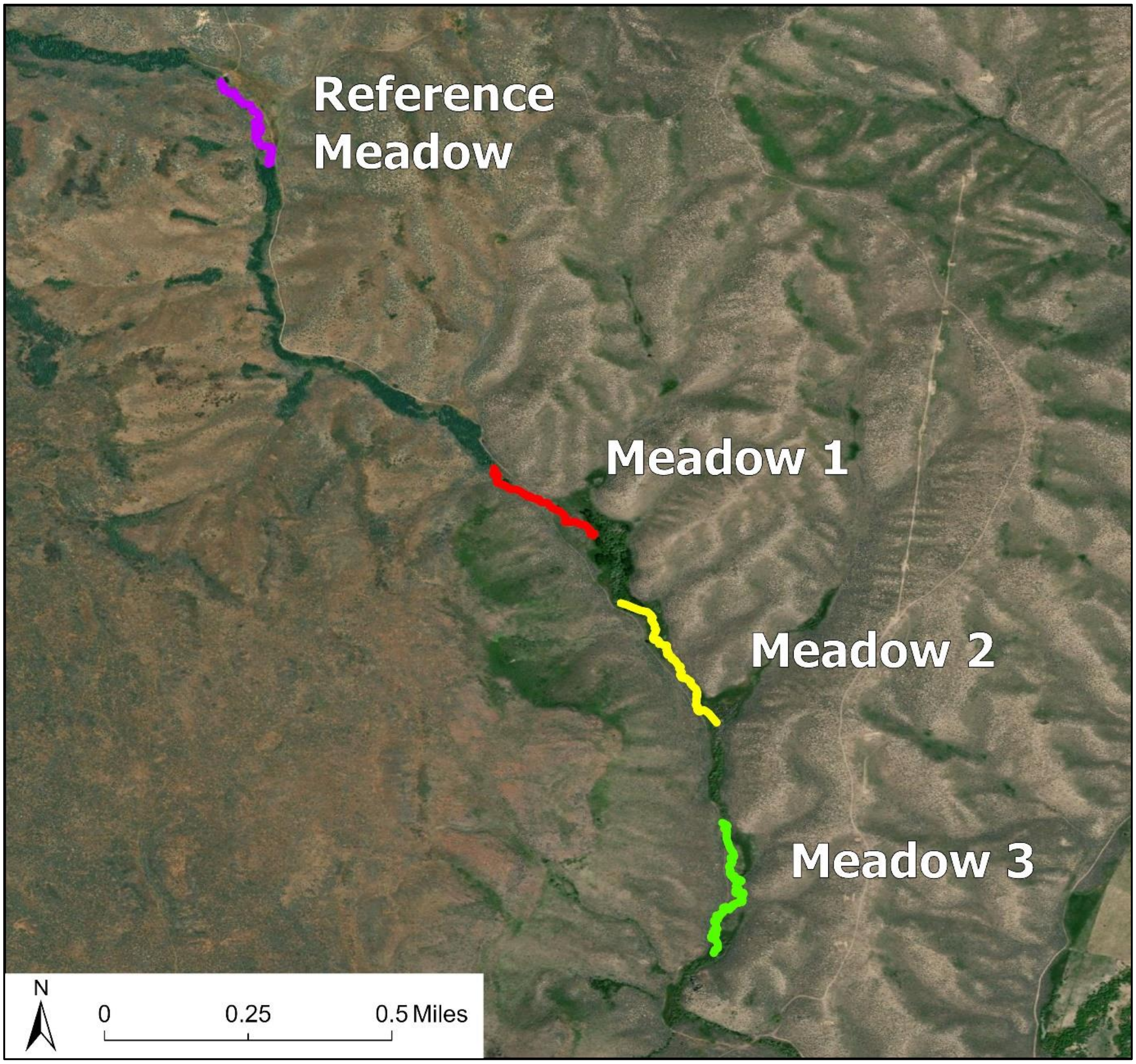


Figure 6. Location of Rinker Rock Creek Ranch near Bellevue, ID. The reference meadow (purple) is located upstream of three treatment meadows (shown in red, yellow, and green); Guy Canyon flows from North to South.

The channel contains both losing and gaining reaches, including a strong losing reach at the bottom of meadow 1 that completely disconnects flow between meadows 1 and 2 for most of the growing season. We began collecting stream measurements in May 2021. During our first sampling period, Guy Canyon was flowing through ~80% of the study area; meadow 1 was already dry except for standing water at the very top. In June, the stream was flowing in 50% of the study area; meadow 1 and the bottom half of meadow 3 were dry. By July, 43% of the study area contained flowing or stagnant water, which was confined to meadow 2 and the top of meadow 3. By September, we observed no flowing water; wetted reaches were confined to meadow 2.

We observed similar trends in 2022, but sampling began in April allowing us to collect surface water samples just after peak snowmelt, when 90% of the study site was wetted. Despite sampling directly after snowmelt-induced peak flows, the bottom section of meadow 1 was already dry. In May, we sampled immediately after a storm event, which reconnected the four meadows with continuous flow. In June, rising temperatures and drought conditions reduced surface water availability; meadow 1 disconnected from the other two meadows and only 85% of the study area had water. By July of 2022, 44% of the study area was wetted; nearly all of meadow 1 and the bottom half of meadow 3 were completely dry.

2.2.2 BDA Installation

In July of 2021, a total of 59 beaver dam analogs were installed throughout the 3 treatment meadows, with 18 in meadow 1, 20 in meadow 2, and 21 in meadow 3. Most of the structures are categorized as Post-Assisted Log Structures (PALS, which include channel-spanning beaver dam analogs and partial channel-spanning structures) (*Figure* 7). Of these, five contained rocks and are thus considered 'Zeedyk' structures (Silverman et al., 2019), although posts and debris were still utilized like the other post-assisted log structures.

Figure 7. An Example BDA in Meadow 2 in April 2022

2.2.3 Water Quality

Water quality samples were collected every month from May - September 2021 and April - July 2022. Sampling sites were located within the reference reach (control) and at the start of each meadow (i.e., above the first BDA), at each piezometer site (two per meadow), and at the end of each meadow. Occasionally, due to the intermittent nature of this stream, samples were collected where water was present inside of the meadows. At each sampling location, we collected two water samples, one for dissolved organic carbon and total dissolved nitrogen analysis and the other to quantify pools of nitrate, ammonium, and phosphate (NO_3^-, NH_4^+, PO_4^3, respectively). Sixty mL of water were collected via sterile plastic syringe and immediately filtered through sterile 0.45 μm glass fiber filters (Whatman) into sterile amber borosilicate vials fitted with Teflon lids. Samples for total organic carbon and total nitrogen analysis were preserved by adding two drops of 98% sulfuric acid to each 60 mL sample, which stops all biological activity and any biogeochemical transformations. All glass vials, used for water sampling, were pre-acid washed to remove inorganic contaminants and combusted for four hours at 450 °C to remove organics. Surface water samples were stored in a cooler with ice for < 24 hours and then transferred to the freezer until instrumental analysis (described below).

In-stream water data were collected using a Hanna Instrument® HI98194 Multiparameter Meter to collect pH, temperature, dissolved oxygen, and total dissolved solids. The multiparameter meter was calibrated using Hanna Instruments® standard solutions for each variable at the beginning of each sampling period. The probe was placed in the stream vertically; measurement values were allowed to stabilize for a minimum of one minute per sample site.

Water flow data was collected using Wingscapes® TimelapseCam Pro Camera and a meter stick mounted in stream parallel to stream flow. Pictures were collected daily at 9 AM, 12PM, 3 PM, and 6 PM for the duration of the sampling period. Flow was then quantified once per month using a USGS Portable Parshall Flume, 3" Fiberglass to measure in-stream flow (Open Channel Flow, Boise, Idaho, USA). The flume had an inlet opening of 10" and handled flow rates up to 835 gallons per minute. Flow data were used to create a linear relationship between stream height (camera) and discharge (flume) using R software. Flow rates were then predicted for each camera measurement.

2.2.4 Soil Health

Soil samples were collected to a depth of 20 cm using an 8 cm diameter soil corer. Soil samples were collected 2, 4, and 6-feet from the edge of the stream channel and paired with piezometer transects installed in each meadow to determine microbial biomass, pH, soil moisture content, soil carbon and nitrogen stocks, and nutrient concentrations (nitrate, ammonium, and orthophosphate). As each treatment meadow had two piezometer transects (n = 3 piezometers per transect), 24 soil samples were collected each year. Each soil sample was put in a sterile Ziploc® bag and transported to the laboratory in a cooler with ice. Soils were sieved to < 2 mm and all roots and rocks were removed. In total, 30 g of wet-weight soil was used to determine soil moisture; the remaining processed soil was placed in a fresh plastic bag and stored at -20 °C until analysis.

Soil moisture was determined using the protocol from Lynch et al., 2018. Briefly, 10 g of processed soil was placed into pre-weighed aluminum tins. Each sample was weighed in triplicate and dried in an oven at 60 °C. Samples were weighed until no change in weight was detected (about three days). Percent moisture was determined by subtracting the dry weight from the initial weight (after subtracting the tin weight).

Soil microbial biomass was determined using a modified method (Lynch et al., 2018). Unfumigated control soil samples were extracted using 0.5 M potassium sulfate and agitated on a shaker table for four hours. Approximately 2 ml of ethanol-free chloroform was evenly distributed over fumigated treatment soils. Glass flasks were capped with rubber stoppers wrapped in aluminum foil (to prevent carbon leaching) and incubated for 24 hours before being extracted as above. Both fumigated and unfumigated extracts were filtered through No. 1 Whatman paper. All samples were bubbled for 1-hour, fumigated samples were bubbled under the hood to remove remaining chloroform. Extracts were then analyzed for total organic carbon and total dissolved nitrogen using a Shimadzu® TOC-L/TN Auto-Analyzer. Microbial biomass carbon and nitrogen concentrations were calculated as the difference between the chloroform-fumigated and non-fumigated soil extracts.

Concentrations of nitrate, ammonium, and orthophosphate were quantified using a SpectraMax® 340 PC 384 Absorbance Microplate Reader and sterile flat bottomed and clear 96-well microplates. For nitrate analysis, standard colorimetric methods were used following the protocol from Doane & Horwáth, 2003. Standards used for surface water analysis ranged from 0 ppm to 1 ppm nitrate, those for soils ranged from 0 ppm to 10 ppm. Ammonium analysis followed the protocol from Weatherburn, 1967 with water sample standards from 0 to 2 ppm and soil extracts from 0 to 10 ppm. Standard colorimetric protocol for orthophosphate analysis was done using methods from Lajtha et al., 1999 with 0 to 1 ppm standards used for both water and soil extract samples.

