THE RESPONSE OF PARAFLUVIAL SOILS TO BEAVER MIMICRY RESTORATION IN A MONTANE STREAM

by

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DEDICATION

I dedicate this work to the people of Agosa, Brong Ahafo Region Ghana. Without their support and openness to working with me as an Agricultural Peace Corps Volunteer, my enthusiasm to protect natural resources and develop meaningful research wouldn't have been realized. In particular, I'd like to dedicate this work to my Ghanaian brother Amponsah Patrick. His commitment to bettering his community and working with me allowed us to make a positive impact in Ghana.

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NOMENCLATURE

Beaver Mimicry Restoration (BMR): the practice of installing small-scale channel-spanning structures in streams to mimic the hydrological and ecological effects of natural beaver dams. The practice generally uses natural and biodegradable materials to aggrade incised streams and reconnect stream channels to riparian systems.

Beaver Mimicry Structure (BMS): low-gradient, channel-spanning structures designed to mimic the influences of natural beaver dams within the stream ecosystem.

Ecosystem functions: biological, chemical, and physical processes that contribute to sustaining the ecosystem.

Ecosystem services: the benefits that ecosystems provide to human societies, such as resources and a livable environment

Hydric Soil: as defined by the US Natural Resource and Conservation Service (NRCS), a soil that formed under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper horizons.

Indicators of Reduction in Soil (IRIS): strips of Polyvinyl chloride coated with manganese oxide or iron oxide paint. When installed vertically in soils, changes in the appearance of the paint indicate areas of anaerobic microbial electrochemical reduction by providing a final electron acceptor for microbial respiration other than oxygen.

Parafluvial Zone: the area of the floodplain that experiences annual sediment scour and deposition by floods and wind; with early successional riparian vegetation.

Soil reduction: the gain of one or more electrons by an electron acceptor due to anaerobic respiration by soil organisms, such as denitrifying bacteria.

Soil water content (SWC): for a given volume of soil, the ratio of the volume of soil water to the total volume.

Riparian areas: as defined by the NRCS, lands that occur along watercourses and water bodies. Typical examples include flood plains and streambanks. They are distinctly different from surrounding lands because of the unique soil and vegetation characteristics that are strongly influenced by the presence of water.

NOMENCLATURE CONTINUED

Riverine Wetland: areas of standing water that occur in flood plains and riparian corridors in association with stream or river channels. Dominant water sources are often overbank flow from the channel or subsurface hydraulic connections between the stream channel and wetlands. However, sources may be interflow and return flow from adjacent uplands, occasional overland flow from adjacent uplands, tributary inflow, and precipitation.

Wetland:

1. as defined by the Clean Water Act (Section 404), areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas.

2. as defined by the Farm Security Act (Section 3801(a)), areas with a predominance of hydric soils. Is inundated or saturated by surface or groundwater at a frequency and duration sufficient to support a prevalence of hydrophytic vegetation typically adapted for life in saturated soil conditions. Under normal circumstances supports a prevalence of such vegetation. For the purposes of FSA and any other act, this term does not include lands in Alaska identified as having high potential for agricultural development that have a predominance of permafrost soils.

ABSTRACT

Beaver Mimicry Restoration (BMR) is a relatively new aquatic restoration practice that seeks to improve deteriorated stream ecological functions. BMR is designed to rejoin hydrologically disconnected streams with their adjacent floodplains via the installation of small-scale, stream-spanning structures derived from natural materials and inspired by the influence of natural beaver (*Castor* spp.) dams. These structures capture sediment, elevate stream stage and groundwater tables, create thermal refugia, and re-establish riparian vegetation. Most research on BMR has focused on the hydrological or botanical results, but little is known about the response of parafluvial soils.

I report measurements of soil water content, soil temperature, soil biogeochemical reduction, and vegetation responses at paired BMR-influenced treatment and non-BMR-influenced control locations from June through September of 2018 and 2019 in a montane stream in southwestern Montana (USA). In comparison to soils at control sites, soils adjacent to BMR activity experienced an extended period of higher water contents (0.23 m^3/m^3 higher), increased anoxic conditions (on average 27% more during the field season), a less variable and cooler soil temperature range (on average 5° C cooler), and supported longer durations of vegetation greenness (additional 20 days) during the dry months. Results demonstrate how BMR produces conducive conditions for the development of new and/or the reestablish of historic hydric soils.

CHAPTER ONE

LITERATURE REVIEW

Introduction

Beaver (*Castor* spp.) activity is widely accepted to shift the structure and dynamics of fluvial systems through the construction of dams, lodges, and canals (Law et al., 2017; Majerova et al., 2015; Pollock et al., 2007; Wegener et al., 2017). Beaver foraging behavior creates a mosaic of riparian vegetation composition and instream nutrient cycling from vegetation litter (Naiman et al., 1988). With the anthropogenically driven loss of beaver and extensive modifications to stream corridor systems, the deterioration of fluvial and riparian ecosystem functions have become commonplace (Johnston et al., 1988; Muller-Schwarze, 2003). To regain natural beaver populations in landscapes devoid of beaver, these areas must be restored to a state that can once again support beaver habitat (Pollock et al., 2014).

Beaver Mimicry Restoration (BMR) reconnects floodplains with the use of Beaver Mimicry Structures (BMS) to store water, increase biota diversity, change riparian plant communities, store sediment, increase instream thermal refugia, and alter nutrient cycling. Many of the impacts of BMR to ecosystem functions have been well documented (Melanie et al., 2018; Miller et al., 1996; Niezgoda, 2019; Weber et al., 2017; Weber et al., 2016), while the restoration of the ecosystem functions associated with rewetting of riparian soils has been more assumed than demonstrated. The following review of research on natural and artificial beaver dams provides a summary from the

literature of the influences that BMR may have on ecosystem functions. This review also reveals gaps in scientific literature addressed by the following research that characterizes the hydrological and biogeochemical response of parafluvial soil to BMR in a montane stream.

Literature Review

Researchers have shown a large range of ecosystem responses to beaver-facilitated ecosystem functions and subsequent services (Figure 1), such as stream temperature, water storage, sediment and nutrient storage, riparian vegetation establishment, stream morphology, and biotic diversity. This review will address how beaver activities drive ecosystem structure which dictates ecosystem functions. Ecosystem functions are defined as the biological, chemical, and physical processes that contribute to sustaining the ecosystem for this review (Galatowitsch, 2012). Ultimately the quality and quantity of ecosystem function define the ecosystem services, or the benefits that ecosystems provide to human societies, are important from an anthropogenic perspective. However, the focus of this review will be on quantifying and understanding beaver activities on ecosystem structure and functions.

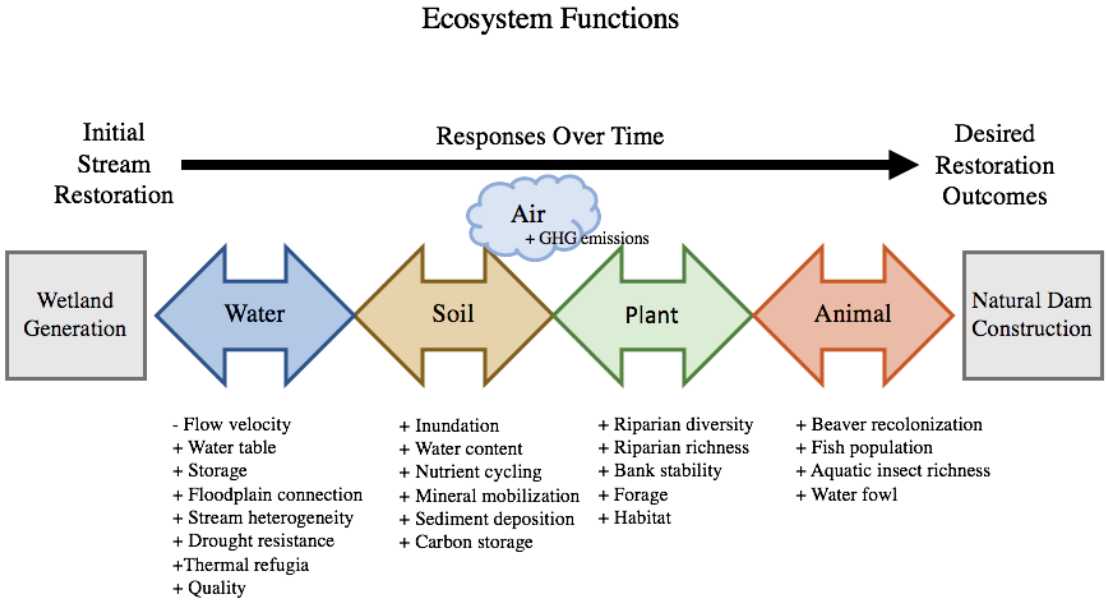


Figure 1: Conceptual model of the cascading effects of Beaver Mimicry Restoration over time. This model shows how the installation of BMS may influence ecosystem structure, functions, and services within the fluvial and terrestrial environment.

Stream Temperature

Beaver activity has a notable influence on instream temperature. The construction of artificial or natural beaver dams has been widely hypothesized to result in higher stream temperatures due to increased surface water area exposure to radiation and decreased vegetation from beaver foraging (Majerova et al., 2015). Several studies have addressed these hypotheses, but they often lack the necessary extent of data across space or time to provide conclusive evidence of a general warming effect (Majerova et al., 2015; Weber et al., 2017). The integrity of stream ecosystems in agricultural catchments is particularly sensitive to instream temperatures due to their increased nutrient loading, and research in these systems has shown that beaver dams may moderate extremes in instream temperatures more than they increase stream temperature overall (Tetzlaff et al., 2020).

Weber et al. (2017) assessed artificial beaver dams and naturally constructed beaver dams and identified two major mechanisms impacting stream temperature: the moderation of extremes in summer stream temperatures at the reach scale (averaging 1.5 river km apart) and an increase in stream-temperature heterogeneity along stream segments (over 28.5 river km). Overall, they found that beaver-dammed stream reaches have water temperatures that more closely correspond with the needs of charismatic biota such as fish. These findings were supported by data showing an increased wetted area above the studied dams. In some cases, reports of over a three-fold increase in wetted channel areas were measured after the establishment of beaver dams (Weber et al., 2016). The assumed associated surface-water groundwater exchange within these ponded areas might be the driving factor for stream temperature changes post-dam construction.

Water Storage

Extended water transit times and increased groundwater storage post dam construction have also been well documented (Tetzlaff et al., 2020). In part, these increases may be due to a 30-40% increase in floodplain connection (Vidon et al., 2020). For example, Puttock et al. (2017) found that the reintroduction of two beavers and their subsequent construction of 13 dams increased water holding capacity in beaver ponds by 1000 m^3. Melanie et al. (2018) found increased channel and pond surface area in two streams in the Upper Missouri River Headwaters Basin (South Long and Alkali Creeks) after BMS construction. Water surface area increased by 19% at South Long Creek and by 21% at Alkali Creek. Groundwater levels were also measured in monitoring wells at

Long Creek and Alkali Creek, and groundwater elevations increased on average by 27 cm and 30 cm respectively in areas affected by BMR installation (Bobst, 2019).

Sediment and Nutrient Storage

In addition to holding water, beaver dams also create conditions that sequester nutrients and sediments from the water column (Allard et al., 2018; Korpak et al., 2019; Niezgoda, 2019; Pollock et al., 2007; Puttock et al., 2018). Globally, freshwater quality is severely impacted by agricultural soil erosion, agricultural runoff, and other anthropogenic land uses (Govers et al., 2017; Xie et al., 2017). The degree to which beaver dams can substantially combat these environmental challenges is an open question. One of the foundational publications on the influence of beaver on stream channel structure showed aggradation due to sediment deposition occurring at rates of 0.47 m per year in beaver ponds (Pollock et al., 2007). Pollock et al. (2017) also showed that as beaver ponds filled with sediment, the deposition of sediment on adjacent riparian terraces increased as well.

Puttock et al. (2018) reintroduced beavers to a 1.8-ha area in SW England containing a first-order stream to address questions about the influence of beaver activity on sediment deposition. Their study estimated that more than 70% of the sediment in 13 beaver ponds was sourced from the managed grassland upstream. Total sediment storage within the ponds averaged 71.40 ± 39.65 kg/m^2. Carbon and nitrogen composition of sediments varied significantly between ponds, with average carbon content at 14.74% and nitrogen content at 0.87% of the total soil mass. These values were significantly higher than those found in channel bed sediments (1.56% C and 0.13% N) both upstream

and downstream from the beaver ponds.

The reintroduction of beaver into a Washington state watershed showed similar carbon and nitrogen retention rates (Fox-Dobbs et al., 2019). Beaver pond sediments averaged four times the carbon content of average steam sediment, and twice the carbon content of sediment in ponds not created by beaver activity. Other work suggests that phosphorus storage within beaver ponds has a positive linear relationship with the age of the beaver system (Levanoni et al., 2017).

Fluvial Morphology

Fluvial heterogeneity and complexity are widely characterized as drivers of valued ecosystem structure due to their associated increase in biological diversity (Daniels et al., 2014; Decamps et al., 1988; Naiman et al., 1988). Studies using frequent aerial photography is adding to the consensus that beaver create instream complexity. By actively felling and transporting trees from riparian zones into aquatic habitat, beaver create permanent and short-term zones of heterogeneity (Cunliffe et al., 2015). This research points out that many of the morphological changes we associate with beaver activity are present within the landscape for a longer duration, while less permanent and small-scale beaver-initiated morphological changes to the stream occur more frequently.

Giriat et al. (2016) assessed 57 dams built by reintroduced beaver and their effects on fluvial structure within the Wisloka River in SE Poland. Their research showed that beaver ponds established mid-channel bars, widened river channels, and elongated pooled steam sections. An associated increase in cascade density and stream riffles also occurred, demonstrating a widespread effect on channel slope, sedimentation, and

channel unit structure. Overall, the influences of beaver activity within aquatic habitats seem to increase morphological complexity and shifted the fluvial channel processes extensively (Giriat et al., 2016; Law et al., 2018).

Riparian Plant Community

The parafluvial zone lines the edges of the dynamic fluvial zone and is characterized by regionally specific riparian vegetation (Stanford et al., 2005). The parafluvial zone is the active interface between the active channel and the less frequently scoured orthofluvial zone, experiencing seasonal fluctuations of high groundwater, sediment deposition, and soil erosion (Stanford et al., 2005). In a 12-year-long study to monitor the impacts of naturally constructed beaver dams on riparian vegetation, Law et al. (2017) found an insignificant increase in plant species occurred in the first two years. However, 10-12 years after beaver dam construction, a substantial increase in plant species diversity had occurred with a 46% increase in species richness relative to the first two years.

The natural construction of beaver dams aids in the establishment of new willow communities by the downstream movement of willow cuttings after forage, dam construction, and dam breaches. Willow cuttings can root and regenerate after being deposited on gravel bars or adequate sediments, and resulting willow stands have been documented to be < 1 km downstream of active beaver dams (Meyer et al., 2019). Beaver seem to influence and establish landscape-scale vegetation patchiness with their foraging and damming behavior. For example, research conducted in New York found that only 21 of 125 total plant species (i.e. 17%) overlapped between beaver engineered and un-

engineered riparian sites (Wright et al., 2002). Dauwalter et al. (2018) found that high beaver dam densities along with livestock exclosure may increase riparian "greenness" by 25% (as indicated by the Normalized Difference Vegetation Index, NDVI). Similar riparian vegetation benefits have been documented at the Silvies Valley Ranch in Oregon, where perennial grasses, sedges, willows, currants, and aspens have naturally reestablished after artificial dam installation (Charnley et al., 2017). Melanie et al. (2018) found similar results, quantifying an average of 20% increase in greenness within a 10-m buffer of the BMR reach and an associated 21% increase in greenness extending for 1-km downstream of the BMS.

Similarly, Fairfax et al. (2018) monitored two streams in northeastern Nevada where cattle had been recently excluded from the riparian area; this resulted in the natural revegetation of the riparian area and eventually recolonization of beaver. In 2013-2016 reaches colonized by beaver were compared to non-colonized reaches using evapotranspiration and NDVI data. The greatest difference between the beaver and non-beaver reaches occurred during the mid-summer months when little to no precipitation occurred. The evapotranspiration rates were 50-150% higher among dammed reaches compared to undammed reaches of the streams. NDVI showed dammed (N = 14) areas with a 6-88% higher greenness compared to undammed areas (N = 14). Their work pushes the idea that the drought buffering mechanism associated with beaver may act like a year-around belowground irrigation system for riparian plants (Fairfax et al., 2018).