Dried soil samples were prepared for elemental analysis by grinding samples using a sterilized ball mill for four minutes each. Once samples were fully homogenized, ~ 15 mg of milled sample was weighed into 4x6 mm tins. We determined %C, $\delta^{13}C$, %N, and $\delta^{15}N$ with a Carlo Erba NA 1500 elemental analyzer (CE Instruments, Lancanshire, UK) coupled to a VG Isochrom continuous flow isotope ratio mass spectrometer (Isoprime, Inc., Manchester, UK).

2.2.5 Data Analysis and Statistics

All collected field and test lab data were organized in an Excel .csv file and imported to R version 4.2.2 for statistical analysis. For water field and nutrient data using a linear mixed effect model, including meadow and sampling month as fixed effects, and the location within each meadow from which a sample was collected, as the random effect. This model allowed for testing of statistical differences across each month (and changes in flow) and/or within or across individual meadows. The general form of the model was:

(eq. 1) model = lme(Parameter ~ Meadow * Month, random = ~ 1 | Meadow Position, data = water)

Equation 2 was used to examine potential environmental drivers on water quality, I used multiple linear regressions to assess the relationship between the environmental factors (e.g., discharge, valley confinement, temperature, precipitation) on surface water quality (e.g., concentrations of nitrate, ammonium, orthophosphate, dissolved organic carbon and nitrogen, etc.). The general form of the model was:

(eq. 2) model=lm(Y ~ flow + $\beta_1\chi_{i1} + \beta_2\chi_{i2} + \ldots + \varepsilon_{ij}$, data = water_2022)

For soil data, a linear mixed model was used to compare 2021 biogeochemical parameters to 2022 soil samples to look at BDA influence. The fixed effects included meadow, distance from stream (2-ft, 4-ft, 6-ft from edge of channel), and year. The random effect was specified as the meadow position where the sample was located. The meadow positions included upper and lower sections in each meadow including the reference site. The following code was used:

(eq. 3) model=lmer(Y ~ Meadow * Distance from Channel * Year + (1|Meadow Position), data=soil)

2.3 Results

2.3.2 Water Quality Indicators by Treatment Location and Month

In general, surface water temperatures were lowest in April (mean = 8.46 °C) and increased throughout the growing season (Table 2). In 2021, average water temperatures at the time of sampling did not differ across meadows. According to t-test results, in 2022 water temperatures were significantly higher in June than July ($p < 0.001$). Dissolved oxygen (DO) concentrations also remained consistent across sampling periods in 2021 (mean DO = 6.82 ppm). In 2022, DO concentrations increased by 11.19% from April to May ($p < 0.001$), declined by 28.84% from May to June ($p < 0.001$), and remained consistent from June to July (mean DO = 7.06 ppm). We observed no statistical differences in the loading of total dissolved solids (TDS) in 2021 (mean TDS = 123.45 ppm). In contrast, TDS trends were variable and inversely related to DO in 2022; concentrations declined by 2% during peak runoff (April to May, $p = 0.03$), increased by 19% from May to June ($p < 0.001$), and increased by another 19% from June to July ($p < 0.001$).

During each sampling period we collected surface water samples from the top and bottom of each meadow and at piezometer transects for more detailed chemical analysis. Due to a limited sample size, we used linear mixed effects regression models to test how water quality varied within and among reference and BDA treatment meadows in 2022.

The following results were found using equation 1. Nitrate concentrations did not differ significantly across meadow or month. In contrast, ammonium concentrations in July were 57.73% and 38.92% higher than in May or June, respectively ($F_{3,20}= 6.36$, $p < 0.001$; Table 2). Phosphate concentrations were also significantly higher in July than April, May, or June, with the largest increase (~ 5 mg/L) observed between June and July ($p < 0.0001$). pH values declined throughout the season from a high of 7.88 in April to a low of 7.31 in July (Table 2). We observed no significant main effects or interactions describing variability in DO or TDS levels. We observed no significant differences between pre- and post-BDA installation, suggesting beaver dam analogs exerted a less direct influence on surface water quality than prolonged drought. Cumulative precipitation was 8 inches in 2021 and 39 inches in 2022 (Water year 2021: October 2020-September 2021). Figure 8 shows the snow water equivalent (SWE) at a nearby SNOTEL site peaked at 21.6 inches in April of 2021 and 23.1

inches in May of 2022. The extreme difference in water availability makes it difficult to directly contrast pre- and post-BDA water quality conditions. As a result, our data are more appropriately viewed as an observation of BDA effects throughout a season rather than a direct comparison one year before and after installation.

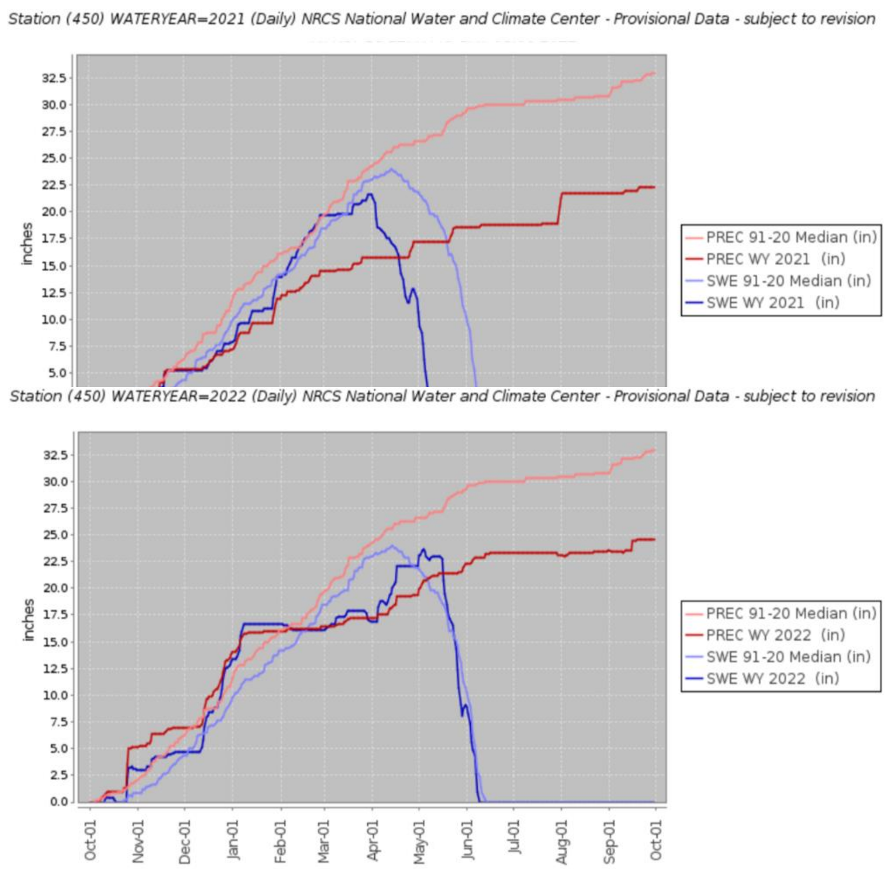


Figure 8. 2021 & 2022 SNOTEL site data from NRCS at a station near Guy Canyon at

“Dollarhide Summit (8,420 ft)”

Table 2. Water Nutrient values and hydrological parameters, where N = number of observations; DO = dissolved oxygen; TDS = total dissolved sediments; and the 7-day cumulative precipitation captures the amount of precipitation the basin received one-week prior to water sample collection.

Year	Month	N	pH	Temperature ◦C	DO (ppm)	TDS (ppm)	7-Day Cumulative Precipitation (in)
2021	May	14	7.74 (0.04)	14.41 (0.44)	6.32 (0.24)	125.09 (1.14)	0.09
	June	14	7.35 (0.09)	24.65 (1.18)	6.26 (0.29)	140.43 (4.52)	0
	July	14	7.43 (0.11)	23.39 (1.83)	6.38 (0.34)	135.33 (7.13)	0
	August	14	7.2 (0.1)	16.58 (0.96)	7.54 (0.4)	100.33 (2.41)	0.03
	September	14	8.15 (0.1)	9.53 (0.1)	6.99 (0.41)	114.67 (4.26)	0.17
	October	3	8.31 (0.19)	11.39 (0.85)	8.26 (0.59)	108.67 (17.32)	0
2022	April	64	7.92 (0.02)	8.85 (0.28)	8.67 (0.11)	110.78 (0.9)	0.07
	May	64	7.68 (0.03)	12.14 (0.12)	9.63 (0.23)	108.41 (0.53)	1.16
	June	64	7.9 (0.04)	15.79 (0.33)	6.86 (0.07)	128.71 (2.12)	0
	July	64	7.09 (0.05)	12.02 (0.19)	7.09 (0.19)	152.5 (2.3)	0.05

2.3.3 Water Quality Indicators and Environmental Drivers

The following results were found using equation 2. pH values were significantly influenced by stream discharge and valley confinement (adjusted r-squared = 0.28; $F_{1,42}$ = 9.63), where pH values were higher under high-flow conditions and in less confined valley bottoms. Dissolved oxygen (DO) concentrations were also related to stream discharge, precipitation, and temperature (adjusted r-squared = 0.64; $F_{1,43}$ = 26.7), where DO levels were higher during wet, high-flow conditions and declined with increasing temperature. Ammonium concentrations were negatively related to stream discharge (adjusted r-squared = 0.17; $F_{1,43}$ = 8.85). Phosphate concentrations were significantly influenced by stream discharge and valley confinement (adjusted r-squared = 0.21; $F_{1,42}$ = 6.81), where phosphate levels declined with discharge and channel confinement. TDN concentrations were higher after precipitation events and increased with water temperature (adjusted r-squared = 0.16 $F_{1,46}$ = 5.39). Dissolved organic carbon values marginally increased with surface water temperature (adjusted r-squared = 0.11; $F_{1,46}$ = 4.02 p = 0.06). TDS and nitrate concentrations were not related to any potential predictor variables. Stream discharge exerted a strong effect on multiple measured surface water parameters.