Climate Change

Wetlands are widely accepted to provide the largest natural contribution of the greenhouse gas methane to the Earth's atmosphere (Zimmermann et al., 2017), thus expanding wetlands created by a recovering beaver population may have a nontrivial influence on methane generation by freshwater systems. Lazar et al. (2014) assessed three beaver-dammed pond's greenhouse gas emissions by utilizing floating static gas chambers to measure the diffuse flux of greenhouse gases from the air-water interface between fall 2012 and summer 2013. Their research found considerable greenhouse gas emissions (18-556 mg m^{-2} d^{-1}) which varied seasonally, with depth, and by organic content.

Although greenhouse gases are of major concern with respect to our warming climate, Lazar et al. (2014) also highlight the increased nutrient cycling and carbon storage that occurs due to beaver pond construction. The parafluvial/orthofluvial boundary conditions governing groundwater drainage may be reset by beaver, increasing the area of saturated soils and switching greenhouse gas sinks into sources. Greenhouse gas emissions from beaver dams should be weighed against historical rates of beaver damming and positive ecosystem functions from damming, such as increased riparian vegetation, increased water storage, and increased aquatic animal diversity.

Aquatic Biota

Weber et al. (2016) monitored fish population dynamics, beaver activity, and stream characteristics for three years before artificial beaver dam construction and four years post-restoration. They were specifically interested in whether the artificial beaver

dam installations would increase juvenile steelhead density, growth, and survival. They found that in restored reaches, fish density was 81 fish per 100 m greater, juvenile production was 175% higher, and juvenile survival was 52% higher relative to control reaches. These findings are echoed by Maia et al. (2018) where construction of instream fish habitat and BMS are linked to improved biotic integrity (37.5%), fish productivity, biodiversity, and sensitive taxa recruitment. These two fish-centric studies may be supported by BMR research showing increases in fish food. The unpublished thesis of James Holden Reinert found increases in aquatic insect abundance and density at BMSs on Long Creek (Reinert, 2020). Other studies have found a similar response in aquatic insects from natural beaver damming such as Roper et al. (2019) and Stenert et al. (2019).

Restoration Ecology and Ecological Restoration

Research has widely shown an array of ecosystem functions associated with natural beaver dam construction; however, beaver populations are still recovering from near extinction and human intervention may be a suitable approach for accelerating the restoration of fluvial and parafluvial ecosystem functions. Restoration of these functions is crucial for restoring the services we depend on, like clean drinking water and recreational activities like fishing. The 2017 EPA's National Water Quality Inventory states that 46% of US river and stream miles are in poor biological condition, citing phosphorus and nitrogen as the most abundant stressors. It also states that 21% of US lakes, ponds, and reservoirs are hypereutrophic (extremely rich in nutrients and minerals) and that 32% of US wetland areas are in poor biological condition (United States Environmental Protection Agency, 2017). Accelerating the restoration of fluvial and

parafluvial systems is essential for restoring the biological condition of our streams and rivers, but precaution towards the installation of the under-researched BMR is advisable (Kelleher et al., 2019) and may also benefit from the utilization of sound restoration ecology principles.

The study of restoration ecology advances the practice of ecological restoration through exploring the chemical, physical, and biological interrelationships of ecological systems altered by restoration practices (Galatowitsch, 2012). Restoration ecology focuses on understanding ecosystem functionality in the context of restoring deteriorated natural processes. However, proposed innovations of ecological restoration frequently outpace understanding of natural causality provided by restoration ecology research. BMS, Beaver Dam Mimicry (BDM), and Beaver Dam Analogs (BDA) are one such example. They are a relatively new aquatic ecological restoration practice that has been widely implemented for the last decade (Pilliod et al., 2018; Pollock et al., 2014).

A diversity of terminology has been used to describe BMR where the construction of artificial, permeable, low-head, and often hand-built dams (BMS) are strategically placed within an incised stream. BMSs are generally designed to hold back streamflow behind the structure, which consequently creates a zone of sediment deposition, elevates surface and groundwater levels, and ultimately re-engages the stream with the floodplain (Figure 2). A variety of construction materials are used and usually located on-site for lower budget restorations. They can consist of cedar posts, willow branches, alder branches, conifer branches, straw bales, native grasses, mud, and alluvial sediments (Figure 3).

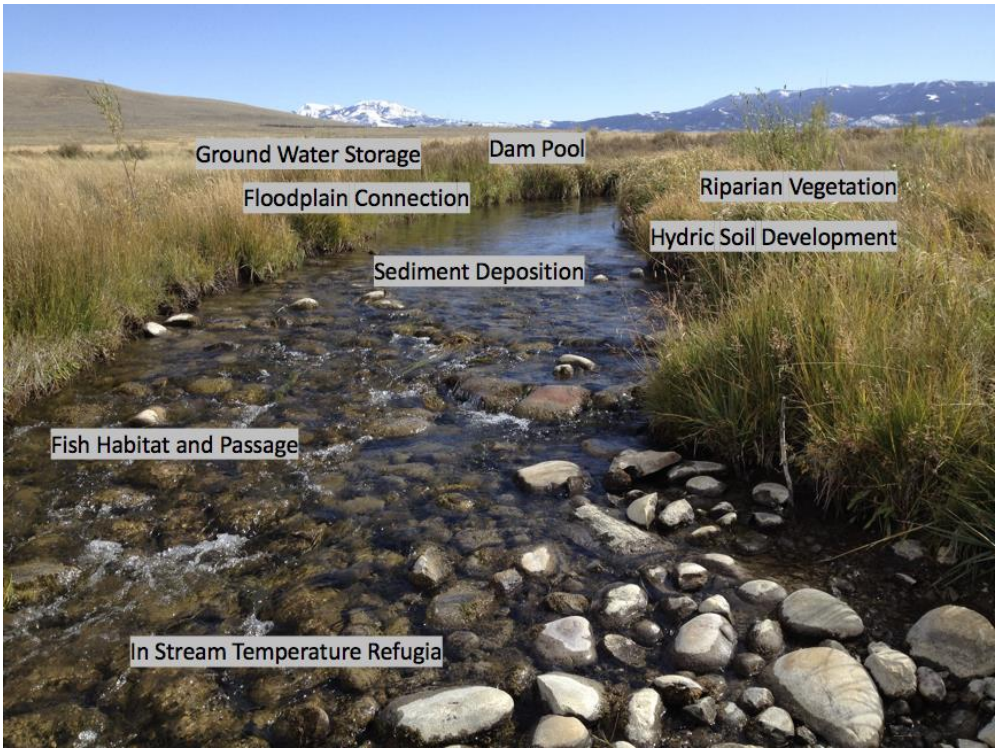


Figure 2: Beaver Mimicry Areas of Ecosystem Functions. 2019 Treatment location of Beaver Mimicry Structure. Annotating is showing some of the general locations of derived ecosystem functions.



Figure 3: Typical Beaver Mimicry Structure. NRCS wetland restoration project constructing double posted BMS in Northern ID. November 2019. The photo was taken by Briana K. Whitehead.

In general, the recovery of fluvial/parafluvial ecosystem functions depends on the restoration of their interaction. BMSs are designed to reconnect the historically beaver-connected fluvial/parafluvial functions of the stream corridor ecosystem, though the effectiveness of common BMR design strategies has been poorly studied (Levanoni et al., 2017; Weber et al., 2017). As Law et al. (2017) eloquently explained: “The major focus in restoration-oriented studies has been on their hydrological and geomorphic effects, with ecological benefits usually assumed rather than evidenced.” Figure 1 shows many ecosystem changes that have been sparsely researched and some that have been assumed

post-BMR. In particular, the current body of evidence showcases the need for additional BMS research directed at answering important ecosystem function-related questions.

Riparian Soil

Riparian soils have been widely attributed with provisioning the ecosystem functions on which the integrity of terrestrial and aquatic ecosystems in the stream corridor depend on (Gregory et al., 1991; Lewis et al., 2003). Soils are not only the physical and nutritional substrate for terrestrial plants, but they also provide a wide range of benefits to aquatic organisms through the filtration of surface water, bank storage of water, and nutrient cycling within interstitial sediment spaces. However, changes in the structure and function of riparian and parafluvial soils following restoration activities have rarely been monitored or quantified (Fournier et al., 2013; Unghire et al., 2011) leaving a gap in the understanding of restoration ecology in riparian systems.

An example of a riparian ecosystem function is soil carbon storage, which can be abundant, especially within hydric soils (Lewis et al., 2003; Richardson et al., 2001). Hydric soils are formed under conditions of saturation long enough during the growing season to develop anaerobic conditions in the upper part (Vaughan et al., 2016). During saturation and inundation events microbial aerobic respiration is first stopped, then slowed, and then switched to anaerobic respiration. (Goldman et al., 2017; Kirk, 2004; Pinto et al., 2020). This switch in respiration allows soil organic matter (SOM) to be metabolized at a slower rate, which results in longer-term soil carbon storage.

The few studies that have quantified the effects of various stream restorations on riparian soils have predominately focused on a handful of soil properties such as, SOM,

total carbon, total nitrogen, and plant available phosphorus (Hale et al., 2018). These measurements result in mixed conclusions, which may be attributed to slow pedologic processes or potentially because these soil properties are not the best measurements to quantify after restoration. Of the studies that have expanded outside of measuring soil carbon, nitrogen, and phosphorus, Celentano et al. (2017) designed a unique study that quantified morphological properties. They measured porosity, water content, bulk density, infiltration, and texture to better understand the impacts riparian deforestation (starting in the 1980s) may have on subsequent ecosystem functions. Although this novel study evaluates the influence of deforestation on riparian soils, rather than BMR restoration efforts, the results are informative of the timescales that may be relevant to the response of soils to restoration activities. They found soils responded to deforestation with 21% lower porosity, 82% lower soil water content, 36% higher bulk density, and a 91% lower infiltration rate compared to conserved riparian forests. Approximately 40 years of intermittent deforestation has drastically shifted these soils and should inform restoration practitioners of reasonable timescales for restoration to impact measurable soil physical attributes.

Soil morphology is fundamental to riparian ecosystem functions in that it allows for the transmission of organisms, nutrients, water, and allows for plant roots to move into the soil profile. Soil morphology can reportedly be quickly developed following restoration (Fournier et al., 2013). Using diversity (soil profiles), typicality (frequency of soil profiles), and dynamism (total number of horizons per meter) Fournier et al. (2013) found that restoration moved soil morphology towards a more transmissive structure, and

improved associated vegetation communities. The biogeochemical processes that occur within parafluvial soils (denitrification, carbon storage, and plant substrate) act as the foundational mechanism for many stream corridor ecosystem functions.

A study at Voyageurs National Park is a rarity in that it attempts to assess biogeochemical changes in riparian soils following beaver damming (Johnston et al., 1995). Although this study did not measure the effects of BMR, its findings are of the few characterizations of changes in soils after small-scale dam installations. The soils (a glaciofluvial hydrosequence) reportedly changed soil matrix colors from dark brown to dark grayish-brown and olive gray, with an increase in redoximorphic features after being submerged for 14 years (Johnston et al., 1995). This change in color indicates an eluviation of iron and manganese from the soil profile and is supported by several studies where soluble iron was found in higher concentrations in and below beaver ponds (Cirmo et al., 1993). This pattern implies a mineral redistribution from the parafluvial to ponds and streambed created by fluctuating water tables and subsequent reducing conditions.

The need for further investigation into how stream, river, and riparian restoration impacts adjacent soils has been noted throughout decades of exploratory research and monitoring. Several studies have successfully quantified changes in soils post-restoration (structure and infiltration) and while others have been limited by the slow-to-respond soil properties measured (carbon and nitrogen ratios). Restoration-induced soil moisture, temperature, and anaerobic reduction may be overlooked aspects and potentially underappreciated outcomes of restoration. These outcomes are likely the first soil properties to respond to restoration induced floodplain connectivity and vegetation

establishment. Over time soil moisture, temperature, and anaerobic reduction may further the capacity of riparian soils to respond to restoration through quantifiable changes in carbon, nitrogen, and phosphorus. My research will add to the scientific understanding of riparian soils post-restoration by monitoring these first response indicators (soil moisture, temperature and anaerobic reduction), and explore adequate methodology for monitoring these properties.

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CHAPTER TWO

PARAFLUVIAL SOIL RESPONSE TO BEAVER MIMICRY RESTORATION IN A MONTANE STREAM

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Introduction

Beavers are a keystone species in ecosystems across their range due to their unique interactions with the physical riparian environment (Puttock et al., 2017). They engineer fluvial systems to create hydrologic conditions that will increase the growth of forage, change the form of aquatic habitat, and establish shelter from predation and climate. Beaver dam and canal building behavior creates and maintains channel and subsurface hydraulics that support ecosystem functions including increased aquatic thermal refugia, increased water storage, sediment and nutrient storage, increased fluvial morphological heterogeneity, and increased riparian/wetland vegetation. These changes are generally related to desirable effects on the biodiversity and density of aquatic biota that are common goals in promoting reintroduction of beaver populations (Cummins et al., 2019; Dauwalter et al., 2018; Fairfax et al., 2018; Law et al., 2017; Pollock et al., 2007; Weber et al., 2017).

Before federal protections began in the early 20th century, North American beaver (*Castor canadensis*) populations neared extinction within their native range, which extended from the North American Arctic to northern Mexico (Johnston et al., 1988). Currently, North American beaver populations are recovering with a population on the order of 6-12 million, which is still a small fraction of the original estimated population of 60-400 million (Johnston et al., 1988; Seton, 1929). The loss of beaver, along with increased human land use and population growth, has resulted in the disconnection of many North American streams from their floodplains (Poff et al., 1997; Tockner et al., 2002). These disconnections have altered the integrity of the associated fluvial and

riparian systems, resulting in undesired changes in water quality, loss of wetlands, lowered groundwater tables, warmer or more extreme stream temperatures, lower base flow, reduced biotic diversity, and reduced habitat diversity (Pollock et al., 2007).

Efforts to reintroduce beaver have become widespread in the effort to restore floodplain connectivity in stream ecosystems (Pilliod et al., 2018). However, many efforts to reintroduce beaver have been unsuccessful in establishing viable long-term populations due to biological and ecological limitations (Fairfax et al., 2018; Gower et al., 2019). In particular, beavers are dependent on hydrophytic plants as their main food source and building material (Baker et al., 2003; Touihri et al., 2018). Landscapes lacking floodplain connectivity and adequate riparian groundwater height cannot support these fundamental plant communities. Therefore, beavers are often limited to areas with existing food and material sources (Pilliod et al., 2018), and intervention is necessary to restore floodplain connectivity and subsequent riparian vegetation to encourage beavers to recolonize degraded stream habitat (Charnley et al., 2017).

Installation of Beaver Mimicry Structures (BMSs) is a common strategy employed to restore stream habitat to a state suitable for beaver reintroduction or recolonization. Beaver Mimicry Restoration (BMR) is intended to restore the structure and function of the stream corridor ecosystem to a state similar to that created by beaver activity (Table 1), including higher biotic diversity (Maia et al., 2018), restored native woody riparian vegetation (Allred et al., 2019), and increased sediment retention and bed aggradation (Korpak et al., 2019).

Table 1: A summary of select studies that investigated the influence of beaver and/or BMR activity on ecosystem structure and function.

Ecosystem Property	**Specific focus**	**Supporting Research**
Biotic Diversity	• Fish assemblage • Steelhead (*Oncorhynchus mykiss*) population • Beetle and plant biodiversity	• Maia et al. (2018) • Weber et al. (2016) • Levanoni et al. (2019)
Drought Resistance	• Riparian evapotranspiration • Pasture health and productivity	• Fairfax et al. (2018) • (Charnley et al., 2017)
Nutrient Cycling	• GHG emissions • Nutrient flux and ecosystem metabolism	• (Lazar et al., 2014) • (Wegener et al., 2017)
Riparian Vegetation	• Vegetation establishment • Semi-arid vegetation productivity • Riparian plant recruitment • Plant species richness	• (Pollock et al., 2014) • (Dauwalter et al., 2018) • (Meyer et al., 2019) • (Wright et al., 2002)
Sediment Deposition	• Pond fluvial processes • Storage and biogeochemistry of sediment • Discharge and sediment transport • Geomorphic changes upstream • Sediment and nutrient storage	• (Giriat et al., 2016) • (Fox-Dobbs et al., 2019) • (Korpak et al., 2019) • (Pollock et al., 2007) • (Puttock et al., 2018)
Stream Morphology	• Biotic drivers of stream planform • Beaver assisted valley formation • Discontinuity of fluvial systems • Minor and major beaver-initiated stream complexities	• (Polvi et al., 2013) • (Westbrook et al., 2011) • (Daniels et al., 2014) • (Cunliffe et al., 2015)
Water Storage	• Groundwater ad sediment aggradation • Surface water and storage capacity • Water storage, attenuation of flow, and mitigates pollution • Hydrological processes and water quality. • Riparian zone hydrology	• (Allard et al., 2018) • (Melanie et al., 2018) • (Puttock et al., 2017) • (Tetzlaff et al., 2020) • (Vidon et al., 2020)
Water Temperature	• Hydrologic and temperature regimes • Alteration of stream temperature	• (Majerova et al., 2015) • (Weber et al., 2017)

BMSs almost certainly also influence the development of upstream riparian soils, but research on the hydrologic and biogeochemical response of riparian soils to BMR has been limited to date. The primary mechanism by which beaver or BMR activity influence soils is likely via changes in the hydrologic inundation regime, capillary action, and the availability of water for plant roots. Therefore, BMR is likely shifting soil and plant resources that may impact ecosystem function.