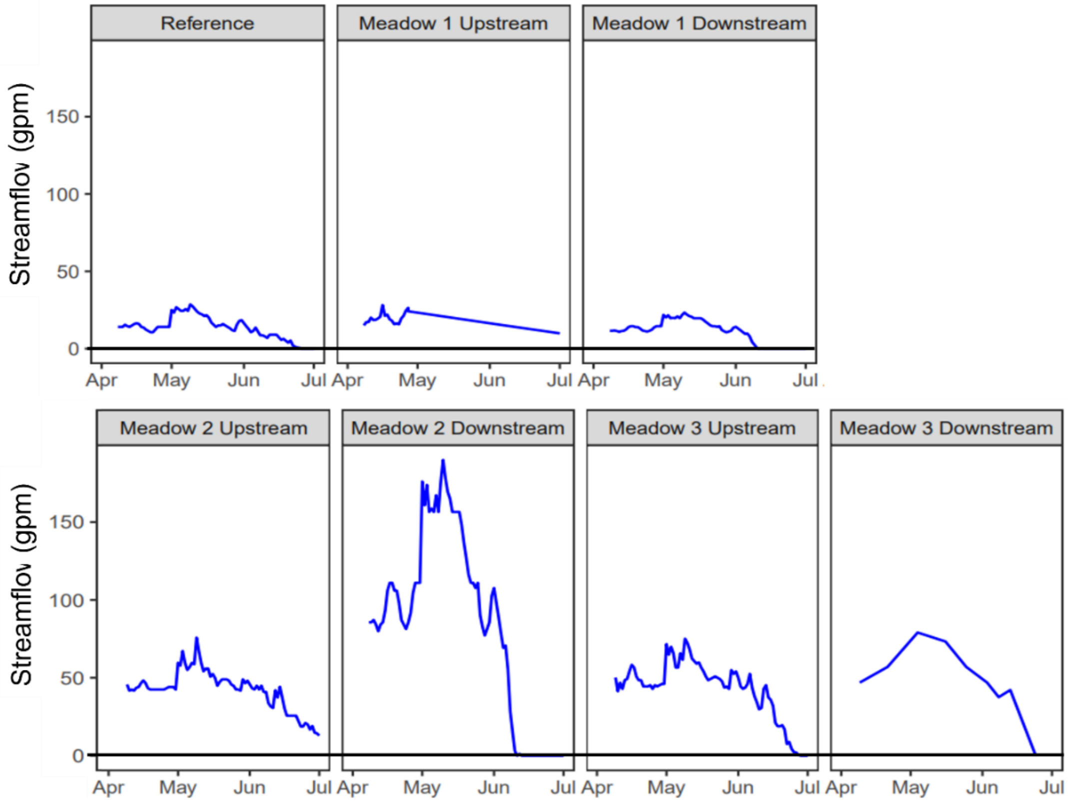


Figure 9. Discharge at beginning and end of each meadow as well as that the reference reach calculated by relating flume measurements in field to measurements on camera.

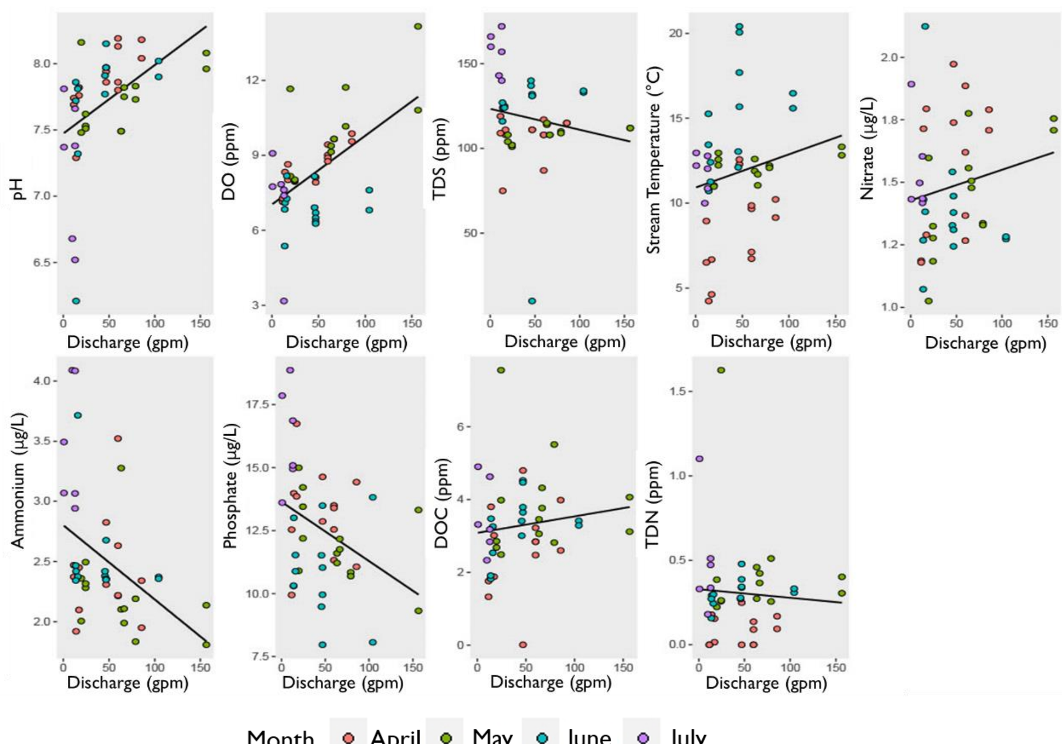


Figure 10. Plots rela

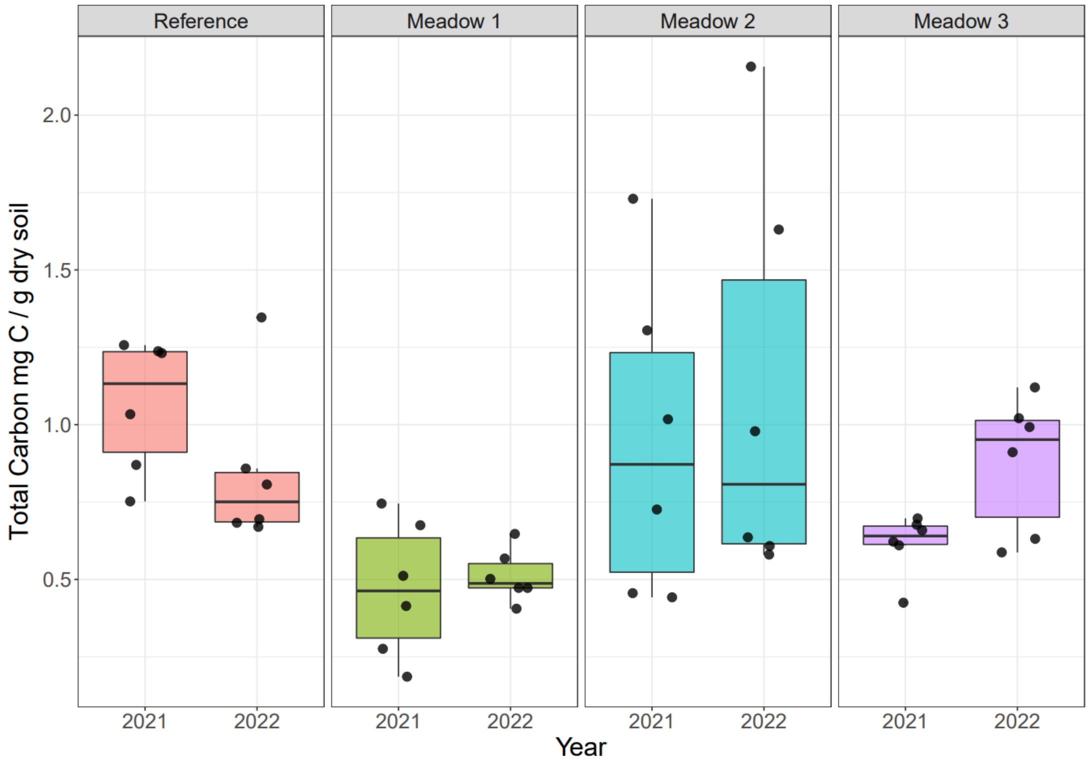
2.3.4 Soil Health Indicators

Soil organic carbon stock in soil increased significantly with distance from the stream channel ($F_{2,20}$= 8.24, p< 0.01). Nearest the channel, average soil organic carbon was 94.97 mg C/g soil, these levels increased by 34% from 2 to 4 feet and by 50% from 2 to 6 feet (Table 3). Soil nitrogen concentrations also increased with distance from the channel, increasing by 37% at 4 feet and 52% at 6 feet relative to an average concentration of 16.48 mg N/g soil at 2 feet ($F_{2,20}$= 11.28, p<0.001). For both soil organic carbon and total nitrogen, we observed marginally significant interactions between meadow and year (Table 3; p = 0.08 and p =0.09, respectively). In meadow 3, average soil organic carbon was 42.64% higher in 2022 than 2021 (p = 0.05) and average total nitrogen concentrations were 49.20% higher (p = 0.02); these increases likely reflect within-meadow heterogeneity. Figure 4 shows the stark differences between stream geomorphology and vegetation that may control carbon and nitrogen cycling processes.

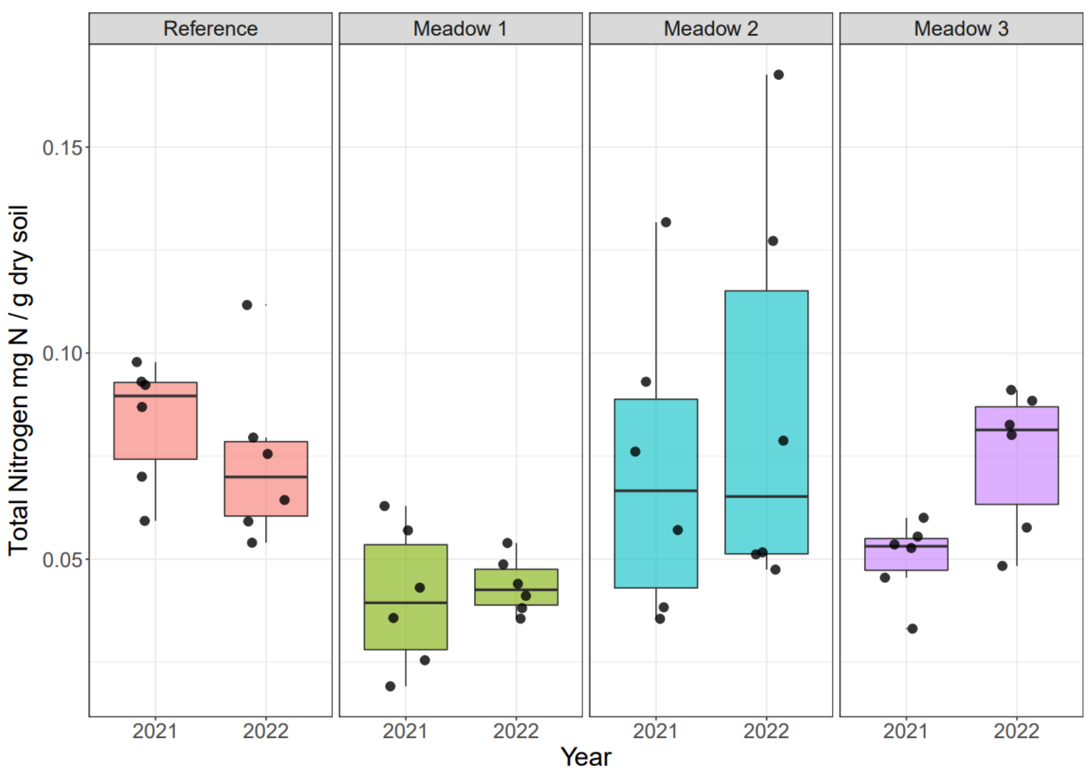
We observed a significant interaction between meadow and year on soil pH (Table 3; $F_{3,20}$= 4.09, p= 0.02). pH remained relatively constant across years within the three treatment meadows, but values in the reference meadow were 26.6% higher in 2022 than 2021 (p-value=0.05) (Figure 11). Average soil moisture values were 75% higher in 2022 than 2021 ($F_{1,20}$= 19.59, p < 0.001) reflecting significant differences in annual precipitation and drought severity. The lack of significant interactions suggests beaver dam analogs did not directly influence soil moisture during the first year of meadow restoration (Table 3).

One-year post-BDA installation, soluble pools of carbon, nitrogen, and phosphorus did not differ between reference and treatment reaches (Table 1). Total dissolved organic carbon exhibited a significant distance from channel by year interaction. Dissolved organic carbon values increased by 82.69% at a distance of four feet from the stream channel ($F_{2,20}$= 6.36, p < 0.001). In general, dissolved organic carbon was 24% higher in 2022 than 2021 (Table 3; $F_{1,20}$= 10.49, p < 0.01) and about 1.6 times lower at 2 feet than 4 or 6 feet from the channel ($F_{2,20}$= 10.57, p < 0.001). We observed an interaction between year and meadow on total dissolved nitrogen, where concentrations were significantly higher in 2021 than 2022 in the reference reach and meadows 1 and 2, with no difference between years in meadow 3 ($F_{3,20}$= 5.64, p< 0.01). From 2021 to 2022, total dissolved nitrogen concentrations decreased by 37.12% in the reference meadow, by 49.10% in meadow 1, and by 33.2% in meadow 2 (Table 3). In contrast, total dissolved nitrogen values in meadow 3 increased by 43.15% from 2021 to 2022. On average, total dissolved nitrogen was 26.6% higher in 2021 than 2022 ($F_{1,20}$= 13.18, $p< 0.01$) and the greatest decrease in total dissolved nitrogen availability occurred in meadow 1 (15.5%; Table 1). Nitrate also exhibited a significant main effect of year, where concentrations were 3.5 times higher in 2022 than 2021 ($F_{1,20}$= 159.60, $p < 0.001$) (Table 3). We observed no significant interactions or main effects for ammonium or phosphate concentrations suggesting neither annual variability nor the presence of beaver dam analogs directly altered nutrient availability in floodplain soils adjacent to the ephemeral stream.

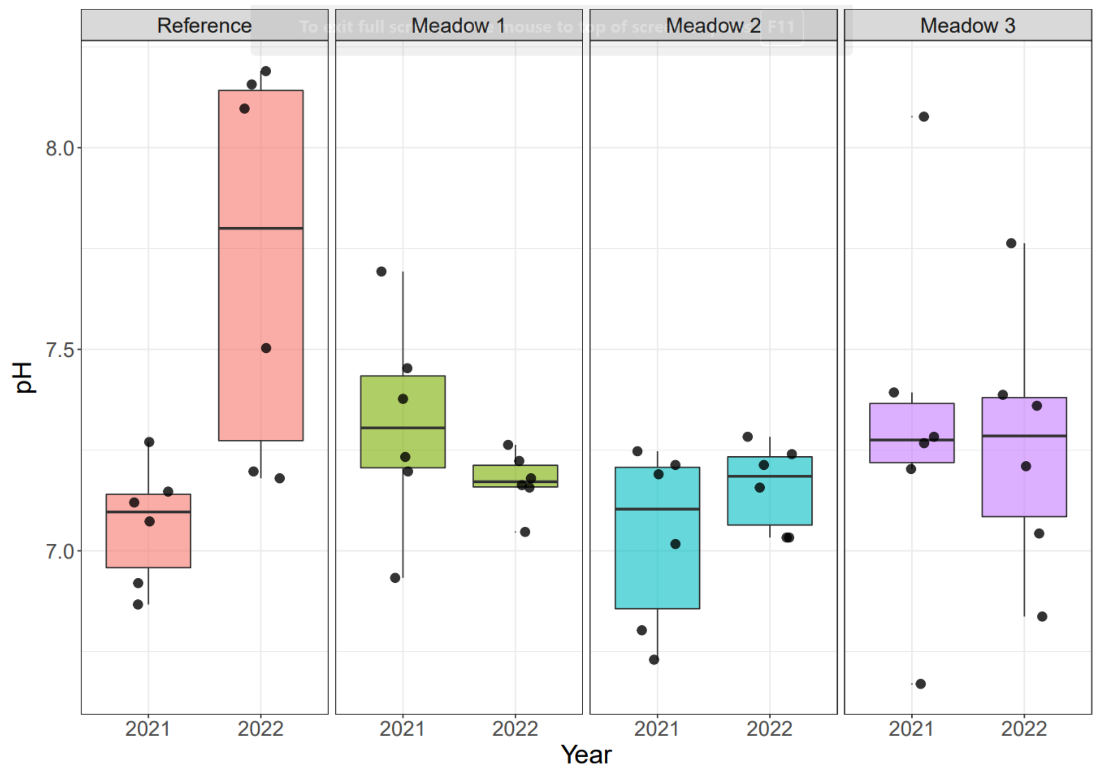
Microbial biomass carbon pools were 1.6 times greater in 2022 than 2021 (Table 3; $F_{1,20}$= 15.59, $p < 0.001$). We observed no significant interactions (e.g., treatment*meadow) suggesting annual changes in environmental factors, such as greater soil moisture in 2022, created conditions favorable for microbial growth (Figure 11).



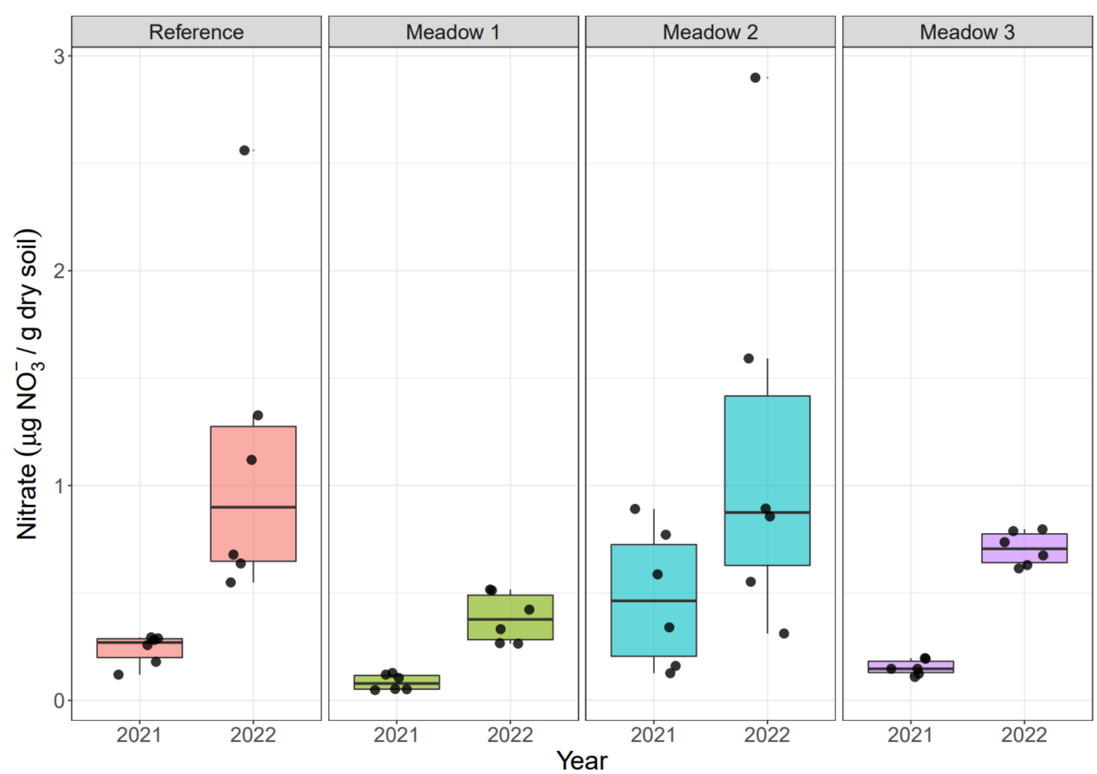
A. Soil carbon in mg C / g dry soil measured by EA-IRMS