Continuously or intermittently submerged soils play a substantial role in food production for beavers and other species, habitat for wetland communities, and stream corridor nutrient cycles (Kirk, 2004). Regular inundation of riparian soils has been shown to provide many stream corridor functions often deemed valuable, such as carbon sequestration, water purification, flood and erosion control, biodiversity, water storage, and water temperature regulation (Fournier et al., 2013; Glanville et al., 2018; Lewis et al., 2003; Unghire et al., 2011; Wohl et al., 2017). The importance of parafluvial soil to the functions and provided by riparian ecosystems is widely recognized, yet few studies have shown how the hydrologic and redox chemistry regimes of soils might change in response to fluvial restoration practices (Hale et al., 2018), including BMR.

The objective of this research is to assess the influence of BMS installations on adjacent parafluvial soils due to the increase in hydrological connections between the channel and floodplain. This objective is addressed with three specific research questions: (1) “How does Beaver Mimicry Restoration affect parafluvial soil water content and temperature regimes?”, (2) “How do BMS installations influence conditions that are conducive to anaerobic processes in upstream soils?”, and (3) “How does the elevated

water table impact riparian vegetation phenology?" This research is a first attempt at measuring the impacts BMS may have on parafluvial soil properties, and results are discussed with the intent of furthering ecological restoration.

Study Area and Methods

Study Area

The Centennial Valley is located in Beaverhead County and is surrounded by the Snowcrest, Gravelly, and Centennial Mountain ranges in southwest Montana, USA (Figure 4). The study area within the Centennial Valley is a 4 km reach of Long Creek within The Nature Conservancy's Centennial Sandhills Preserve (44° 40' 11.98" N, 112° 6' 19.87" W). Long Creek is a third-order stream (Strahler, 1957) that flows south out of the Snowcrest and Gravelly Ranges, continues through the western side of the Centennial Valley, and drains into the Red Rock River upstream of the Lima Reservoir. Long Creek's average of daily flow between 2016-2018 is approximately 0.31 ± 0.01 CMS (Reinert, 2020).

Within the study area, Long Creek's mean thalweg elevation is 2026 m and most resembles a Type E stream (Rosgen, 2007) characterized by low gradient and high sinuosity (Miller et al., 1996). In 2018, Long Creek was described as entrenched with an inset floodplain, highly sinuous (2.5), low gradient (<1%), and lacking floodplain connection (Boyd et al., 2018; Brummond, 2018). By Rosgen Stream Classification the average bankfull width-to-depth ratio is very low (2.74) and the average entrenchment ratio (ratio of flood-prone area to bankfull width) is moderately entrenched (1.64).

The Long Creek watershed once drained into a glacial lake (ca. 10,000 ya), such that the surrounding substrate shifts from alluvial sediments in the north to fine-grain lake bottom sediments in the south (Whitlock et al., 2012). Floodplain soils surrounding Long Creek are predominantly mapped by the USDA NRCS as the Crookedrun series (45%)

and classified as Oxyaquic Haplocryolls. They host a ustic soil moisture regime, a cryic soil temperature regime, and alluvial parent material with a fluctuating water table. Long Creek floodplain soils are also mapped as Vertic Cryaquolls and Ustic Haplocryolls, with series names of Narrows (30%) and Trailhollow (15%), respectively (USDA NRCS, 2020b).

Floodplain vegetation consists of a mixture of native sedges, rushes, and shrubs, including upland grass and shrub species on the upper floodplain terrace (Whitlock et al., 2012). The parafluvial zone is dominated by riparian species, such as the rush *Juncus balticus* and the sedge *Carex nebrascensis*. A mixture of native and non-native perennial grasses are interspersed among the riparian plants from historic grazing practices.

The mean annual precipitation is approximately 286 mm and the mean annual air temperature is 3.9 °C (USDA NRCS, 2020b). A frost-free period of approximately 50 to 70 days between June and September provides a short growing season for local vegetation (USDA NRCS, 2020a). Long Creek's elevation, climate, and growing season were used to define the field season for this research.

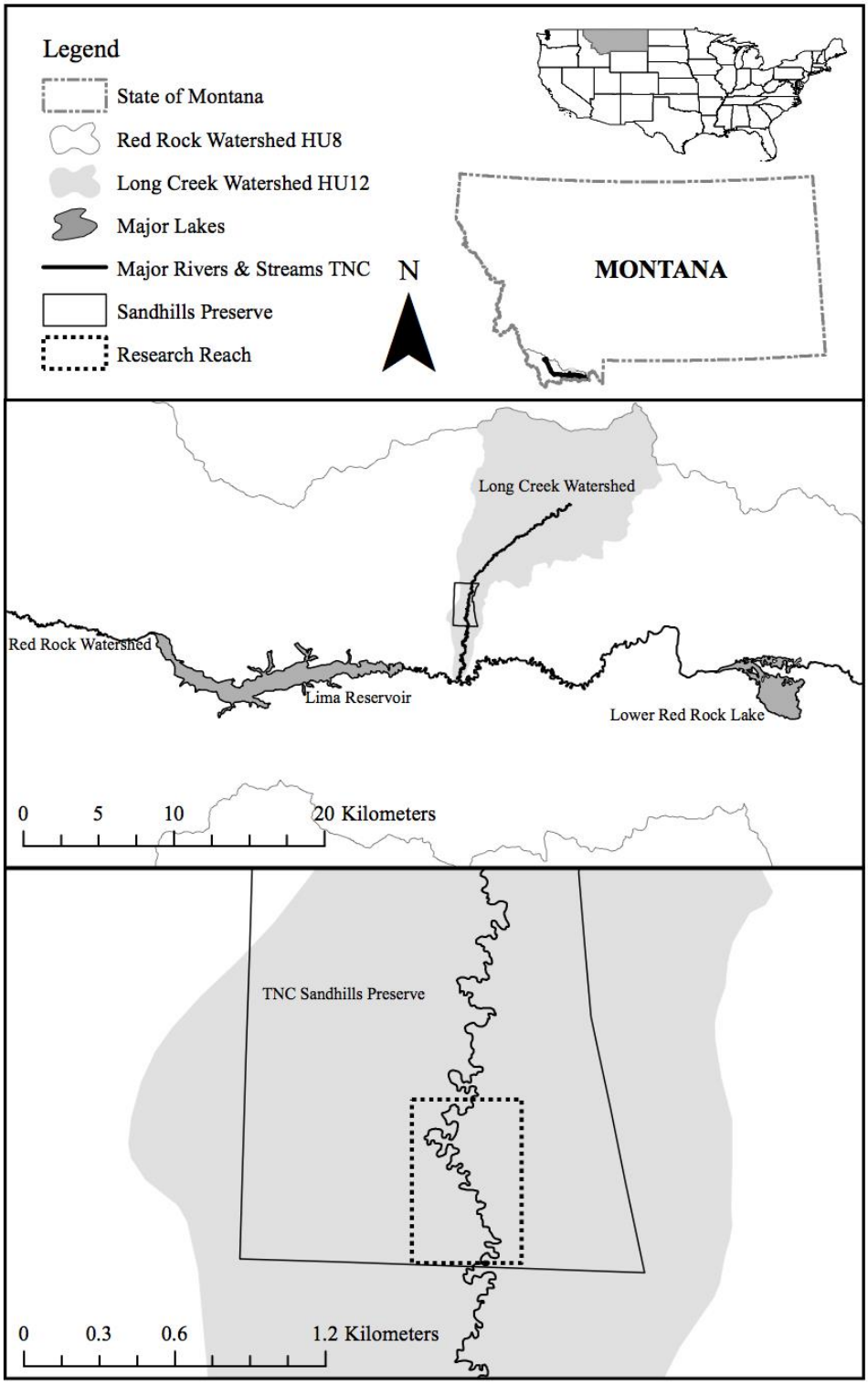


Figure 4: Location of the Red Rock Watershed containing The Nature Conservancy's Centennial Sandhills Preserve on Long Creek, Montana, USA.

Site History and Selection

Historic management of rangeland along Long Creek has resulted in the extirpation of beaver, reduction of connections between the stream channel and floodplain, and altered riparian vegetation (Boyd et al., 2018). The Nature Conservancy (TNC) took ownership of the study reach in 2009 and stream restoration projects began in 2011. TNC restoration objectives were to reconnect the incised channel to portions of the floodplain during peak flows and raise groundwater levels underlying the floodplain during base flow (Gillilan et al., 2015). To meet these objectives, TNC installed nine artificial stream riffles designed to act as BMSs over 1.5 km of stream in the summer of 2016 (Gillian, 2015). Constructed riffles were chosen over more discontinuous forms of beaver dam analogs to allow passage for migration of arctic grayling (*Thymallus arcticus montanus*) (Gillilan et al., 2015).

Each structure was designed to elevate the stream stage ca. 0.6 m above the existing streambed at the crest of the constructed riffle (Figure 5). All substrate material was extracted from on-site resources. Riffles were built to elevate the stream bed using a target composition of ca. 20% 256-64 mm diameter material (cobble), 20% 64-32 mm very coarse gravel, and 60% <32 mm gravel/sand/silt (Figure 2). The surfaces of streambed riffles were armored with approximately 19 cubic meters of > 76 mm cobble.

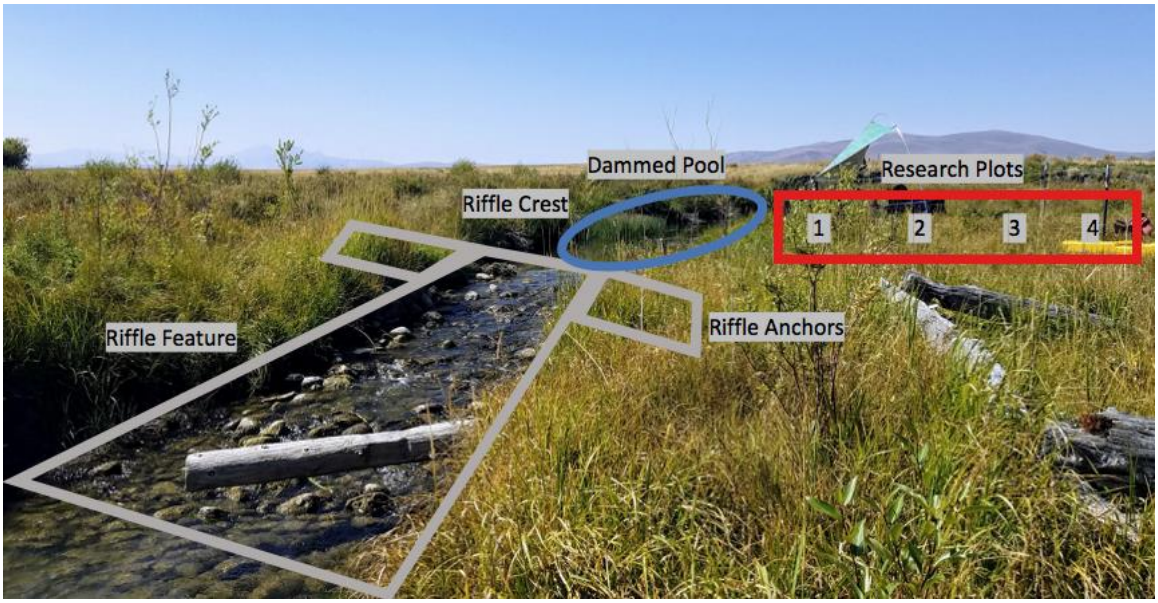


Figure 5: Photo of a constructed riffle near the 2018 Treatment monitoring location. The photo was taken at low flow in September looking upstream. Logs in the frame are debris from historic ranch buildings and are not part of the restoration.

For each of the 2018 and 2019 field seasons, I selected a pair of stream meanders representing a chronosequence of restoration: one immediately adjacent to a BMS representing the post-installation "treatment" state (T18 and T19), and one far from a BMS representing the "control" pre-installation state (C18/19; Figure 6). The proximity of the pre- and post-installation sites (averaging < 280 m apart) is designed to minimize the influence of confounding variables (such as elevation, weather, and general substrate composition) in conclusions regarding the influence of BMS on differences in soil properties between the sites.



Figure 6: Meander specific treatment and control locations of field installations for the 2018 and 2019 field seasons.

The T18, T19, and C18/19 stream meanders were surveyed to compare the stream bank and bed topography (Figure 7). T18, T19, and C18 meander lengths (channel distance between the apex of two sequential meanders) were similar (T18 = 131 m, T19 = 108 m, C18/19 = 124 m), as well as their associated meander length ratios (stream meander length divided by bankfull width) (T18 = 18, T19 = 15, C18/19 = 24). Sinuosity (ratio of thalweg length to valley length) measurements were also relatively similar among treatment locations (T18 = 5, T19 = 6) while the control had a larger sinuosity (C18/19 = 24), as were meander wavelengths (linear distance along the valley gradient between the apex of two sequential meanders) (T18 = 32 m, T19 = 50 m, and C18/19 = 39 m).

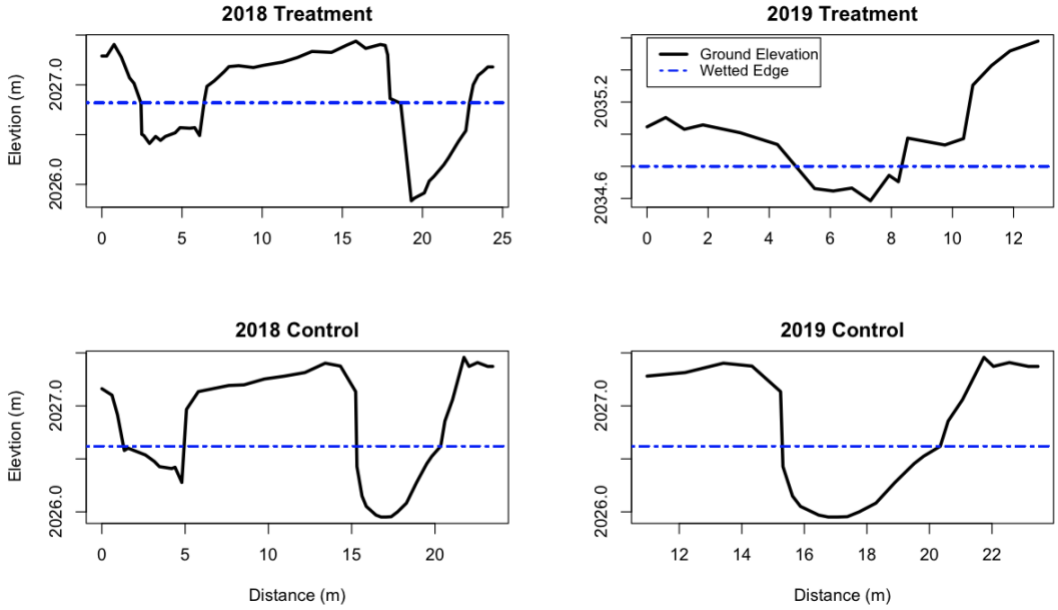


Figure 7: Stream cross-sections from 2018 and 2019 treatment and control stream meanders.

To contrast 2018 results with a treatment site with differing vegetation, I selected an alternative treatment meander (T19) paired with the same control meander (C18/19). I selected T19 due to its similarity to C18/19 fluvial morphology and plant community, which hosts more facultative species compared to the wetter T18. The T18 BMS is located on the downstream section of the meander. Therefore, the T18 site was potentially affected by water level increases on both the upstream and downstream sides of the meander. This contrasts the drier T19 BMS location which is on the upstream end of the meander potentially only elevating the water level on the upstream side of the meander.

Field Instrumentation

An array of response indicators expected to capture the effects of floodplain connectivity on soil properties were measured across each T18, T19, and C18/19 meander. The orientation of each array was installed to capture a gradient of responses to the BMS (Figure 8 & 9) based on the presumed gradient in the elevated water table approximately parallel to hyporheic flow paths. During the field seasons of 2018 and 2019, the arrays were installed on the inside bend of each meander.

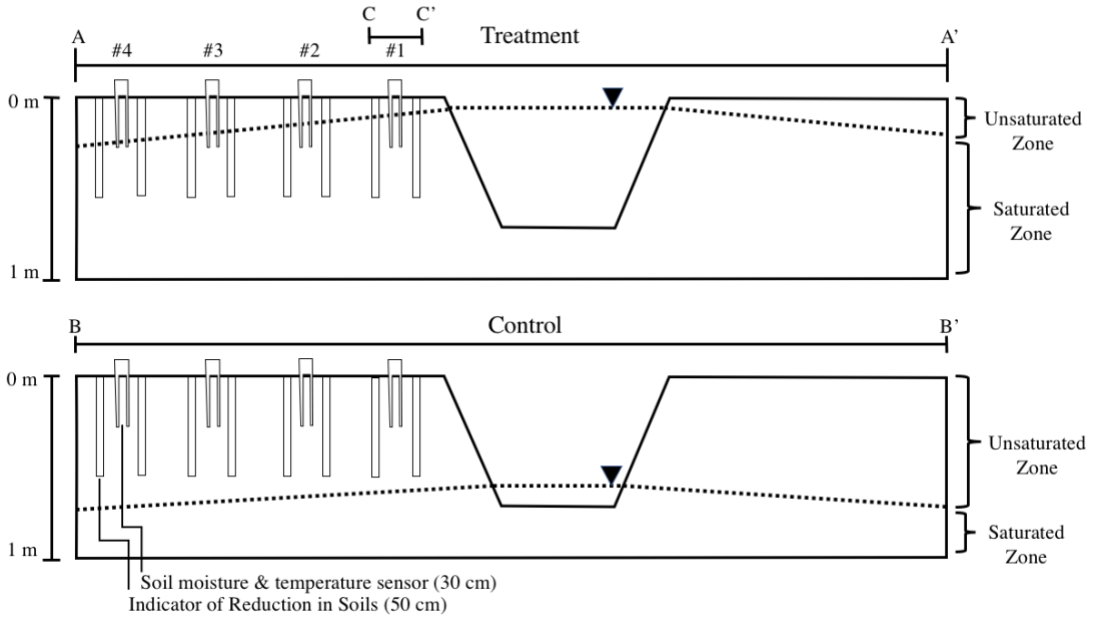


Figure 8: Field site installation diagram showing 2018 installation at the treatment and control stream cross-sections. The 2019 installation was reduced to three soil sensors and three plots of Indicators of Reduction in Soils (IRIS) films. The dotted line represents the hypothetical water table.

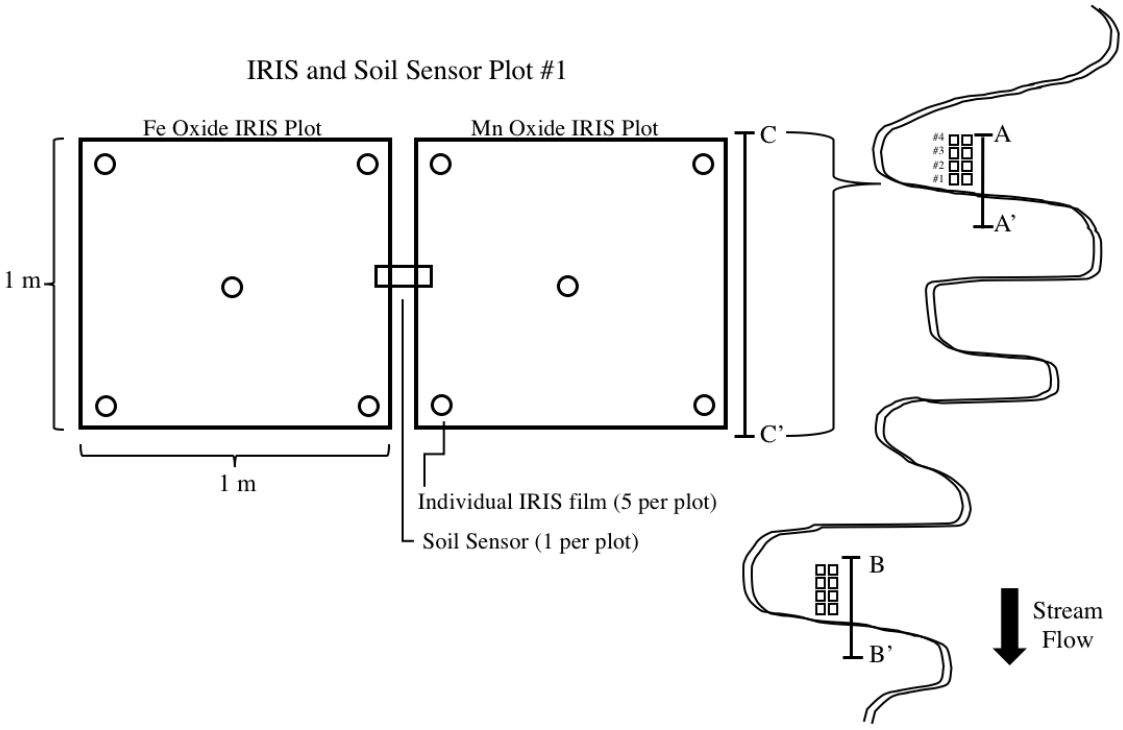


Figure 9: Field installation plot design in 2018. 2019 field installation was reduced to three soil sensors and only three sets of Mn IRIS. In 2019, soil sensors were installed within the center of each Mn IRIS plot.

Indicators of Reduction in Soils (IRIS) are a 50-cm by 7.5-cm strip of white polyurethane film coated with iron oxide or manganese oxide paint (Martin Rabenhorst et al., 2006). They were installed vertically in the soil profile within a pilot hole created by a one-inch diameter soil probe. Under anaerobic conditions, IRIS films are transformed by providing soil microbes with a terminal electron acceptor for anaerobic respiration, which reduces the iron or manganese oxide paint into a more soluble form and thereby mobilizes it away from the film. Therefore, bare areas on the film provide a time-aggregated and depth-specific indicator of regions in the soil that experienced anoxic conditions over the course of the IRIS deployment. Because manganese is a more thermodynamically favorable terminal electron acceptor compared to iron (Martin

Rabenhorst et al., 2018; M. C. Rabenhorst et al., 2017), the installation of both types of IRIS films in 2018 provided insight into how Fe and Mn coatings may provide different perspectives on the degree of reducing conditions experienced across each study site.

In 2018, five iron-oxide-coated films were installed within a 1-m^2 plot in addition to five manganese-oxide-coated films within an adjacent 1-m^2 plot along the same east-west axis (i.e. approximately the same distance from the stream). In total, each plot of five films gives a soil surface interaction area of 1875 cm^2 to a depth of 50 cm. A total of four paired Fe-Mn film installations spanned the north-south neck of the stream meanders, creating an array of IRIS films designed to capture any small-scale gradients in floodplain connectivity (see Figure 9) with distance from the channel. Each film was deployed for a total of four weeks before it was removed from the soil and a fresh film was installed for the next four weeks. This sequence of installations occurred from June through August and totaled 240 films in 2018. In 2019, the IRIS installation was reduced to three plots of the more anaerobically sensitive Mn Oxide films at each treatment and control location, and films were replaced once each month. In total, three deployments of IRIS films (90 films total) were deployed from June through August 2019 (Appendix A Tables 3-4).

Each film was cleaned, dried, and scanned in an A Contex IQ Quattro scanner to create digital images for pixel analysis. Binary images were created by delineating the zones of removal (white) and the zones of no change (orange or brown) to create a black and white image where removal zones are black, and zones of no change are white. Black pixels were then summed by vertical centimeter increments on each film (resized to 2,500

pixels) with the use of code written in Matlab (Martin. Rabenhorst, 2018) and compared against results from the R package Imager (Barthelme et al., 2020). Each centimeter-increment of the film was then assigned a percentage of black pixels, which was used to calculate a total film removal percentage and a removal percentage by depth. These calculations enabled an assessment of anaerobic soil reduction percentage of each film and whether those percentages met the National Technical Committee for Hydric Soils (NTCHS) Standard for Hydric Soil (National Technical Committee for Hydric Soils, 2015) (Figure 10). The analysis included depth-specific statistics of portions of the films and full film anaerobic reduction percentages. Welch's Two Sample t-tests and ANOVA were used to assess differences between treatment and control anaerobic reduction percentages derived from the full depth of the films.

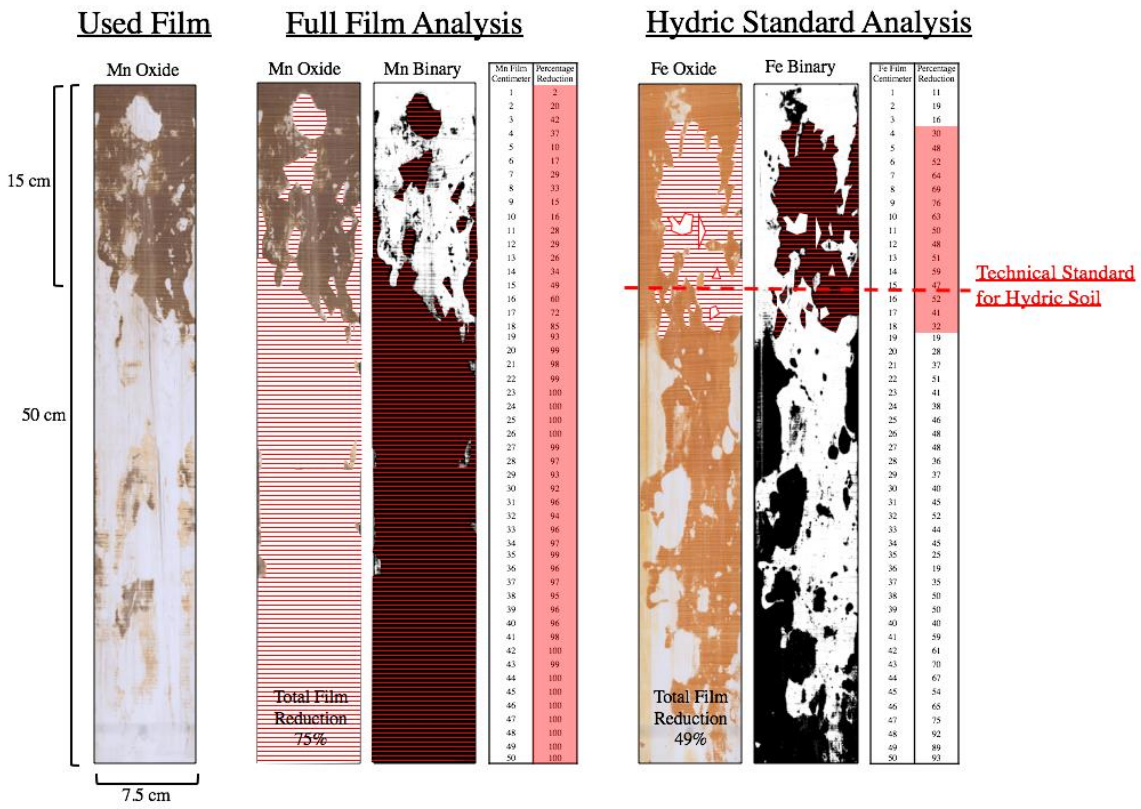


Figure 10: A typical example of Fe oxide and Mn oxide IRIS films after deployment. The binary images are analyzed by one cm vertical increments (1-50 cm) and percent removal is then used to assess Full Film reduction percentage and the NTCHS Standard for Hydric Soil (National Technical Committee for Hydric Soils, 2015): A minimum of three of five Indicator of Reduction in Soil (IRIS) within a 1-m by 1-m plot must have 30 percent iron/manganese removed from a zone 15 cm (6 in) or thicker. The zone of removal must begin within 15 cm of the soil surface for all soil textures.

In addition to IRIS films, arrays of time-domain reflectometry soil water content and temperature sensors (CS650, Campbell Scientific Inc., Logan, UT, USA) were installed in the same locations as IRIS installations (4 sensors in 2018 and 3 sensors in 2019, Figures 5 & 6). Each soil sensor was installed vertically and approximately 2 m apart, integrating water content measurements in the top 35 cm of soil. Measurements included soil water content (SWC), soil temperature (T), soil electrical conductivity (EC), and relative dielectric permittivity within a zone of influence of a 7.5 cm radius around the probe and 4.5 cm below the probe (7800 cm^3). The basic function of the TDR probes was checked before each field season based on measuring saturated conditions in clean sand, and the default CS650 calibrations for estimating water content from electrical permittivity were used for all measurements.

A Campbell Scientific data logger (CR850) was used to scan the array of soil sensors every sixty seconds, average the measurements for each sensor over an hour, and record that average. Additionally, a tipping bucket Campbell Scientific Metric Rain Gage (24.5-cm orifice, TE525MM-L) was installed with the control datalogger to monitor local precipitation events. The rain gage measures in 0.1 mm increments and is accurate to 1% for rainfall intensities up to 50 mm/h. Precipitation, soil moisture, and soil temperature

analysis were conducted in Rstudio with the use of the time-series package zoo (Zeileis A, 2005) and xts (Ryan et al., 2020).

To compare belowground measurements with an above-ground vegetation response, an automated camera (TimelapseCam, Wingscapes, EBSCO Industries, Inc., Birmingham, AL, USA) was installed at the control and treatment meanders during both field seasons. The cameras were mounted 5 m from the IRIS/soil sensor instrumentation array, approximately 1 m off the ground, and the lens was focused at a range of 5.2 - 9.1 m. Cameras captured a 2560 x 1920-pixel image of the stream meander vegetation every hour from 09:00 am to 06:00 pm local daylight savings time June through September. In total, ~1,000 color (RGB) images were collected from each control and treatment locations every field season. The R package Phenopix (Filippa et al., 2016) was used to process the images, through a standardized workflow.

Photos were preferentially selected at 10:00 am US mountain daylight savings time for each day due to better illumination and less variable localized climatic conditions. If an individual day's 10:00 am photo was poor, the closest photo with adequate illumination was selected. Photos were then loaded into R for Phenopix processing. A region of interest (ROI) was drawn to show Phenopix where the vegetation is located. In this case, the ROI was drawn on top of the IRIS and soil sensor array (Supplement Material Figure S1). The Phenopix analysis used digital numbers for red, green, and blue bands from each image set (T18, C18, T19, and C19) to calculate green chromatic coordinate (GCC) values for each ROI per picture. The GCC values were then

used in several phenological statistical models (primarily Klosterman and Spline) to find the best model fit for changes in plant phenology over time.

Plant Community and Soil Profile Characterization

Local plant community composition was surveyed at treatments and control locations to assess wetland status. Percent cover was established with the use of a one-meter square frame divided into one percent (100 cm^2) grid and laid on top of each 2018 and 2019 soil sensor monitoring plots. Random selection of frame placement was not used because the plant community composition associated with the monitored soils was needed. Each plant was categorized by family and a specimen was collected for in-lab identification to species. The wetland status of each species was determined by using the USDA plant database (USDA, 2020). Hydrophytic vegetation Indicators 1, 2 and 3 from The Regional Supplement to the Corps of Engineers Wetland Delineation Manual: Western Mountains, Valleys, and Coast Region (Version 2.0) were used to assess the plot-specific wetland status (United States Army Corps of Engineers, 2010).

Soil profiles were characterized by soil pits dug in September 2018 and September 2019. Each soil pit was approximately 40 cm^2 and pit depth was determined by the depth to the parent material. Soil color, structure, texture, effervescence, pH, percent rock, and redoximorphic features of each horizon were recorded using the Natural Resource Conservation Service technical standards. A sequence of soil pit and soil cores were characterized across each meander to assess floodplain soil variation.

In total, three pits were dug at each treatment and control meander. Each 2018 and 2019 treatment soil profiles were similar across their respective treatment meanders, and

control soil profiles consistently similar at the control location. Each pit respective to its treatment or control location was assessed to make a single aggregated soil profile description, in addition to a single Upland Terrace soil profile description which helped in understanding the landscape pedology.