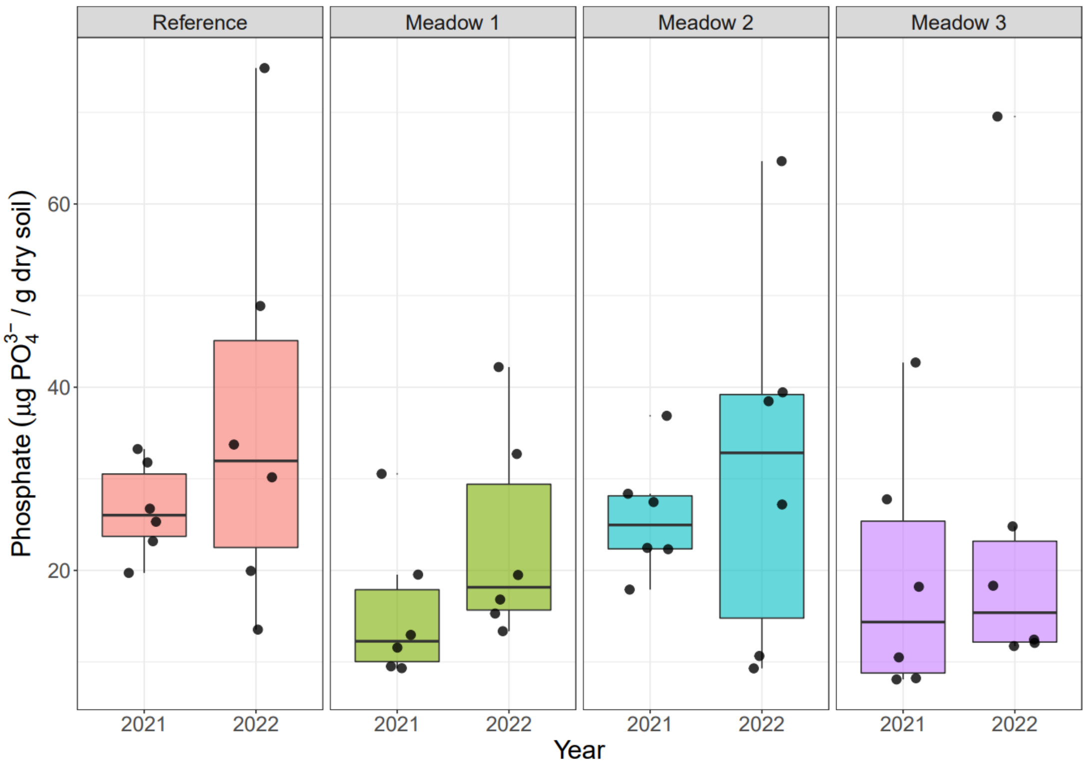
B. Soil Nitrogen in mg N / g dry soil measured by EA-IRMS



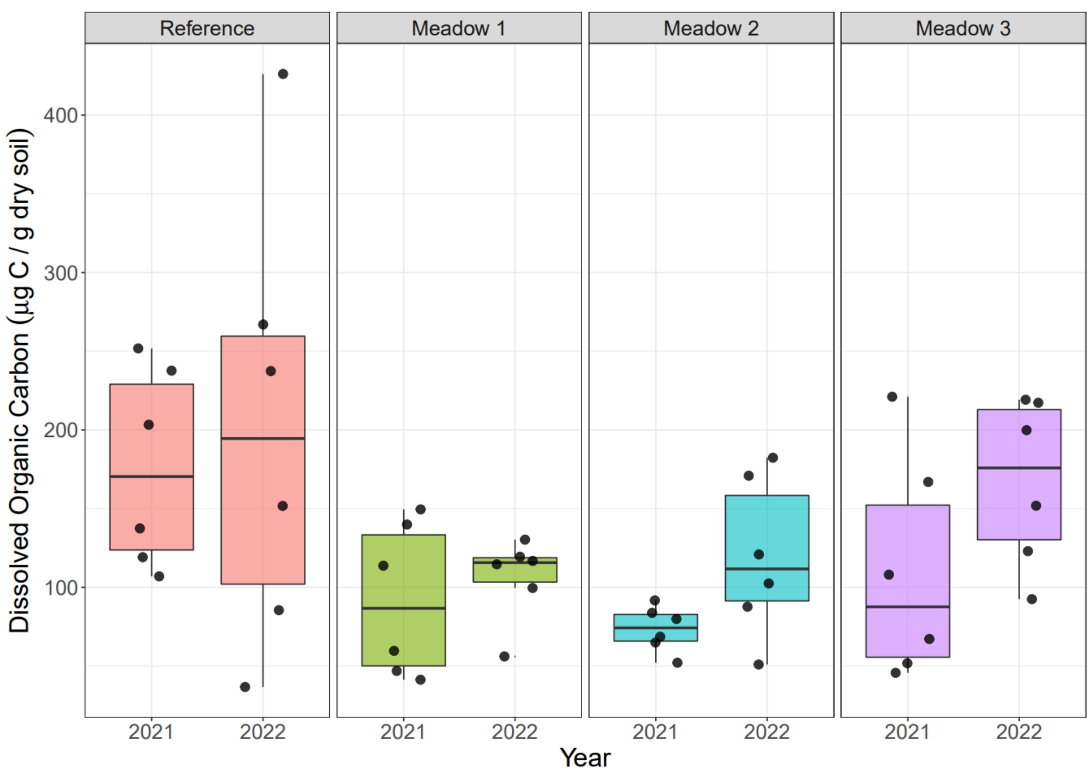
C. Soil pH values



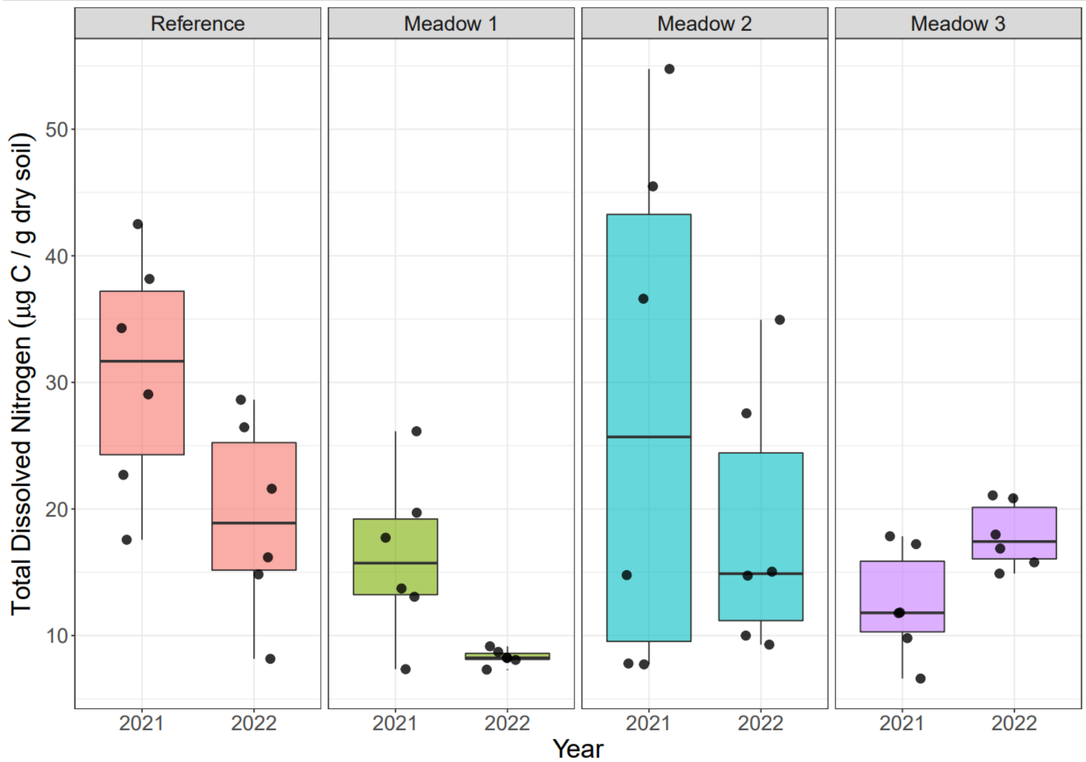
D. Nitrate values in micrograms per dry soil



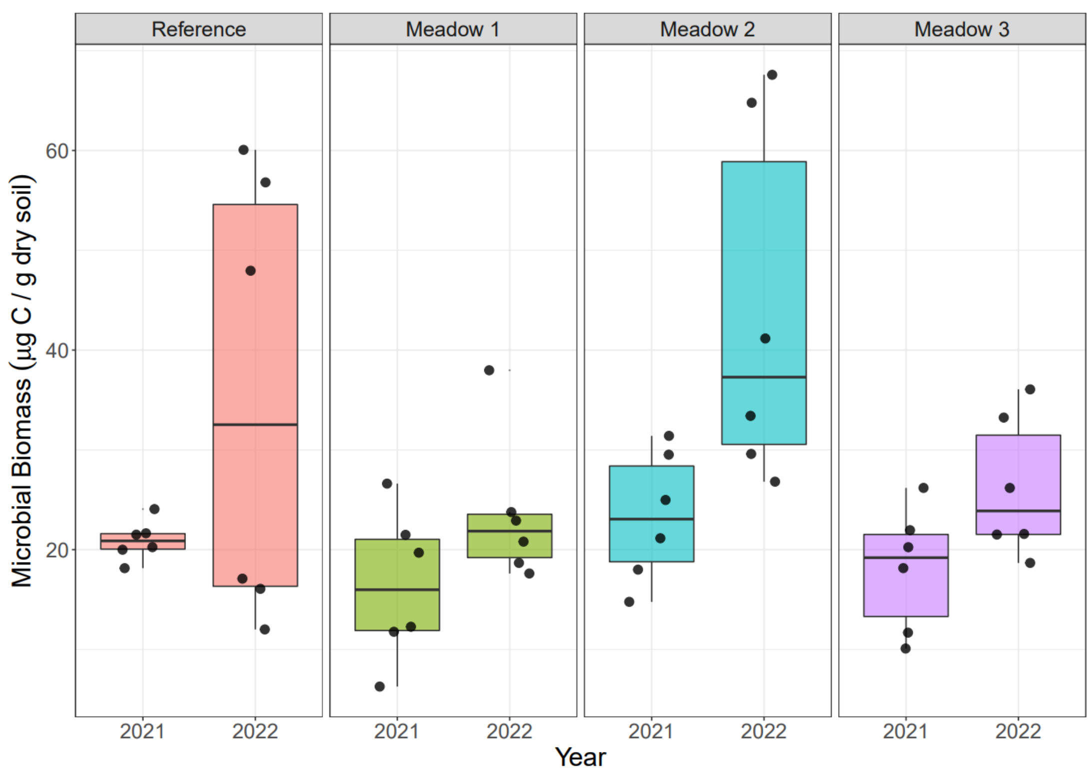
E. Phosphate values in micrograms per gram of dry soil