Results

Soil Taxonomy and Character

The taxonomic classifications for T18, T19, and C18/19 soils are Typic Cryaquolls, while the cut bank exposure on the upper floodplain terrace is a Calcic Argicryolls. Riparian soils across all treatment and control sites are considered hydric by meeting the USDA NRCS field indicators A11 - Depleted Below Dark Surface, F3 - Depleted Matrix, F6 - Redox Dark Surface, and F12 - Iron-Manganese Masses (Vasilas et al., 2018). The upland terrace soil met none of the field indicators for hydric soils.

Table 2: Soil pit characterizations from T19, T18, C18/19, and the upper terrace. Soil descriptions utilize standard nomenclature from the NRCS Soil Survey.

2018/2019 Control			Taxonomic Class: Typic Cryaquolls					
Horizon	**Depths (cm)**	**Efferv**	**Structure**	**Color (Moist)**	**pH**	**Texture**	**% CF**	**Redox/Notes**
A1	0-8	VE	2VFGr	7.5YR 3/2	8.2	Silt Loam	0	NA
A2	8-16	VE	2FGr	10YR 3/2	8.2	Silt Loam	0	7.5 YR 4/4 (Fe) 3%
Bkg1	16-30	VE	3M Platy	10YR 3/1	8.2	Loam	1	10YR 3/6 (Fe) 10YR 2/1 (Mn) 3%
bkg2	30-51	ST	1MSBK	10YR 3/1	8	Sandy Clay Loam	0	10YR 3/6 (Fe) 10YR 2/1 (Mn) 2%
Cg1	51-74	SL	Massive	2.5YR 3/1	8	Sandy Loam	5	10YR 3/6 (Fe) 10YR 2/1 (Mn) 20%
Cg2	74-90	VS	Single Grain	5Y 3/1	7.8	Loamy Sand	40	10YR 3/6 (Fe) 10YR 2/1 (Mn) 3%
								Water table 90 cm

2018 Treatment			Taxonomic Class: Typic Cryaquolls					
Horizon	**Depths (cm)**	**Efferv**	**Structure**	**Color (Moist)**	**pH**	**Texture**	**% CF**	**Redox/Notes**
A1	0-7	SL	1FGr	10YR 2/2	8.2	Silt Loam	0	NA
A2	7-15	ST	1FGr	10YR 3/1	8	Loam	0	5YG 4/6 (Fe) 2% 10YR 2/1 (Mn)
Bkg	15-34	ST	2FSBK	10YR 3/2	8	Loam	0	5YG 4/6 (Fe) 2% 10YR 2/1 (Mn)
Bkg	34-52	SL	2FSBK	YR 4/1	7.8	Loam	0	5YG 4/6 (Fe) 10YR 2/1 (Mn) 8%
C1	52-77	SL	Single Grain	10YR 4/1	7.4	Sandy Loam	3	5YG 4/6 (Fe) 10YR 2/1 (Mn) 15%
C2	77-90	VS	SIngle Grain	2.5Y 4/1	7.4	Sandy Loam	50	Soup/Slop
								Water Table 68 cm

2019 Treatment			Taxonomic Class: Typic Cryaquolls					
Horizon	**Depths (cm)**	**Efferv**	**Structure**	**Color (Moist)**	**pH**	**Texture**	**% CF**	**Redox/Notes**
A1	0-9	ST	1VFGr	10YR 2/1	8.2	Silt Loam	0	7.5Y 4/6 1% 10YR 2/1 (Mn)
A2	9-22	ST	2MGr	10YR 3/2	8.2	SIlt Loam	0	7.5Y 4/6 10YR 2/1 (Mn) 2%
Bkg	22-54	VE	3MSBK	2.5Y 3/2	8	Sandy Clay Loam	2	7.5Y 4/6 10YR 2/1 (Mn) 2%
bkg	54-76	VE	3MSBK	2.5Y 3/2	8	Loam	50	7.5Y 4/6 10%
C	76-90	VE	Single Grain	10YR 4/2	7.8	Loamy Sand	30	7.5Y 4/6 5%
								Water table 90 cm

Upland Terrace (cut bank exposure)				Taxonomic Class: Calcic Argicryolls				
Horizon	**Depths (cm)**	**Efferv**	**Structure**	**Color (Moist)**	**pH**	**Texture**	**% CF**	**Redox/Notes**
A	0-13	SL	1M Granular	10YR 2/1	8.2	Loam	0	NA
Btkn	13-30	SL	3M prismatic p/t ABK	10YR 3/2	9	Clay	0	80% CLF PFP Ph purple.
Btn/E	30-47	ST	3F ABK	10YR 5/2 & 10YR 4/2	9	Silty clay	0	50/50 individual distinct material, texture mix of two. 20% CLF PFD. Ph purple.
Btn	47-96	VE	2VF SBK	2.5Y 6/1	9.6	Clay loam	0	Very fine sand. 20% CLF PF F. Ph purple.
Bk	96-128+	ST	2F SBK	5Y 5/2	8.4	Clay loam	0	Ph purple. Sodium likely.
								Water table 128 cm

Soil Water Content and Precipitation

Total annual precipitation from the nearby Divide SNOTEL site (NRCS SNOTEL, 2020) reported 2018 and 2019 accumulative precipitation of 63.5 cm. The watershed has a snowmelt dominated hydrologic regime that received 76% of long-term average snow accumulation in 2018 and 104% in 2019. Local rainfall at the research reach (see Figure 6) measured during the 2018 field season (96 mm) was similar to that of 2019 (92 mm). The frequency of rain events differed between the field seasons, with 29 days of measurable precipitation in 2018 and 44 days of measurable precipitation in 2019. The difference in precipitation regime between the two years is most evident in the difference between the average precipitation per event (2018 = 0.82 mm, 2019 = 0.72 mm).

On 6 June of the 2018 field season, the average SWC across the T18 transect was 0.60 m_3/m_3, which was nearly sustained into mid-August with an average on 18 August of 0.57 m_3/m_3. The control data were markedly different with a transect average of 0.54

m^3/m^3 on 6 June decreasing to 0.28 m^3/m^3 by 18 August (Figure 8). By the end of the 2018 field season (September 8) the difference between the average SWC at the T18 and C18 transects had increased to 0.29 m^3/m^3.

The 2019 field season showed similar trends in soil water content, however; the differences between T19 and C19 were less pronounced. On 6 June 2019, the average SWC across T19 transect was 0.55 m^3/m^3, similar to C19 at 0.54 m^3/m^3. Soil water content at T19 and C19 diverged over time and by 18 August T19 (0.51 m^3/m^3) was consistently higher than C19 (0.41 m^3/m^3). The difference between T19 and C19 transects by the end of the 2019 field season (8 September) had increased to 0.12 m^3/m^3 (Figure 11).

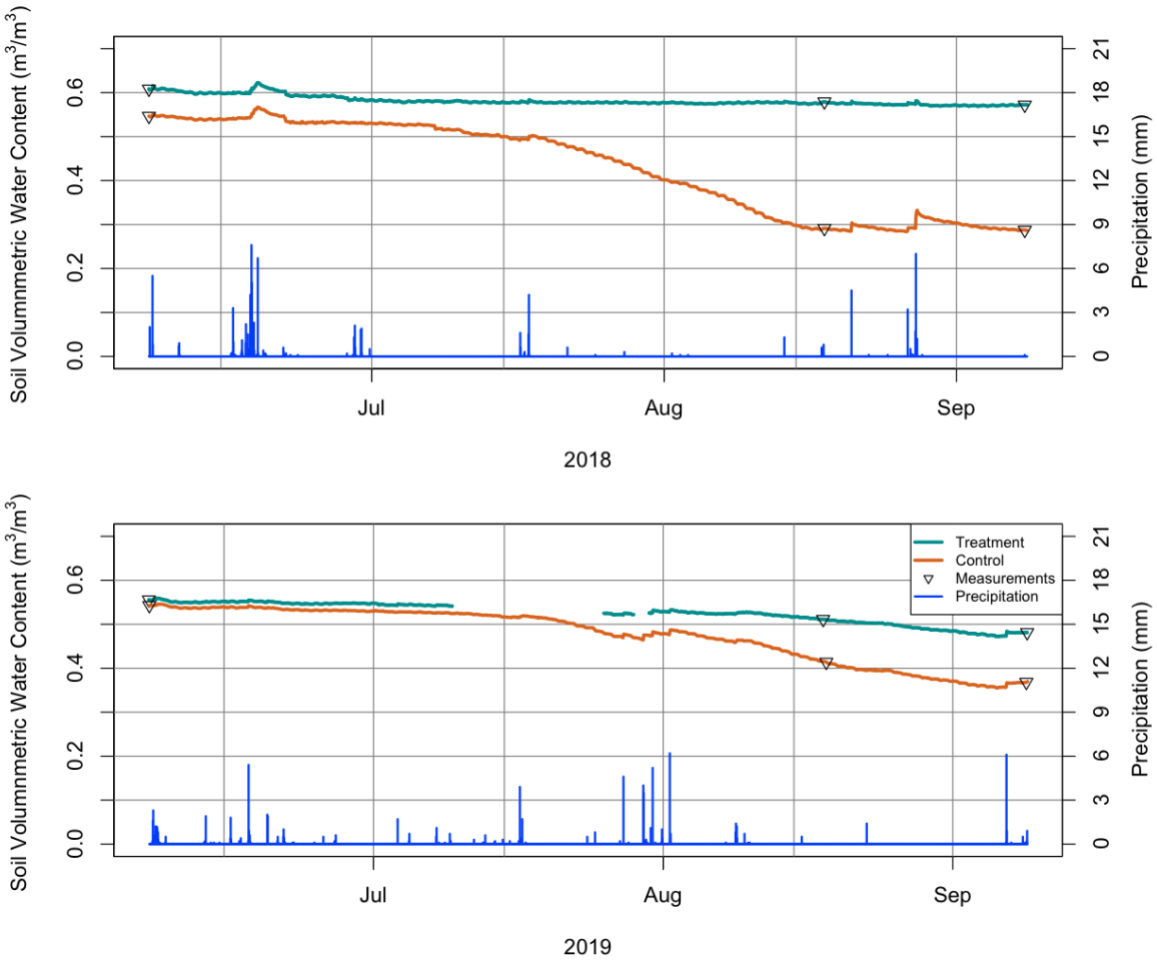


Figure 11: Hourly soil volumetric water content (SWC) and precipitation during 2018 (top) and 2019 (bottom) study periods. Total precipitation was 96 mm during the 2018 study period and 93 mm during the 2019 study period. Treatment data were unavailable from July 9th - July 24th, 2019 due to equipment failure.

Soil Temperature

Spatially aggregated soil temperatures at the treatment sites were, on average, 5° C cooler than the control site during the driest part of the field season (Figure 12). Overall, T18 and T19 soils experienced an average field season daily maximum temperature of 19° C, while C18 and C19 field season daily maxima temperate averaged 22° C and 20° C respectively. Average field season daily minimum soil temperature was consistent across all locations, T18, C18, T19, and C19 (7° C).

The average field season daily soil temperature range (max-min) also differed between treatment and control sites where T18 and T19 (12° C and 11° C) showed less variability compared to C18/19 (average 15° C and 13° C). In August, the differences between daily temperature ranges became most evident, when both T18 and T19 averaged 11° C and the control averaged 17° C in 2018 and 15° C in 2019.

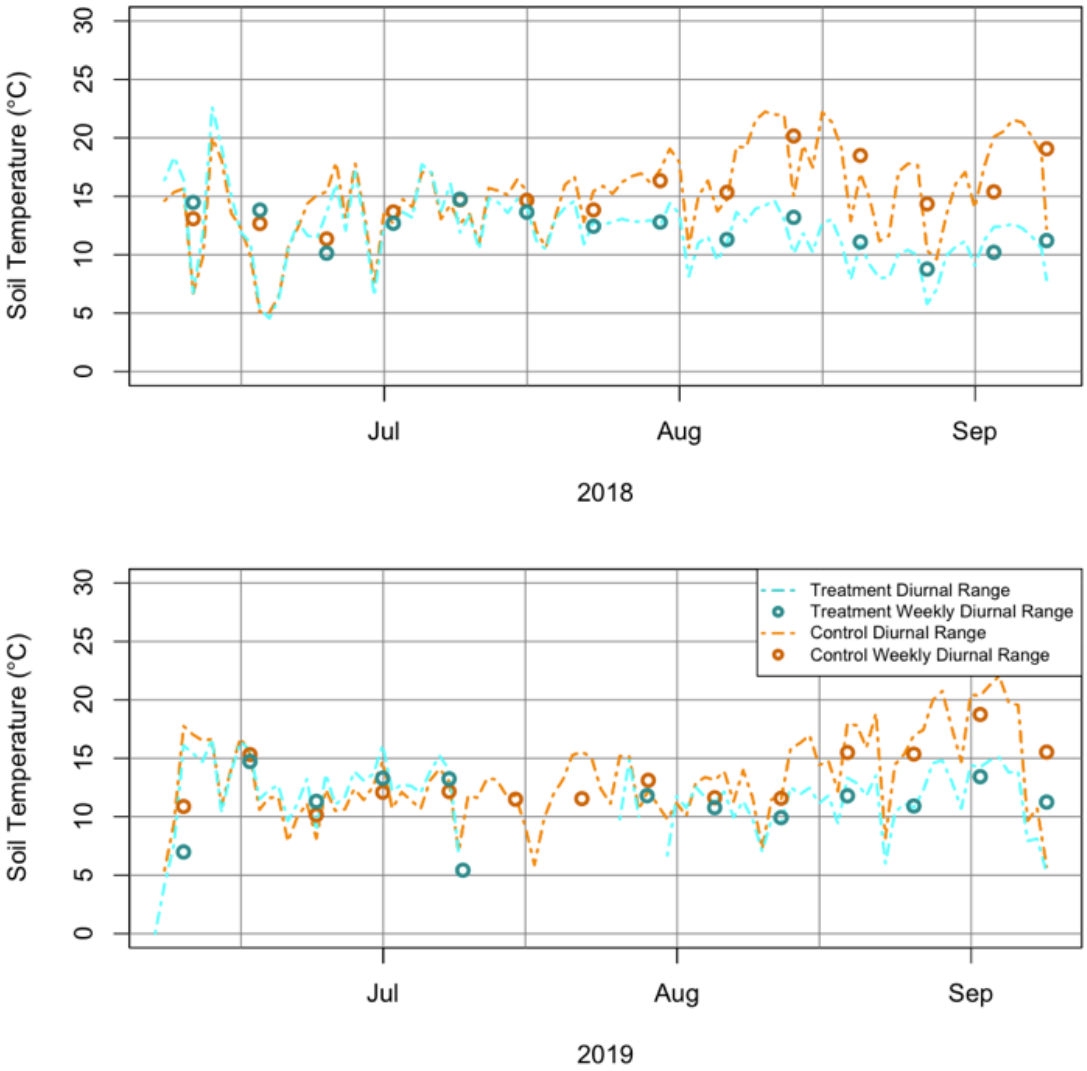


Figure 12: Spatial average of soil temperatures from parafluvial treatment and control plots. The dashed line shows the diurnal range of soil temperatures for three months and

points show a seven-day moving average soil temperature diurnal range. Treatment data were unavailable during much of July 2019.

Anaerobic Soil Reduction

Field season average anaerobic soil reduction as measured by the full IRIS films (50 cm by 7.5 cm) were 45% for the T18 site and 8% for the T19 site; both treatment site percentages were greater than corresponding percentages at the C18/19 (3% and 4%; Figure 13). Full film anaerobic reduction percentage from both 2018 and 2019 (N= 165 treatment, N= 165 control) is significantly greater in the BMS treatment plots (Welch's Two Sample t-test p-value < 0.001, t = 10.348, df = 241.71, 95% CL= 21.37-31.43). Total field season full film anaerobic reduction percentage between T18 (N= 120) and C18 (N=120) plots was significantly different (p-value < 0.001 t = 11.908, df = 180.97, 95% CL=28.91-40.39), but not between T19 (N= 45) and C19 (N= 45) (p = 0.23, t = 1.2144, df = 79.658, 95% CL= -2.81-11.63).