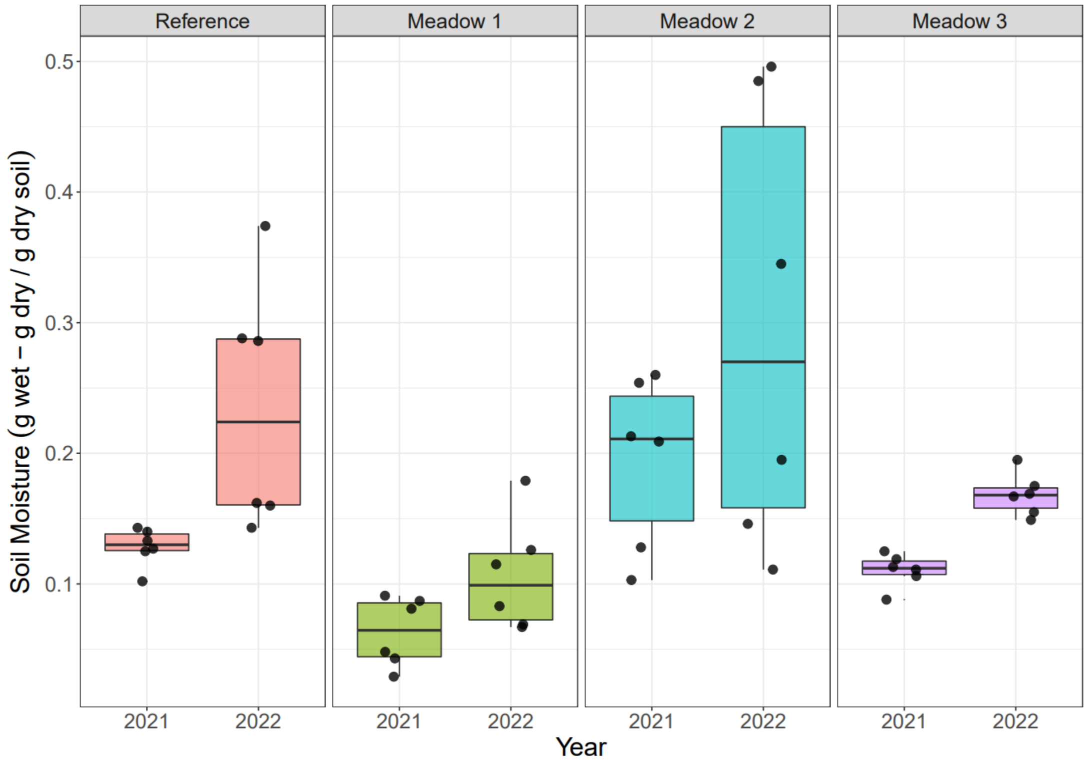
F. Dissolved organic carbon in micrograms C / g dry soil measured by Shimadzu TOC/TN Analyzer



G. Total dissolved nitrogen in micrograms N / g dry soil measured by Shimadzu TOC/TN Analyzer



H. Microbial biomass in micrograms C / g dry soil



I. Microbial biomass in micrograms C / g dry soil

Figure 11 A-I. Soil parameters compared by treatment meadow and reference between 2021 (no installation of beaver dam analogs) and 2022 after beaver dam analog installation.

Table 3. Three-way anova interaction significance for soil data before and after beaver dam analog installation.

Year	Meadow	pH	Soil Moisture (g wet-g dry)/g dry	Nitrate (ug /gdw)	Ammonium (ug /gdw)	Phosphate (ug /gdw)	Biomass (µgC/g soil)	TOC (µgC/g soil)	TDN (µgC/g soil)	Soil C (mg C)	Soil N (mg N)
2021	Reference	7.07 (0.07)	0.13 (0.01)	0.24 (0.03)	1.89 (0.24)	26.66 (2.09)	20.93 (0.81)	176.04 (25.68)	30.71 (3.86)	1.06 (0.09)	0.08 (0.01)
	Meadow 1	7.31 (0.11)	0.06 (0.01)	0.08 (0.02)	0.58 (0.08)	15.58 (3.36)	16.36 (3.07)	91.8 (19.77)	16.28 (2.63)	0.47 (0.09)	0.04 (0.01)
	Meadow 2	7.03 (0.09)	0.19 (0.03)	0.48 (0.13)	1.83 (0.32)	25.9 (2.69)	23.3 (2.66)	73.47 (5.87)	27.86 (8.35)	0.95 (0.21)	0.07 (0.01)
	Meadow 3	7.32 (0.18)	0.11 (0.01)	0.15 (0.01)	1.19 (0.19)	19.25 (5.61)	18.05 (2.517)	110.1 (28.84)	12.51 (1.77)	0.61 (0.04)	0.05 (0)
2022	Reference	7.72 (0.2)	0.24 (0.04)	1.15 (0.31)	1.54 (0.39)	36.85 (9.08)	35 (9.09)	200.73 (57.5)	19.31 (3.15)	0.84 (0.11)	0.07 (0.01)
	Meadow 1	7.17 (0.03)	0.11 (0.02)	0.39 (0.04)	0.6 (0.18)	23.31 (4.71)	23.62 (3.02)	106.11 (10.79)	8.29 (0.254)	0.51 (0.03)	0.04 (0)
	Meadow 2	7.16 (0.04)	0.3 (0.07)	1.18 (0.39)	3.53 (1.37)	31.62 (8.48)	43.89 (7.33)	119.19 (20.49)	18.59 (4.23)	1.1 (0.27)	0.08 (0.02)
	Meadow 3	7.27 (0.13)	0.17 (0.01)	0.71 (0.03)	1.09 (0.09)	24.82 (9.18)	26.21 (2.87)	167.25 (21.65)	17.91 (1.05)	0.88 (0.09)	0.08 (0.01)
meadow											
distance								***		**	***
year			***	***			***	**	**		.
meadow*dist											
meadow*year		*							**	.	.
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meadow*dist*year											

2.4 Discussion

Interest in beaver dam analogs has grown throughout the Western United States because they offer an accessible restoration technique due to their low cost (Pilliod et al., 2017). A total of 59 beaver dam analogs were installed throughout 3 treatment meadows. Our first hypothesis that organic carbon and nutrients would increase in surface waters was not supported, as we observed no significant differences in these constituents throughout 2022. Beaver dam analogs did not alleviate climate-induced drought at our study site, but also did not reduce dissolved oxygen or increase phosphate concentrations within the stream water over the 2022 season. We also did observe any indicators of stream eutrophication, even with extreme heat and drought, suggesting beaver dam analogs did not increase environmental risk at our study site. Finally, we did not find any evidence supporting our third hypothesis, however, increases in microbial biomass with soil moisture (2022 compared to 2021) suggests rewetting and floodplain inundation could stimulate microbial growth and create an environment for more efficient soil microbial processes. Continued monitoring efforts at Guy Canyon should assess how discharge varies above and below beaver dam analogs and among meadows, capturing additional seasonal and annual variability in drought conditions.

Southeastern Idaho is a dry environment, with drought regularly occurring in Blaine County over the last twenty years (Figure 12). The sampling years themselves differed dramatically: in 2021, a period of exceptional drought occurred late in the summer, while in 2022 extreme drought was only briefly reached, with most of the summer experiencing moderate drought conditions. Differences in drought intensity between our sampling years strongly influenced surface water availability and quality, independent of the beaver dam analogs.

In arid regions, where water resources are limited, beaver dam analogs are being installed as a means of increasing water retention on the landscape. In some instances, these restoration projects may be successful. However, without sufficient water supply, beaver dam analogs cannot function as intended and are unlikely to deliver expected ecological benefits. At Rinker Rock Creek Ranch, significant drought conditions limited water supply in the intermittent stream, restricting water availability for sampling and restoration. At the beginning of each summer, we observed a very weak pulse of snowmelt-driven flow that pushed water through and around each structure but had insufficient volume to dramatically increase pooling or sediment storage within the channel. Peak runoff was then followed by a long low-flow period beginning in June of each year. During this period, trickling and stagnant water flowed through the meadows, which began to hydrologically disconnect from each other. By early July, most of the channel had dried up, leaving the installed structures high and dry. With only about a month or two of flowing water in the stream, the newly installed beaver dam analogs were functioning at reduced capacity for only a small subset of the water season. Flow was not extended any further than compared to 2021 because of the lack of natural water supply.

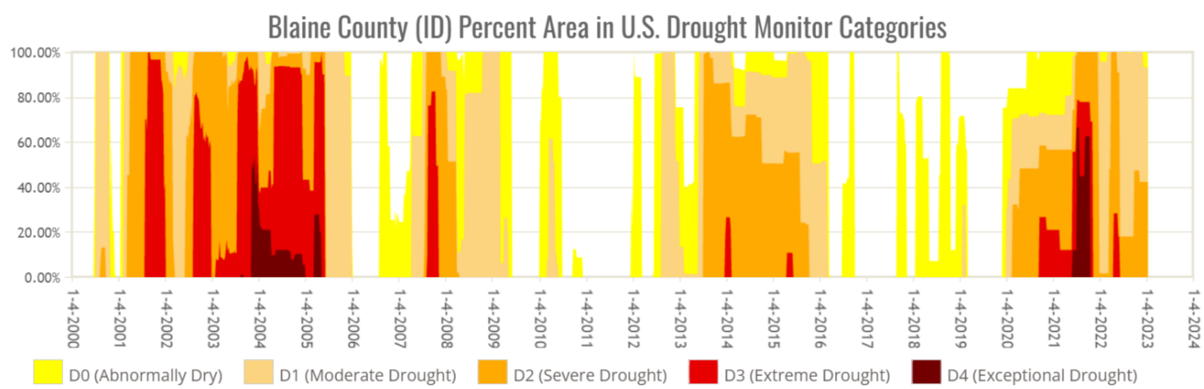


Figure 12. Blaine County, Idaho has experienced multiple periods of extreme and exceptional drought in the last two decades. For the duration of our study period (April 2021—current), the site experienced continuous drought stress, ranging from abnormally dry to exceptional drought conditions.

2.4.1 Water

Air temperatures in 2021 were higher for longer (June-August) than the same period in 2022. Not surprisingly, our surface water temperature data followed similar trends. Water temperatures increased throughout the summer in both years. Since Idaho is in a high desert climate, nights are significantly colder than daytime temperatures, so sampling time differences can have a major impact on water temperatures in a small, intermittent stream. An example of this in our dataset occurred in 2022, where our sampling date in July appears to be colder than June. However, July data were collected earlier in the morning than

June data, which were collected midday. All our water quality data are temperature-corrected, so this will not bias our results, but should be considered when comparing trends in biogeochemical parameters across time.