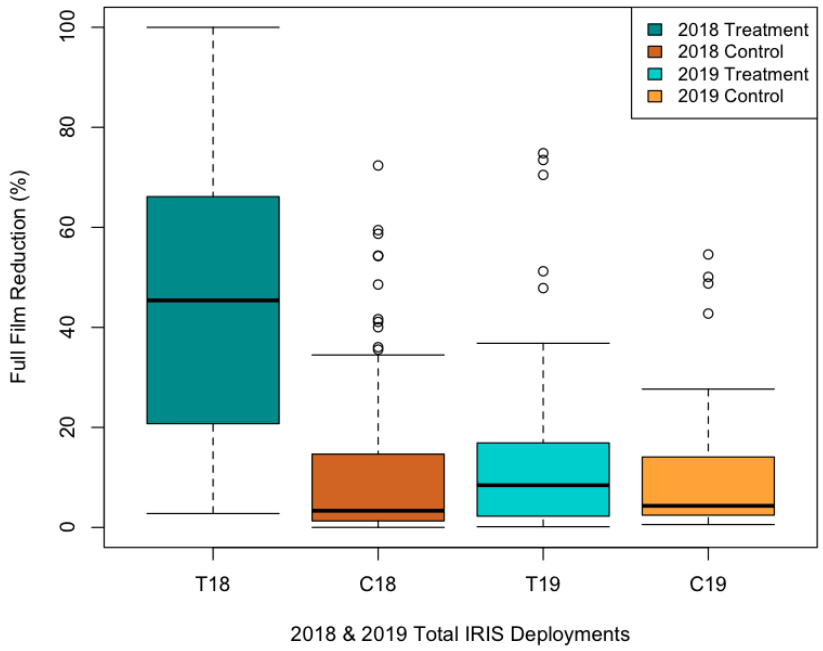


Figure 13: IRIS Full Film Fe/Mn oxide anaerobic reduction percentages for each field season year in treatment (T) and control (C) plots during 2018 (T18 N=120, C18 N=120) and 2019 (T19 N=45, C19 N=45).

Full film anaerobic reduction was greatest during June at T18 (61% and 82% Fe and Mn, respectively); June C18 full film average anaerobic reduction averaged 17% and 34% (Fe and Mn, respectively) (Figure 14, Appendix A: Table 1). T18 Fe and Mn anaerobic reduction percentages in June met the NRCS Technical Standard for Hydric Soils criteria (Figure 15, Appendix A: Figure 8), namely that a 30% anaerobic reduction occurred above or at 15 cm below the soil surface and continued deeper for 15 cm, creating a continuous zone of film anaerobic reduction (see Figure 10). Average July film anaerobic reduction was 27% Fe and 44% Mn at T18, while average film anaerobic reduction percentages at site C18 were much lower at 3% Fe and 5% Mn (Figure 15, Appendix A: Table 1). The average film anaerobic reduction at T18 in August was 15% Fe and 41% Mn (which met NRCS Technical Standard for Hydric Soils), while average film anaerobic reduction at C18 at the same time was 1% Fe and 1% Mn (Figure 15, Appendix A: Table 1).

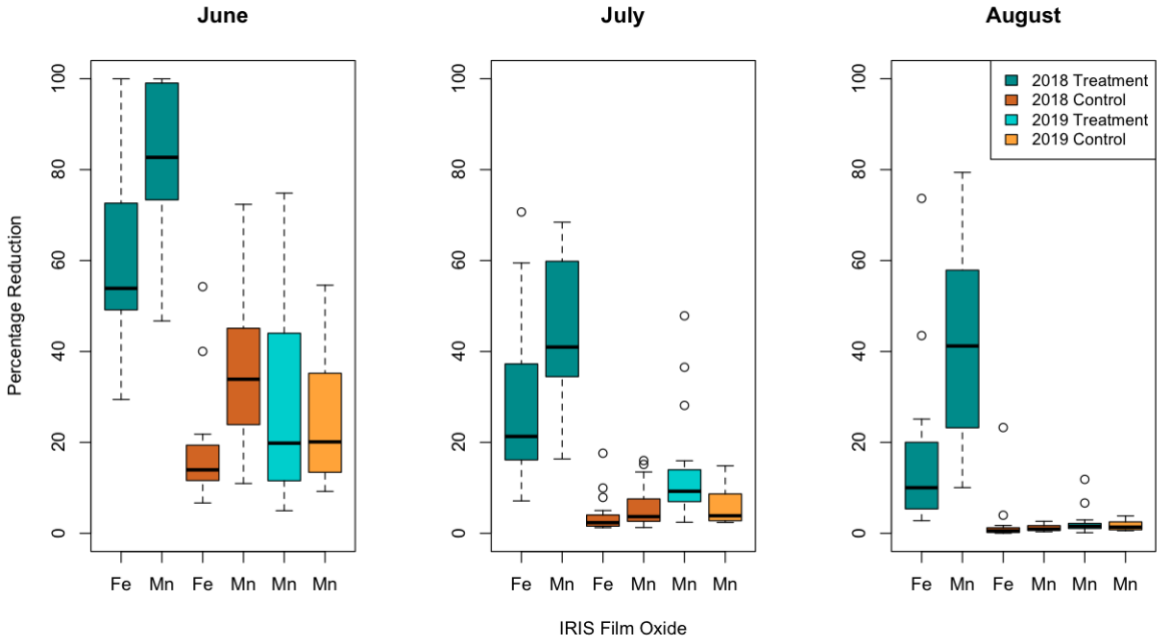


Figure 14: Monthly boxplots of 2018 and 2019 full film (7.5 cm width by 50 cm height) anaerobic soil reduction of IRIS Mn and Fe.

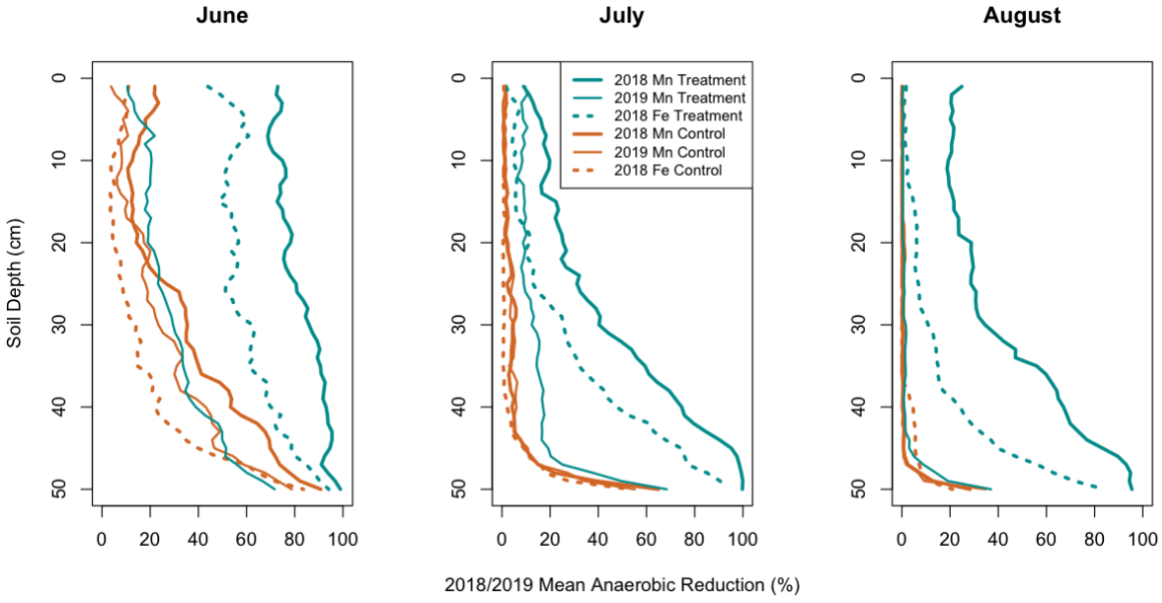


Figure 15: Average of 2018/2019 field season IRIS reduction results by depth. Each line is the depth-specific average of the entire month's deployment of IRIS films per treatment and control (2018/month N= 80, 2019/month N=30). Conventionally, the

median of each five-film plot is used to assess the NTCHS Hydric Standard, but for simplicity the mean of all plots is shown here.

Within T18 and C18, the highest percentages of Fe and Mn soil anaerobic reduction occurred within T18 plot #1 which has the closest proximity to the stream channel influenced by BMS. The other T18 plots #2, #3, and #4 showed decreasing reduction levels by distance from the stream (Appendix A: Figure A9).

Site T19 showed less anaerobic reduction compared to T18; however, the anaerobic soil reduction percentages were still greater than both C18 and C19 (see Figure 13, F-value = 106.7, $P \leq .001$). During the June T19 sampling period, the average full Mn film reduction was 29% in the treatment and 25% in the control (see Figure 14, Appendix A: Table A2). T19 July soils showed an average full film anaerobic reduction of 14% Mn, while C19 showed an average 6% Mn film anaerobic reduction. Anaerobic soil reduction as measured by IRIS was minimal at both treatment and control during August sampling with full film reduction at T19 averaging 2% Mn and control anaerobic reduction resulting in an average 1% Mn (see Figure 14: Appendix A: Table A2).

The plot nearest the stream at the T19 site met the NRCS Hydric Soil criteria during the June sampling period (see Figure 15, Appendix A: Figure A9). No other IRIS plots within T19 or C19 sites met the criteria for Hydric Soils.

Floodplain Vegetation Composition

The U.S. Fish and Wildlife Service assigned wetland indictors status for most species found in the U.S. spanning across upland (UPL), Facultative Upland (FACU), Facultative (FAC), Facultative Wetland (FACW) and Obligate Wetland (OBL) (Lichvar

et al., 2016). Across the eleven plots, no UPL or FACU species were found. The majority of plant species were FAC and/or FACW and a few dominant OBL species (Supplemental Tables 1, 2, & 3). The C18/19 was dominated by the rush *Juncus balticus* (37% cover); T18 was dominated by the sedge *Carex nebrascensis* (60% cover); the T19 was dominated by *Juncus balticus* (91% cover). All eleven vegetation plots at T18, T19, and C18/19 passed the Hydrophytic Vegetation Prevalence Index (United States Army Corps of Engineers, 2010) with scores indicating dominantly hydrophytic plant communities adapted for hydric soil conditions.

Vegetation Greenness

Although camera data collection was lost during June 2018, the GCC characterizations show that T18 vegetation stayed above 0.5 for the field season, while C18 hovered around 0.5 before dropping to 0.4 at the end of the field season. Similar GCC values resulted from the T19 and C19 vegetation where T19 stayed above 0.5 and C19 dropped below 0.5 after mid-June and continued a downward trend for the remainder of the field season (Figure 18). The missing GCC values from 2018 are likely impacting the Phenopix statistical models used to explain the green-up, maturity, senescence, and dormancy (Figure 19). However, the complete dataset from 2019 shows that the plant community at the treatment mature sooner and senesce an additional 20 days later than control vegetation. Plant dormancy differs between T19 and C19 as well, where vegetation at the treatment site experiences dormancy approximately 40 days after senescence and control vegetation experiences dormancy approximately 10 days after senescence.

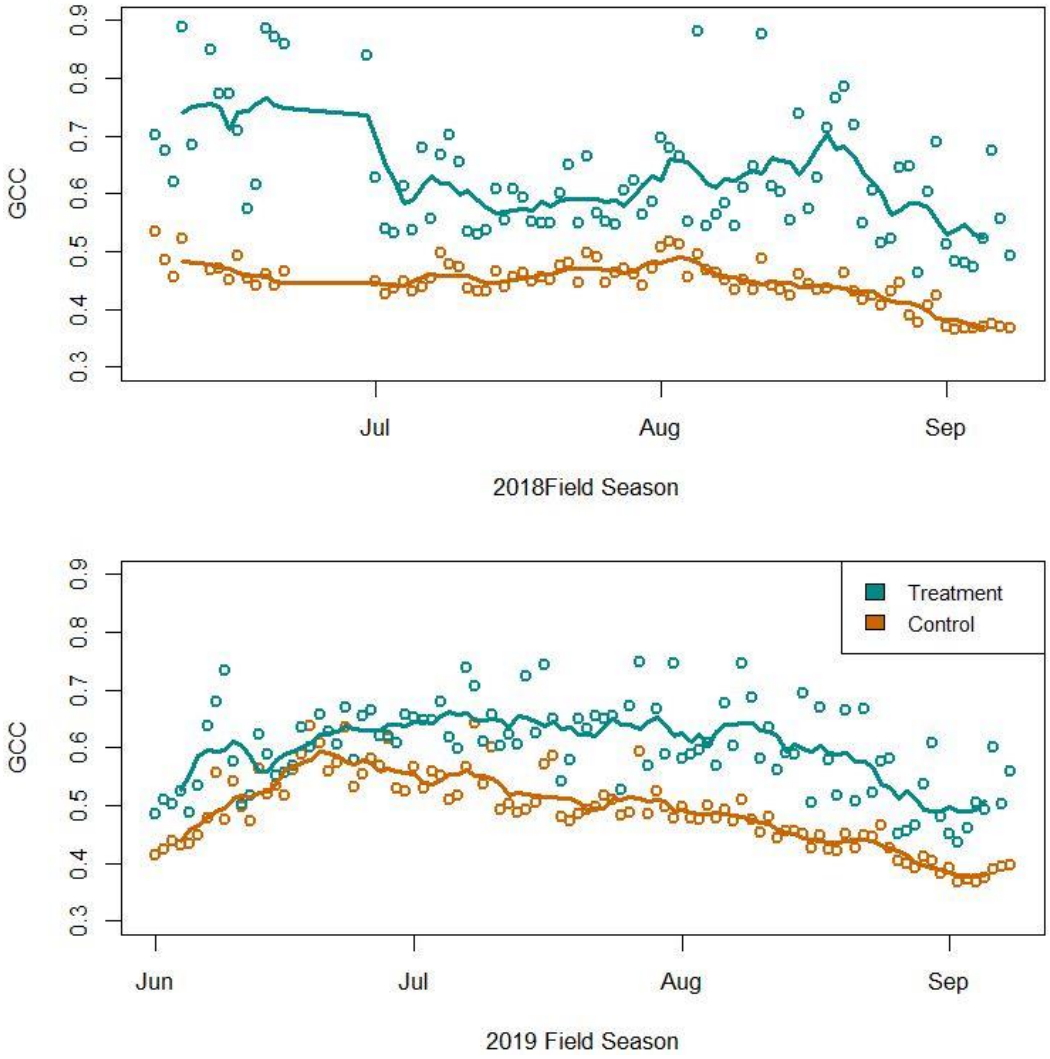


Figure 16: Green Chromatic Coordinate (GCC) values from the 2018 (N = 170) and 2019 (N = 200) field seasons. The trend lines placed through the data are a simple moving average.

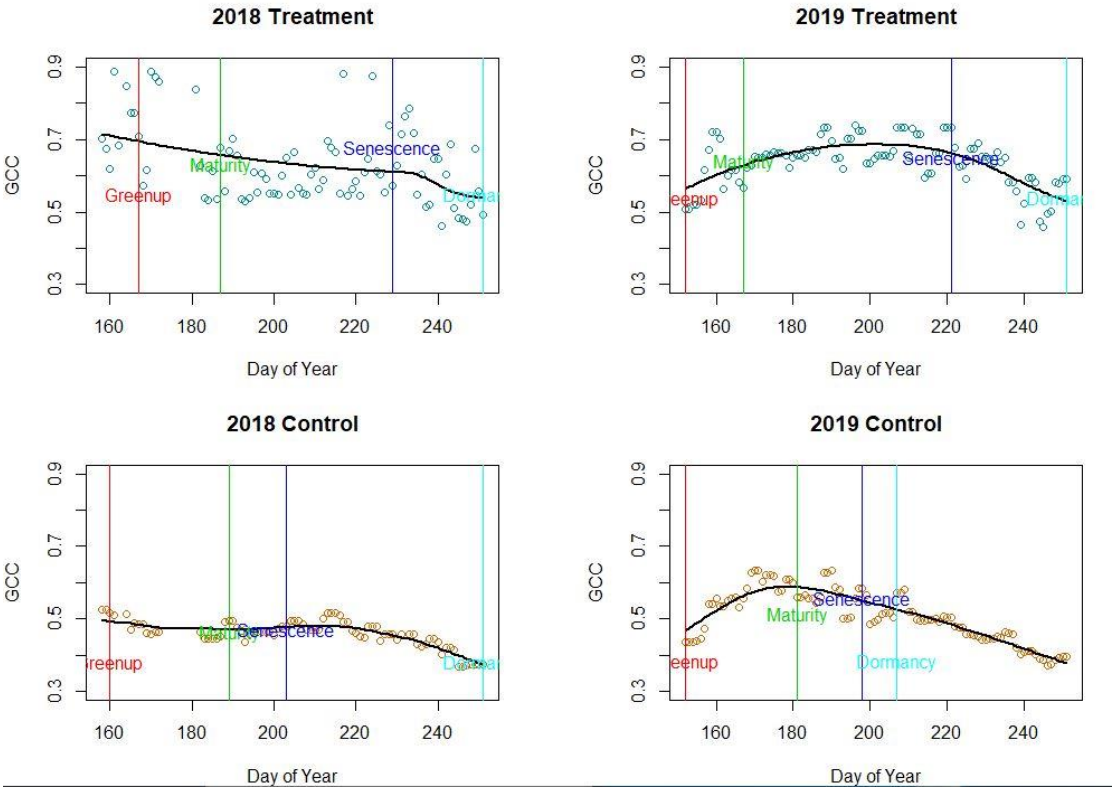


Figure 17: Statistical models of plant phenology using the green chromatic coordinate from the Phenopix package. T18 (N = 85) had the best fit (RMSE = 0.04) with a Klosterman curve and Klosterman Phenophase. T19 (RMSE = 0.03) (N = 100), C18 (N = 85) (RMSE = 0.01), C19 (N = 100) (RMSE = 0.02) all had the best fits with a Spline curve and Klosterman Phenophase.