Dissolved oxygen concentrations did not differ across sampling times in 2021 but were significantly more variable in 2022 (Table 1) and appear to be tightly linked with precipitation. In 2022, we sampled during May when appreciable precipitation (seven-day cumulative precipitation = 1.2 inches) was associated with the highest DO concentrations (~9.6 ppm). In contrast, the driest sampling period was in June (seven-day cumulative precipitation = 0 inches) and associated with significantly lower DO concentrations (~6.9 ppm). The relationship between precipitation and DO is well documented. In non-urban environments, DO is likely to increase with more precipitation because rain saturates with oxygen as it travels through the atmosphere to the land surface (Piffer et al., 2021). We monitored DO levels as a water quality parameter to determine whether beaver dam analogs increased eutrophication risk. At our study site, DO levels did not differ between reference and treatment reaches, suggesting beaver dam analogs influenced DO concentrations to a lesser extent than seasonal trends in temperature and precipitation. DO was also measured to assess local environmental conditions. For example, high DO concentrations permit microorganisms to utilize aerobic metabolic pathways and catalyze biogeochemical reactions (e.g., transform ammonium to nitrate). Hyporheic zones created by dams can boost dissolved oxygen and nutrient exchange (Wade et al., 2020). Lending support to this argument, we found no significant differences in nitrate concentrations in the treatment meadows. Without consistent increases in dissolved oxygen and nutrient availability or substantial leaching from adjacent soils, we would not expect nitrate levels to differ. Relatively high DO levels and moderate nitrate concentrations throughout treatment meadows suggest the beaver dam analogs did not appreciably increase the risk of eutrophication.

The concentration of total dissolved solids in the stream did not change significantly across sampling periods in 2021, but was highly variable in 2022, suggesting the installation of beaver dam analogs may have influenced sediment transport. When the stream was flowing, TDS levels were lower than when sections of the stream disconnected, and stagnant pools developed. Suspended solids, or particulates, are often enriched in nutrients, which can stimulate the growth of aquatic vegetation and microbial communities and the consumption of dissolved oxygen (Chounlamany et al., 2017). As a result, DO levels are often negatively correlated with the concentration of suspended solids. Beaver dam analogs are expected to accumulate sediment and organic carbon through deposition with decreased water velocity and physical trapping of sediments behind the structures (Scamardo & Wohl, 2020). Increase water riffling downstream of these structures has been shown to be a successful effect of beaver dam analogs, much like natural beaver dams (Reinert et al., 2022). Riffling downstream of beaver dam analogs can provide hyporheic exchange increasing dissolved oxygen and dissolved solids, allowing for increased biogeochemical processes (E. G. Norman, 2020; Wade et al., 2020). While TDS and DO levels may serve as sensitive indicators of stream water quality, further work is needed to disentangle the effects of BDA installation from precipitation and drought.

When focusing on meadow differences in 2022, we can extrapolate potential impacts from beaver dam analogs when comparing meadow water availability and water quality parameters to the upstream reference reach. When looking at DO and TDS on a spatial scale, we observed no significant differences by location or month, suggesting climate as the likely factor driving variability in both parameters. Because beaver dam analogs have the potential to change residence time of water and sediment accumulation, dissolved oxygen and total dissolved solids could both decrease with pooling and slowing of water. Though, with an increase in hyporheic exchange from beaver dam analogs, dissolved oxygen directly below a beaver dam analog could increase. With more and prolonged available water, beaver dam analog impacts on these parameters could be better understood.

In contrast, ammonium concentrations increased over the summer, which could be due to decreasing water availability in the stream, which concentrates nutrients in shallow surface water pools. Interestingly, a study comparing intermittent and perennial streams found a higher concentration of ammonium in the intermittent stream when compared to a nearby perennial stream (von Schiller et al., 2008). This pattern was largely attributed to the higher surface area ratio of water in contact with stream bed sediment in a slow moving, smaller intermittent stream than a larger stream. Larger streams have a higher velocity of water which could increase nitrogen export downstream. Microbial and plant competition for available nutrients would also influence nutrient distribution within a stream. In a productive riparian meadow, we may expect high biological demand for nutrients would decrease ammonium availability and reduce concentrations exported from the soil to the stream channel. As biological demand relaxes (e.g., following fall senescence), nitrate and ammonium export might again increase (Hobbie & Chapin, 1996; Treat et al., 2016).

Phosphate concentrations also increased over the summer, mimicking trends in ammonium, which could indicate a lack of plant uptake, or a high surface water to sediment ratio. Phosphate also displayed a significant relationship with valley confinement, which may indicate potential losing and gaining reach influences. If valley confinement is lower, there is likely more pool and riffling enhancing hyporheic exchange, this allows for phosphorus deposition in sediments and therefore a lower concentration of phosphate in water (D Tonina & Buffington, 2007). Phosphorus would be more easily transported in areas of higher flow, resulting in areas of lower phosphate concentrations (Nagel, 2014). In the absence of animal inputs and litter degradation, new phosphate inputs are provided through weathering of parent material (Feng et al., 2019; Whitfield et al., 2019). As a result, phosphate cycles much more slowly than other elements, including carbon and nitrogen (Skoulikidis & Amaxidis, 2009; Tzoraki et al., 2007). In addition, phosphate is not soluble in water and is harder to transport in or out of a stream system. Because cattle were not present in the study site, the decrease in phosphate over time is expected (Heathwaite & Johnes, 1996). Similarly, plant and microbial demand for phosphate often increases into the growing season and could explain decreases in phosphate concentrations towards the end of each summer (von Schiller et al., 2008). Alternatively, declines in phosphate could be driven by decreases in stream discharge over the season, which may lead to less sediment transport (Jarvie et al., 2002).

Hyporheic zones exist within the stream system due to natural losing and gaining reaches that are controlled by geomorphology and groundwater levels. Hyporheic zones can help maintain microbial processes and stimulate biological activity within the surrounding soil (Coulson et al., 2021). Beaver dam analogs have been shown to increase groundwater levels above dam sites, allowing for increased hyporheic exchange (Wade et al., 2020) that mimics processes found in natural beaver meadows (Castro, 2017). In addition to increasing nutrient and oxygen exchange (Wade et al., 2020)more extensive hyporheic zones facilitate microbial and vegetation growth.

Overall, our results suggest beaver dam analogs did not increase the risk of eutrophication (e.g., nitrate and dissolved oxygen concentrations were similar between reference and treatment reaches and we observed no algal blooms). Extending water availability throughout the season allows for an expanded timeline of nutrient cycling in snowpack-dependent environments with limited growing seasons, which has been shown with natural beaver dams (Water Quality Responses to a Semi-Arid Beaver Meadow in Boise, Idaho). Beaver dam analogs therefore have strong potential to restore the soil and water systems throughout arid lands in the US and increase environmental resilience to global change factors (e.g., warmer temperatures, greater drought risk). However, further work is needed to test their utility in a wider variety of settings (e.g., gaining versus losing reaches, intermittent versus perennial streams, etc.) and long-term monitoring is needed to ensure they do not impair surface water quality (e.g., by increasing eutrophication risk) or quantity (e.g., by reducing downstream water availability, etc.). More work is needed to test whether beaver dam analogs will be an effective restoration technique in drought-impaired environments. While beaver dam analogs have strong potential to re-establish channel-floodplain connectivity, sufficient flow within the stream channel is required for them to work. Intermittent streams in drought-prone regions may thus not be suitable locations for BDA-based restoration efforts.

2.4.2 Soil

Soil organic matter forms over hundreds to thousands of years and is controlled by biological and abiotic factors (Abatzoglou & Williams, 2016). Because of this, the biogeochemical properties of soil depend on complex feedbacks between plant, microbial, and climate factors. Dryland systems are less productive than other ecosystem types (e.g., forests, wetlands) because low precipitation limits biological productivity (e.g., plant and microbial) and the belowground cycling of carbon and nutrients. Across ecosystems, organic matter concentrations tend to decline with warmer temperatures, which stimulate microbial activity and the mineralization of soil carbon to carbon dioxide or methane (Feng et al., 2019). Tracking soil organic carbon, moisture, and nutrient availability is critical to monitoring the impact of beaver dam analogs on the fluvial carbon cycle since changes in water availability and organic matter can have major effects on soil biogeochemical parameters.

We found that soil organic carbon stock increased significantly with distance from the edge of the stream channel (2-ft, 4-ft, & 6-ft), with no significant differences by year. Because soil biogeochemical processes require time to respond to changes in land use, it is not surprising that one year of restoration did not appreciably increase soil carbon levels; this variable will likely take decades to change (Samaritani et al., 2011). The positive relationship between soil carbon and distance from the active stream channel may be the result of geomorphic processes, where soil carbon is more likely to accumulate in areas with less erosion (Hancock et al., 2019). It is also possible that areas further away from the stream have finer grain sizes and therefore can hold more organic carbon than larger grain sizes (G. Li & Pang, 2014).

We did observe a significant interaction between meadow and year on soil organic carbon concentrations. Between 2021 and 2022, soil organic carbon levels increased significantly in meadow 3, but not within the reference reach or the other two restored meadows, suggesting the increase was not driven by climate differences between years. We did observe longer water availability in meadow 3 than meadows 1 and 2, which were dry for much of each growing season. Meadow 3 had trickling streams that were witnessed in the field during May and June, helping support longer term water availability. The greater water availability in meadow 3 could support microbial and plant productivity and may eventually increase the effectiveness of beaver dam analogs in restoring ecosystem function. Alternatively, the heterogeneity in organic carbon concentrations is extremely high within soils. Longer-term monitoring will be required to test whether soil organic carbon sequestration continues increasing within meadow 3, or whether variability in soil carbon stocks was due to insufficient sample size.

We also observed a significant interaction in soil nitrogen availability between year and meadow, with significantly higher concentrations in meadow three. Meadow three also had greater water availability, which may have increased local rates of nitrogen uptake and immobilization within microbial biomass. Microbial communities require nitrogen to build biomass and decompose plant litter (Meadows, 1979). Microorganisms tend to immobilize nitrogen in biomass when soil carbon to nitrogen ratios are high (e.g., during periods their growth is nutrient limited). In contrast, when nitrogen availability increases, microbial communities may mineralize nitrogen, increasing the relative abundance of ammonium or nitrate in soil porewaters. Ammonium can interact with soil minerals, reducing its transport to the streams, relative to nitrate, which is highly vulnerable to leaching (Van Keulen, 1979). If greater water availability stimulates biological activity, nitrogen retention within soils could increase, a pattern that is often detected in arid environments (S. L. Johnson et al., 2005).

Precipitation can increase nutrient leaching and the erosional loss of nitrogen (Nearing et al., 2005). The increase in nitrogen concentrations in soils located further from the stream channel with less steep features such as vertical cutbanks, which reduces erosional risk during rain, snowmelt, and flooding events. Drought can also facilitate nitrogen accumulation in upper soil horizons (Cregger et al., 2014). Because there was more available water in the stream at meadow 3, but no difference in precipitation across meadows, the increase in nitrogen and organic carbon was likely driven by greater biological productivity, including carbon fixation by plants and the production of microbial biomass.

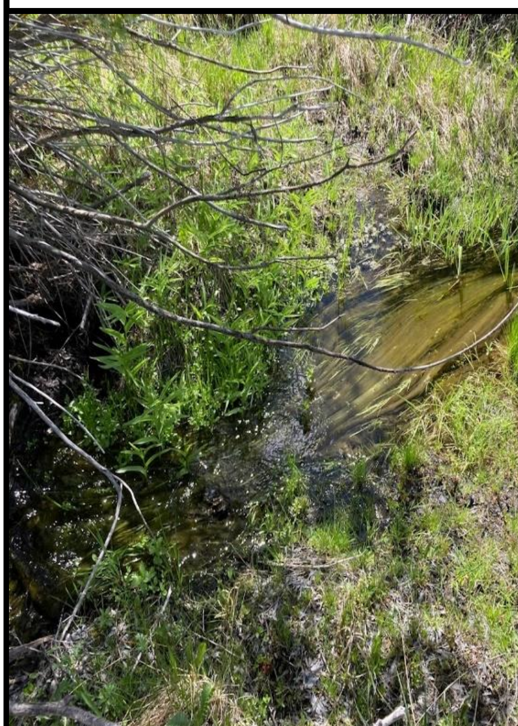
Visually, meadow 3 had the most plant growth, including within the stream channel itself. Figure 13 shows that while all meadows had water in May at the start of summer, water quantity slowly diminished through July. Meadow 1 was by far the most incised system in our study, followed by meadow 2 with some significant cut banks, and meadow 3, where channel incision was minimal (maximum = ~2-ft deep). Meadow 3 therefore has more stream-floodplain connectivity and greater biogeochemical feedbacks between terrestrial plants and microbiomes than the other meadows. Not only does low incision allow for more surface area contact between the water and streambed/floodplain, but it also provides water to shallow root systems enabling more riparian vegetation to grow.

Meadow 3 May 2022

Meadow 1 June 2022

Meadow 2 June 2022

Meadow 3 June 2022



Meadow 1 July 2022

Meadow 2 July 2022

Figure 13. Pictures from May through July of 2022 differences between meadows in water availability, vegetation (e.g., biomass and species composition), and stream channel morphology.

Soil moisture was significantly higher in 2022 than 2021 across all locations and is a major environmental factor driving biological productivity and soil organic matter sequestration. Greater precipitation in 2022 is the most probable cause leading to observed increases in soil moisture. For example, our experimental site received 0.05 inches of cumulative precipitation seven days before sampling in 2022 and 0 inches during the same period in 2021. Unfortunately, we observed no floodplain inundation during the summer months following the installation of beaver dam analogs. Soil moisture was also significantly higher in the reference meadow suggesting annual variability in precipitation was the likely driver. Shifts in soil moisture could play a significant role in explaining annual variability in biogeochemical parameters such as microbial biomass (M. C. Fernandes et al., 2022). In soils, the movement and activity of bacterial populations is constrained to wetted soil pores. As a result, insufficient soil moisture will not only limit belowground nutrient transport but could also reduce microbial dispersion and activity (Six et al., 2006).

We observed no significant changes in soluble carbon, nitrogen, or phosphorus pools across year or meadow. The lack of change in soil carbon and nutrient stocks is not surprising as there were no changes in grazing or land use during our study period (Miller et al., 2014). Similarly, both carbon and nitrogen stocks are large in soils, and concentrations are heterogeneous, making it difficult to detect small changes in concentration over time. Similarly, because floodplain rewetting did not occur in the first year following BDA installation, nutrients would not be differentially exchanged between surface water and soil compartments. On a longer-term scale, we do expect to observe changes in nutrient input and availability. For example, rotational grazing will continue in this location, which will increase nitrogen inputs via cattle excretion (Miller et al., 2014). Rewetting events can release nitrogen and other nutrients like dissolved organic carbon from leaf litter into the soil, including through erosion, which transports nutrients from the soil into the water system (Skoulikidis & Amaxidis, 2009). During the dry season, litter accumulation in the stream bed can slowly release nutrients and dissolved organic carbon to the stream channel, particularly following more active decomposition periods, such as during the wet season. Similarly, in a non-drought year, a deeper snowpack could increase early season flows and enhance the effects of beaver dam analogs within the stream channel. During wetter winters, installed structures could influence water flow into the summer, increasing nutrient delivery from soils to streams and increasing rates of biological activity and decomposition (Tzoraki et al., 2007).

Beaver dam analogs have not been shown to effect soil floodplain dissolved organic carbon levels, so this parameter may not be a targeted way to examine the effects of rewetting and water table rise impacts on soil (Pearce et al., 2021b). However, we did observe a significant interaction between year and distance, where dissolved organic carbon concentrations increased by 80% between 2021 and 2022 in midland sites (4-ft from the stream channel), by 36% at the near-channel sites (2-ft distance), and by 6% in the upland site (6-ft distance). Since we did not observe a distance by year interaction for soil moisture, it appears that variability in dissolved organic carbon concentrations are not driven by water availability alone. If data collection efforts continue at this site, it would be important to monitor changes in dissolved organic carbon as a function of stream discharge and water table height. For example, I would expect that dissolved organic carbon levels would be highest nearest the channel, particularly if periods of floodplain inundation stimulate biological productivity. Similarly, transitions in vegetation composition (e.g., replacement of woody upland shrubs like sagebrush by mesic meadow plants like sedges and grasses) could increase dissolved organic carbon production in near- and mid-channel sites. Although nitrate concentrations were 3.5 times higher in 2022 than 2021, they did not vary systematically across the three treatment meadows or between reference and treatment reaches. Drought severity and other climate factors thus appear to exert greater influence on nitrate cycling than short-term effects related to the installation of beaver dam analogs.

Microbial biomass increased by over 1.5 times from 2021 to 2022. Some factors that may affect microbial biomass concentrations include pH and soil moisture (Williams, 2007). Since soil moisture and precipitation were significantly higher in 2022 than 2021 this likely explains variability in microbial biomass. Because moisture is so integral to soil processes, water quality and quantity will play a significant role in determining soil health and should be monitored during stream restoration projects.

A sufficient inflow of water from winter snowmelt is needed for beaver dam analogs to influence local hydrologic conditions. Moderate and severe drought conditions meant that each meadow had flowing water within the channel for 1-2 months, and at levels and flow rates too low to be significantly impacted by channel obstruction. In a high-water year, we would expect installed structures would trap water behind them, causing water to spill onto surrounding floodplain soils where it can re-establish soil pore connectivity and stimulate biological productivity. Greater plant biomass and net primary productivity would further stimulate microbial activity by releasing root exudates and high-quality (e.g., more nitrogen-rich) leaf litter to the soil surface. Reducing drought stress in a riparian environment could in turn elevate carbon and nitrogen storage, by facilitating nutrient uptake and immobilization in microbial biomass, the precursor of soil organic matter (Camenzind et al., 2023; Cotrufo et al., 2013; Liang et al., 2019). However, major changes in soil conditions will take time to manifest; one season will not be enough time since shifts in biogeochemical processes work on a longer timeline. In the absence of ongoing drought, I would expect beaver dam analogs could alter channel morphology within several years, and that the movement of water into adjacent floodplains could reduce channel incision, accumulate sediments, trap nutrients, increase plant and biological productivity, and extend peak flows later in the summer in arid drylands of southern Idaho. Continued monitoring of soil properties (e.g., carbon and nitrogen levels, dissolved pools of inorganic nitrogen and phosphorus, and microbial biomass) should be used to test the effectiveness of ecosystem restoration at this site.

2.5 Conclusion

Arid lands make up about a quarter of the land in America (AghaKouchak et al., 2013). These systems are especially vulnerable to climate change because they are already water- stressed (Overpeck & Udall, 2020). Arid land riparian systems are not exempt from the negative impacts of a warmer climate and efforts must be made to restore stream function and create more resilient ecosystems. Beaver dam analogs offer a cost-effective restoration approach to mimic the beneficial impacts of natural beaver dams without the issues associated with establishing natural beaver populations in multi-use rangelands.

Both natural beaver dams and beaver dam analogs have been used to restore arid and semi-arid riparian environments, increasing the supply of water and nutrients for plant and microbial growth (Fesenmyer et al., 2018; L. M. Norman et al., 2022; Pilliod et al., 2018; Silverman et al., 2019). It has been observed that beaver dams and beaver dam analogs create a nutrient sink that stores organic matter, nitrogen, phosphorus, and other water constituents stored in streambed sediments and dissolved in water (Čiuldienė et al., 2020; Wade et al., 2020). Increasing water and nutrient availability in aridlands is vital for sustaining riparian function.

Beaver dam analogs increase water availability later in the season by increasing surface water storage and groundwater recharge (E. G. Norman, 2020; Scamardo & Wohl, 2020). Hydrograph trends in snowpack-dominated systems show the trend of a decreased period of water availability and a shift in timing of peak discharge to earlier in the spring, when water demand from plants is not as high (Dettinger et al., 2015). Water storage and capture of spring snow melt is thus critical to sustain year-round water availability for local ecosystems.

Our research shows promise in riparian soil health indicators of carbon and nitrogen storage. The increase in microbial biomass in soil samples between years may be correlated with precipitation, but this shows promise with re-wetting caused by beaver dam analogs as they become more established. If conditions allow for enough water to fill the stream channel, beaver dam analogs could extend water presence for longer in the season to support vegetation and microbial growth. In addition to soil impacts, beaver dam analogs in Guy Canyon could improve surface water quality by increasing nutrient capture and recycling from cattle waste. We observed no concerning signs in water quality parameters, such as a massive decrease in dissolved oxygen or overaccumulation of nutrients in the treatment sites compared to the control reach. While little research has been done with water quality response to beaver dam analogs, it is well established that natural beaver dams filter and store nutrients that stimulate net primary productivity (Čiuldienė et al., 2020). However, sustained drought conditions for the duration of our study period meant we were unable to complete a comprehensive water quality assessment. Continued monitoring at Guy Canyon is therefore needed to evaluate longer-term changes in ecosystem processes (e.g., soil health, water quality) as a result of the beaver dam analogs.

Overall, it is clear that more research is needed to assess the effectiveness of beaver dam analogs in drought impacted systems. In particular, it remains unclear whether re-establishing floodplain-stream connectivity will accelerate soil nutrient cycling and net primary productivity, and whether those changes could improve ecosystem function. Monitoring changes in ecosystem processes over a longer period of time is thus needed to assess the viability of beaver dam analogs for restoration of intermittent streams.

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