Discussion

BMS installations at Long Creek, Montana influence hydrologic and biogeochemical ecosystem functions within parafluvial soils. Compared to the control location, the parafluvial zone near BMS increased soil water content, increased anaerobic reduction capacity, lowered soil temperatures, and decreased daily range in soil temperature, especially at the sites analyzed in 2018. These results agree with the notion that BMR can increase floodplain connection, subsequent soil-water content, and restore deteriorated parafluvial ecosystem functions like hydrophytic vegetation.

A common goal of beaver mimicry restoration is to facilitate beaver translocation (Pilliod et al., 2018) and the success of these translocation is attributed to proximity to palatable woody species, a moderate stream gradient and valley form (Allen et al., 2000). Beaver are located upstream from the TNC Sandhills Preserve and the restoration assumed that the reestablishment of woody hydrophytic vegetation communities adjacent to the BMS would facilitate their translocation. Hydrophytic plants, such a as *Salix spp.*, are essential components of beaver lifecycles, their reestablishment, along with the physical connection to the floodplain, lay the groundwork for natural beaver recolonization (Allen et al., 2000). These hydrophytic woody species are tolerant of hydric soil conditions and can easily out-compete less palatable, upland species if hydric soil is present. We see here that the BMS facilitated the creation of such conditions within the parafluvial after BMR through greater floodplain connectivity and raised groundwater leading to increased soil-water content that drives the development and maintenance of hydric soils.

The extent to which these hydric soil conditions are being restored is likely limited to the areas directly influenced by BMS installations. The resulting mosaic of soil ecosystem function occurs across the riparian zone, where preferential flow paths are being reestablished and/or newly created. Increased SWC within the riparian mosaic recreates the conditions ultimately necessary for hydric soil functions, such as denitrification, carbon storage, and maintain conditions suitable for hydrophytic plant communities.

Floodplain topography and the proximity to BMS coalesce to elevated SWC conditions necessary to develop or restore historic hydric soils. For example, physical description of field indicators indicate that the soils in the 18/19 control site are hydric according to the NRCS Field Indicators of Hydric Soils (Vasilas et al., 2018). However, Indicators of Reduction in Soils (IRIS) results suggest that soils at control sites remain hydrologically disconnected and oxygenated near the surface for the majority of the growing season. A lack of recent anaerobic reducing conditions within C18/19 suggests that field indicators derived from soil pit characterizations may be keying in on relicts from historic hydrologic connections and should not be considered representative of current conditions. The 2018 treatment soil also met the NRCS hydric field indicators and their physical descriptions were very similar soils to C18/19. However, site T18 showed IRIS anaerobic reducing conditions near soil surface throughout the field season. Although T18 and C18/19 are relatively close together, it seems that BMS influences on anaerobic conditions are constrained by proximity to the areas where BMS substantially increases groundwater levels.

Anaerobic conditions in the upper part (top 15 cm) of the soil profile is a fundamental aspect of delineating and classifying hydric soils. This standard is designed to highlight hydrologic conditions that directly impact the plant rooting zone and ultimately dictate plant community composition. Although this standard is essential for wetland identification to meet jurisdictional requirements, it may be less important when attempting to understand the impacts of ecological restoration. For example, IRIS results suggest a higher probability of anaerobic soil conditions occur in deeper soils (closer to the water table elevated by the BMS). These deeper soils are performing desirable ecosystem functions and subsequent services (such as denitrification and carbon storage), even though they are not within 15 cm of the surface.

BMS create pools upstream from the structure facilitating connectivity of the water table with parafluvial soils which, in turn, increase the probability of anoxic conditions. The eventual outcome of BMR may lead to higher rates of SOM storage, which allows for improved soil structure and water holding capacity. These changes in soil may lead to higher rates of surface water infiltration, long term pedogenic hydric development, and subsequent downstream ecosystem services such as improved water quality.

Decreased parafluvial soil temperature in an additional benefit of floodplain connectivity achieved by BMS. Parafluvial soil temperature plays a large role in biogeochemical processes that occur in hydric soils; for example, lower soil temperatures reduce the biological activity of soil microbes and therefore have the potential to increase SOM storage within riparian soils (Papenfuss et al., 2018). Wetter soil surface conditions

may buffer heat transfer from the atmosphere and sun to the underlying water table, allowing for more cold water refugia where groundwater reemerges to the stream channel. Thermal refugia like this is another common BMR goal; however, it is generally associated with hydrophytic vegetation reestablishment.

Evidence showing that parafluvial soils temperatures directly influence hyporheic and instream water temperatures is less common. BMR research has shown that instream water temperature moderation is increased post-BMR, but any connection of this effect to the influence of parafluvial soils is not clear (Maia et al., 2018; Weber et al., 2017). Studies have shown that plant shade directly impacts water temperatures (Berman et al., 2001; Hamilton et al., 2020), therefore BMS influences on soil temperature and soil moisture may heighten the potential of hydrophytic vegetation communities ability to directly shade streambanks and surface water over the growing season.

BMR attempts to initiate the hydrophytic plant development that beavers accomplish by building dams through the elevation of groundwater and SWC within the parafluvial zone. The mechanisms for BMR induced changes in vegetation are less clear. Research analyzing groundwater levels has documented a clear influence of BMR on the water table (Bobst, 2019), but soil moisture changes within the rooting zone following BMR has been generally assumed rather than evidenced (Fairfax et al., 2018; Melanie et al., 2018). This work provides foundational datasets (Whitehead, B. K. (2020). The Response of Parafluvial Soils to Beaver Mimicry Restoration, HydroShare, http://www.hydroshare.org/resource/03cc4add78b245bcbbfe900ac52a2812) that directly illustrate mechanisms by which soil SWC maybe restoring and/or advancing soil

conditions that sustain hydric conditions and hydrophytic plant communities. Vegetation communities at T18, T19, and C18/19 met the criteria for hydrophytic vegetation, indicating that these riparian areas may be adequately supporting herbaceous wetland plant communities. However, without the necessary floodplain connection created by the BMS at T18 and T19, it is evident that C18/19 lacks SWC (especially during July and August) which may limit the establishment of woody riparian species that beaver depend on. Woody riparian vegetation establishment is slower than the immediate BMS induced hydric soil conditions and establishment of subsequent herbaceous vegetation communities. Overtime, elevated SWC and hydric soil development will likely recruit and support the development of woody hydrophytic vegetation.

This study has shown, at least for Long Creek, that soils adjacent to BMS have the capacity to support the establishment of woody species that beavers use as forage, building materials, and habitat. Once the abundance of woody species within the BMR locations are capable of supporting beaver, it is expected that beaver successfully living upstream will migrate into these ecologically restored systems. Beaver recolonization and subsequent dam building/maintenance will continue the ecological restoration of the parafluvial by creating an expanding mosaic of ecosystem functions necessary for their populations to survive.

Conclusion

My research shows that BMS elevated water tables provide wetter soil conditions in the first 30 cm of soil from soil surface, leading to conducive conditions for the development of and/or reestablishment of hydric soils. These conditions also influenced hydrophytic vegetation on the soil surface by extending the period of photosynthetic growth. Soils adjacent to BMS compared to soils not influenced by BMS experienced an extended (~46 days) and wetter moisture regime (0.23 m^3/m^3 more), increased anoxic conditions (27% more), a dampened soil temperature range (2.5 °C less) and cooler soil temperatures (5 °C less), and supported vegetation greenness (~20 additional days) during the driest months of the field season. The highest degree of these influences of BMS on soils occurred in plots closest to BMS structures, indicating that further research should explore exactly where within floodplain soils response to elevated water tables. This first step towards understanding BMR influences on parafluvial soils furthers our understanding of how soils response to instream restoration and may lead to more robust soil research post-BMR.

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Supplemental Material

Table S1: Treatment vegetation composition and wetland status.

Treatment 2018 Vegetation				
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot #1	*Carex nebrascensis*	OBL	97	1.02
	Argentina anserina	OBL	1	
	Deschampsia cespitosa	FACW	1	
	Carex praticola	FACW	1	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot #2	*Carex nebrascensis*	OBL	97	1.03
	Almutaster pauciflorus	FACW	3	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot #3	*Juncus balticus*	FACW	90	1.96
	Carex nebrascensis	OBL	4	
	Almutaster pauciflorus	FACW	4	
	Argentina anserina	OBL	1	
	Poa pratensis	FAC	1	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot 4#	*Juncus balticus*	FACW	55	1.58
	Carex nebrascensis	OBL	42	
	Almutaster pauciflorus	FACW	1	
	Argentina anserina	OBL	1	
	Poa pratensis	FAC	1	
			100	

Table S2: 2018 Control vegetation composition and wetland status.

Control 2018/2019 Vegetation				
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot #1	*Argentina anserina*	OBL	54	1.57
	Juncus balticus	FACW	31	
	Poa pratensis	FAC	10	
	Almutaster pauciflorus	FACW	2	
	Hordeum brachyantherum	FACW	1	
	Trifolium eriocephalum	FAC	1	
	Carex praticola	FACW	1	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot #2	*Juncus balticus*	FACW	50	2.38
2019 Plot #3	*Poa pratensis*	FAC	31	
	Almutaster pauciflorus	FACW	12	
	Trifolium eriocephalum	FAC	7	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot #3	*Juncus balticus*	FACW	36	2.41
2019 Plot #2	*Poa pratensis*	FAC	33	
	Almutaster pauciflorus	FACW	15	
	Trifolium eriocephalum	FAC	12	
	Argentina anserina	OBL	4	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
2018 Plot #4	*Juncus balticus*	FACW	33	2.22
2019 Plot #1	*Iris missouriensis*	FACW	28	
	Poa pratensis	FAC	20	
	Argentina anserina	OBL	8	
	Trifolium eriocephalum	FAC	5	
	Phleum pratense	FAC	5	
	Hordeum brachyantherum	FACW	1	
			100	

Table S3: 2019 Treatment vegetation composition and wetland status.

Treatment 2019 Vegetation				
Year / Plot #	Species	Status	% Cover	Prevalence
Plot #1	*Juncus balticus*	FACW	84	1.99
	Deschampsia cespitosa	FACW	10	
	Carex nebrascensis	OBL	3	
	Potentilla fruticosa	FAC	2	
	Muhlenbergia filiformis	FACW	1	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
Plot #2	*Juncus balticus*	FACW	95	2
	Argentina anserina	OBL	2	
	Poa pratensis	FAC	1	
	Potentilla fruticosa	FAC	1	
	Almutaster pauciflorus	FACW	1	
			100	
Year / Plot #	Species	Status	% Cover	Prevalence
Plot #3	*Juncus balticus*	FACW	94	2.02
	Poa pratensis	FAC	2	
	Phleum pratense	FAC	1	
	Carex nebrascensis	OBL	1	
	Argentina anserina	OBL	1	
	Potentilla gracilis	FAC	1	
			100	



Figure S1: Image used in the R package Phenopix to select the ROI for pixel analysis.

CHAPTER THREE

SUMMARY OF FINDINGS AND FUTURE RESEARCH

Land Classification Implications

My research has shown that BMR has the capacity to influence parafluvial soil moisture, which leads to higher anaerobic soil reduction, cooler soil temperatures, and the sustainment of hydrophytic vegetation growth. These results have several considerable implications for landowners and restoration practitioners. For example, the development of hydric soil post-BMR may change land classifications from unregulated to regulatory wetlands. Understanding potential land classification and regulation changes may inform future restoration design and goals, and future landowner participation.

Currently, two major United States wetland definitions are in use (Table 3), (1) The Clean Water Act's (CWA) and (2) the Food Security Act's (FSA). The Army Corps of Engineers (ACOE) uses the CWA definition (in conjunction with three criteria: plants, soils, and hydrology) and is the main regulatory agency with jurisdictional Waters of the U.S. (WOTUS) including wetlands (United States Army Corps of Engineers, 2010). Riparian areas bordering streams that are being considered for restoration may already contain hydrophytic vegetation but lack the hydrologic connections and soil development requirements to meet the criteria for a jurisdictional wetland classification and its subsequent protection. A shift in floodplain connection following BMR may produce the

conditions necessary for wetland hydrology and the eventual development (or redevelopment) of hydric soils.

Table 3: US federal definitions used in official wetland and hydric soil delineation.

Source	Definition
Clean Water Act Wetland Definition (United States Environmental Protection Agency, 1948)	"Wetlands are areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas."
Food Security Act Wetland Definition	"Land that has a predominance of hydric soils. Is inundated or saturated by surface or groundwater at a frequency and duration sufficient to support a prevalence of hydrophytic vegetation typically adapted for life in saturated soil conditions. Under normal circumstances supports a prevalence of such vegetation. For the purposes of FSA [Food Security Act] and any other Act, this term does not include lands in Alaska as identified as having high potential for agricultural development that have a predominance of permafrost soils."
The National Technical Committee for Hydric Soils (NTCHS)	A soil that formed under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper part.

Depending on the restoration and landowner goals, the potential development of a wetland-mosaic landscape may be desired or undesired. The 2018 IRIS results show that BMS treatment soils met the Hydric Soil Standard during June and August and that control soils never met the standard for the entire field season (Figure A8). What this may indicate is that before BMS installation, the treatment location hydrology and soils may have not met the ACOE criteria for wetland classification, however; post-BMS the

hydrology and soils may shift and eventually change the regulatory nature of the landscape.

Currently, soil changes seem to be contained within close proximity to the BMS; however, long-term impacts at the landscape scale may be more diverse as the channel aggrades. Due to soils' inherently slow development potential increases in SOM, flow paths, redoximorphic features, changes in chemical composition, and structure will likely go unnoticed post-BMR. It would not be unreasonable to forecast far into the future of these soils and observe greater hydric soil properties due to BMS reconnected floodplain, willow requirement, and eventual beaver recolonization.

Additional IRIS Film Analysis & Field Installation Considerations

Field indicators of hydric soils were documented at the control location; however, based on the IRIS results from the C19/19, these field indicators are likely relict features persisting within the soil from history when greater floodplain connection existed. At the time of sampling, I was unaware of the field technique that can help to distinguish between relict and recent redoximorphic features. This technique consists of breaking or cutting open redox mottles and nodules to assess whether the boundary separating the reduced from the oxidized iron is sharp or defused (Vepraskas, 1992). Employing the technique may have influenced the hydric soil indicator results from treatment and control soils.

Another consideration regarding soil reduction results is the impact of highly oxygenated rainwater that may provide enough oxygen to delay or subdue soil anaerobic

respiration in the upperpart (Richardson et al., 2001). The field season of 2019 experienced more frequent rain events compared to the 2018 field season; however, the soil moisture at C18/19 was very similar during both field seasons. The frequency of rain events may have increased SWC at T19 and may also explain the lack of soil reduction in the upper portion of T19 IRIS films.

Soil reduction in shallow horizons is a foundational criterion for classifying hydric soils, which broadly indicates the presence of saturating conditions close to the soil surface. Field indicators and IRIS standards both set thresholds of physical or chemical criteria that must be met within the first 15 cm of soil surface. Although this depth may seem arbitrary, it sums up the robust field experience of the NRCS soil survey scientists and sets a reasonable standard that should be met if sufficient hydrology is present within the rooting zone of hydrophytic plants.

I made several interesting field observations that may aid future discussion of field indicators of hydric soil and IRIS film standards. The first observation concerned the placement of the T19 IRIS array right above the BMS, this was where the most floodplain connection seemed to be occurring in May 2019. The first plot (#1) was right next to the wetted edge and the last plot (#3) was the furthest inland, away from the stream. I thought this placement was reasonable since trying to estimate where flow paths and hyporheic exchange was occurring worked well in 2018, it would work well again while selecting the installation for 2019. However, the estimation I made may have missed belowground preferential flow paths.

These concerns may have been justified by the lack of T19 IRIS reduction. The second observation concerned the smell of hydrogen sulfide gas throughout the field season at T19 during the removal and installation of IRIS films. Anaerobic soil respiration only utilizes SO_4^{2-} when oxygen, nitrogen, manganese, and iron are unavailable. Thus, pulling a completely unreduced manganese oxide film from the soil profile while simultaneously smelling the distinctive odor of reducing soil conditions was perplexing. One explanation may be the high pH of the soils (see Chapter Two Table 2), which minimizes the reduction of iron and manganese. The more reasonable explanation is that my selection for IRIS installation was poor and that a better estimation of where floodplain connection would occur may have been able to show anaerobic soil conditions right next to or in close proximity to T19.

The installation of IRIS plots should be informed by vegetation, soils and hydrology in any attempt to capture soil reducing conditions. The gradient of proximity to the BMS was insightful but may be more useful with slight field design improvements. For example, in 2018 I installed a total of eight plots (20 Mn and 20 Fe films) per month at the treatment location and again at the control location (80 films total per month). This plot arrangement led to my third observation, which was that IRIS films may have been better utilized by deploying two plot proximity transects to stream edge up and downstream of the BMS. This may have provided better insight into exactly where, at what depth, and to what degree hydric soil conditions occur post-BMR.

The IRIS film analyses I preformed mostly utilized the full film anaerobic reduction percentage and the mean of five film plot anaerobic reductions by depth.

Traditionally, the median from a five-film IRIS plot is used to assess whether the soils have met the IRIS hydric standard. In this case, it became evident that T18, T19, and C18/19 parafluvial soils only met or approached the hydric standards during June. Therefore, discussing the full film and mean by depth anaerobic reduction percentages established a more appropriate method for discussing general trends of anaerobic reduction between treatments and control. These IRIS statistics may be better suited for parafluvial soils, which undergo variable saturation.

Additional Soil Water Content & Temperature Analysis

To approach my data with an analysis that included higher sample sizes and a random selection, I designed a statistical approach with the guidance of several statisticians. We assumed that the localized precipitation would suffice as a randomized treatment of BMS and non-BMS influences on soils. The hypothesis was that BMS soil would be wetter and non-BMS soils would be drier, especially during the months of July and August, which would highlight a difference in how soil moisture responded to rain. Each rain event greater than or equal to 1 mm initiated the start of a sampling unit and each subsequent hour that rain occurred the associated soil moisture was cataloged. We assumed that the difference between pre-rain soil moistures and post-rain soil moistures would be greater in non-BMS soils due to their lack of floodplain connectivity. Although several of our sampling units supported this theory, the influences of upstream rain events elevating local groundwater and a lack of large rain events during the driest months limited the scope of this approach.

Building off of Dr. Anthony Hartshorn's idea, I analyzed average IRIS five-film-plot anaerobic soil reduction percentages from the top 30 cm of soil with respect to water content and temperature (Figure 20 and Figure 21). In some cases, these figures support well-documented anaerobic reduction processes, such as an increase in reduction with water content and temperature (Kirk, 2004; Papenfuss et al., 2018; Richardson et al., 2001). They also show the differences and similarities between treatment and control soils over time. Additionally, the data from 2018 highlight the sensitivity of Mn oxides over Fe oxides IRIS film (Figure 20).

We know that as C18/19 soils dried (decreasing the capacity for anaerobic respiration), their temperatures also increased. Interestingly, the soil temperatures in 2018 seemed to have peaked in July and then returned to cooler temperatures in August. Higher soil temperatures would be expected to increase microbial action, but in this case, less anaerobic respiration occurred due to the lack of water. Site T18 soils showed a similar temperature trend, but anaerobic reduction percentages increased with temperature because water contents also remained higher.

In 2019 the soils show a simplified and interesting difference between treatment and control. Soil moistures from June and July were very similar between T19 and C19; however, more anaerobic reduction occurred at T19. This relationship is mirrored in soil temperatures, where June and July T19 and C19 host similar soil temperatures, but more anaerobic reduction occurred at T19.

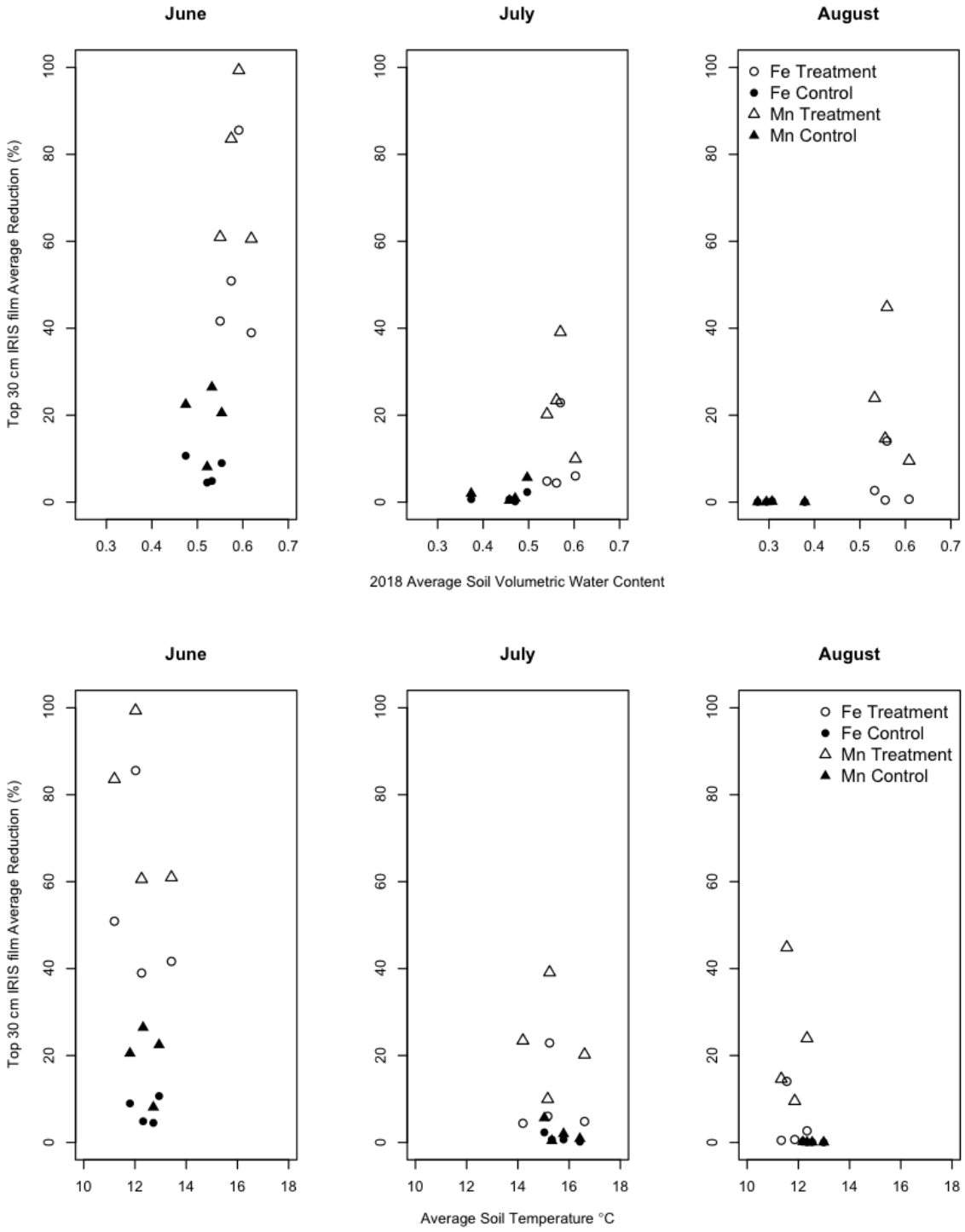


Figure 18: Average anaerobic soil reduction of the top 30 cm of IRIS film plots (N=5) with associated average monthly soil moisture and temperature from the 2018 field season.

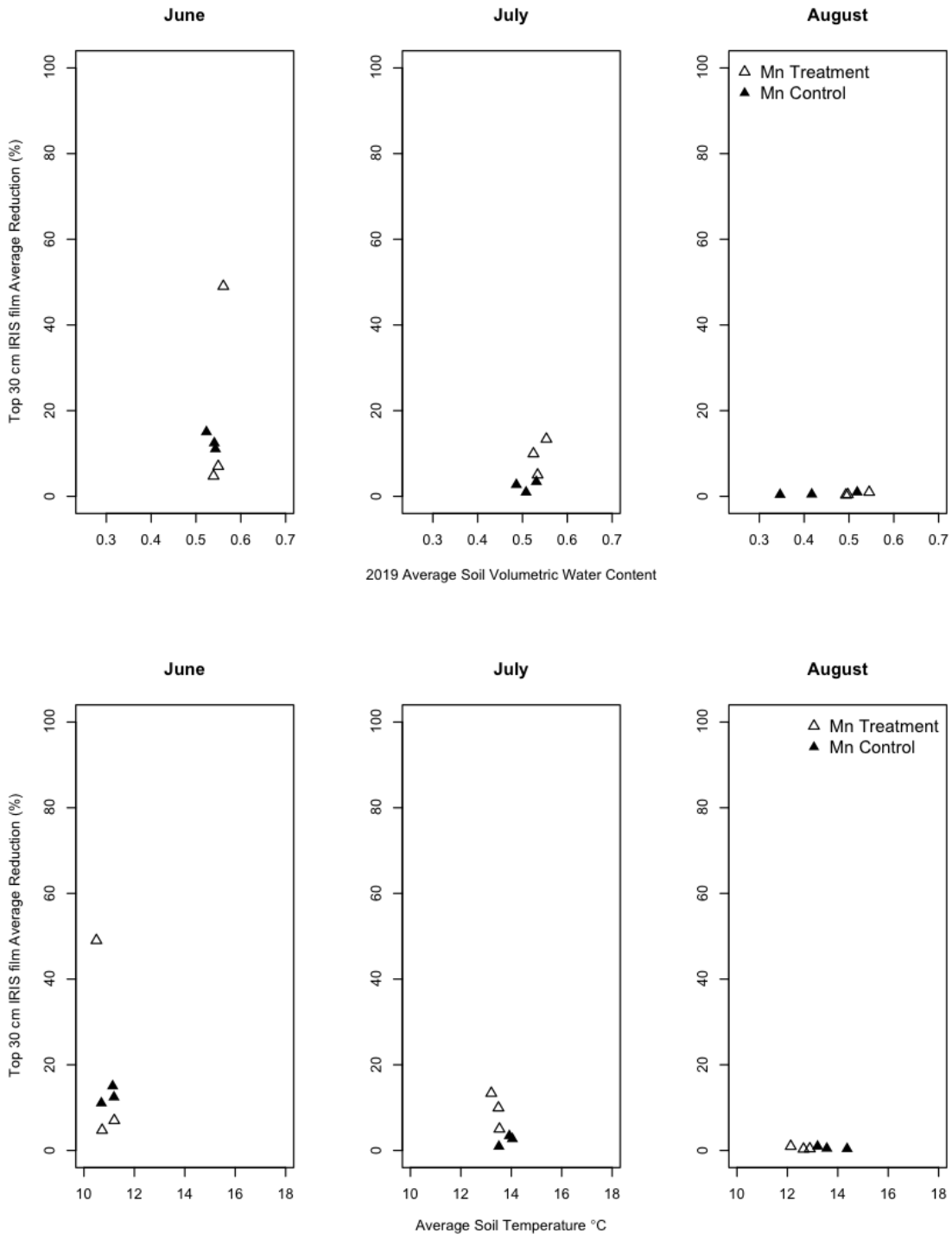


Figure 21: Average anaerobic soil reduction of the top 30 cm of IRIS film plots (N=5) with associated average monthly soil moisture and temperature from the 2019 field season.

Future Field Instrumentation Considerations

Several other field design improvements should be considered for future research. More transferrable conclusions would be gained by selecting BMS across a large range of stream orders and soil types at a sample size that would allow for a more robust and generalizable statistical analysis. Installation of monitoring equipment should be deployed at all BMS sites that are either complemented by paired controls or even more ideally, equipment should be deployed at the same site before BMS installation. The most reliable instruments I used were the soil sensors and IRIS, while the PlantCams provided adequate data for analysis.

PlantCams should be installed using the appropriately designed equipment to secure them in the field. They should also be protected from wildlife and livestock. The photos I collected in 2019 had much higher quality due to the improved orientation of the cameras. During the 2018 field season, I installed the cameras facing west to assess all eight IRIS/soil sensor plots. This resulted in poor lighting and afternoon/evening glare. In 2019 I installed the cameras to face north, which resulted in better lighting and greenness exposure of the cameras. When installing PlantCams to assess the differences between treatment and control locations, one should also take notes of the differences in vegetation. Species-specific drought resistance and general biology may be starkly different between herbaceous plants and limit the actual differences found within photo analysis.

The R package Phenopix used to analyze the PlantCam photos hosts a variety of options that may be of benefit for future work. When selecting the Region of Interest

(ROI) in the photo for greenness analysis within the Phenopix package, there is a new 2020 option to select multiple ROIs. This would have been an interesting option to plan for in 2018 and 2019 when installing cameras. A gradient of ROIs could have been established in proximity to the BMS and paired with soil sensors and IRIS anaerobic reduction results. The differences between plots may have been subdued, but on a larger scale multiple ROI's may have been able to isolate where and when plants no longer have sufficient water.

Future research would also benefit from utilizing the current investigations into the need to create new pilot holes for every IRIS film installation. Researchers in Laramie, WY are currently testing this idea in alpine wetlands by comparing the reduction percentages with the use of prior IRIS film pilot holes to the reduction percentages from newly created pilot holes. Not only will this research narrow the IRIS installation procedures, but it may shift the way IRIS is used in riparian areas. Deployment of IRIS technology in new pilot holes requires extensive physical labor in drier conditions, and in this study, installations became more difficult as the field season advanced and soils dried. I also became concerned that the additional pilot hoes in the same 1 m^2 plot may have created an unintended experimental effect on the oxygenation of the soil profile. Special considerations may need to be used when utilizing IRIS films in seasonally wet riverine wetlands and riparian wetlands.

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APPENDICES

APPENDIX A

SOIL REDUCTION

Table A1: 2018 IRIS Total Film Reduction summary statistics.

2018 Treatment Total Film Reduction 2018									2018 Control Total Film Reduction 2018							
	Fe				Mn				Fe				Mn			
	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
June	61%	18%	29%	99%	82%	16%	46%	99%	17%	11%	6%	54%	34%	17%	10%	72%
July	27%	17%	7%	70%	44%	16%	16%	68%	3%	3%	1%	17%	5%	4%	1%	16%
August	15%	16%	2%	73%	41%	21%	10%	79%	1%	5%	0%	23%	1%	0%	0%	2%

Table A2: 2019 IRIS Total Film Reduction summary statistics.

2019 Treatment Total Film Reduction 2019					2019 Control Total Film Reduction 2019			
	Mn				Mn			
	Mean	SD	Min	Max	Mean	SD	Min	Max
June	29%	25%	4%	74%	25%	15%	9%	54%
July	14%	12%	2%	47%	6%	4%	2%	14%
August	2%	2%	0%	11%	1%	1%	0%	3%

Table A3: 2018 IRIS film installation and total.

2018 Control Instrumentation/Month for Growing Season					
Instrument	Plot #1	Plot #2	Plot #3	Plot #4	Monthly Totals
Fe IRIS films	5	5	5	5	20
Mn IRIS films	5	5	5	5	20
Soil Sensors	1	1	1	1	4
Total Season IRIS Films Used:					120

2018 Treatment Instrumentation/Month for Growing Season					
Instrument	Plot #1	Plot #2	Plot #3	Plot #4	Monthly Totals
Fe IRIS films	5	5	5	5	20
Mn IRIS films	5	5	5	5	20
Soil Sensors	1	1	1	1	4
Total Season IRIS Films Used:					120

2018 Growing Season IRIS Films Used:	240

Table A4: 2019 IRIS film installation and total. Reduced to 3 plots and only Mn films to minimize labor and analysis.

2019 Control Instrumentation/Month for Growing Season				
Instrument	Plot #1	Plot #2	Plot #3	Monthly Totals
Mn IRIS films	5	5	5	15
Soil Sensors	1	1	1	3
Total Season IRIS Films Used:				45

2019 Treatment Instrumentation/Month for Growing Season				
Instrument	Plot #1	Plot #2	Plot #3	Monthly Totals
Mn IRIS films	5	5	5	15
Soil Sensors	1	1	1	3
Total Season IRIS Films Used:				45

2019 Growing Season IRIS Films Used:	90

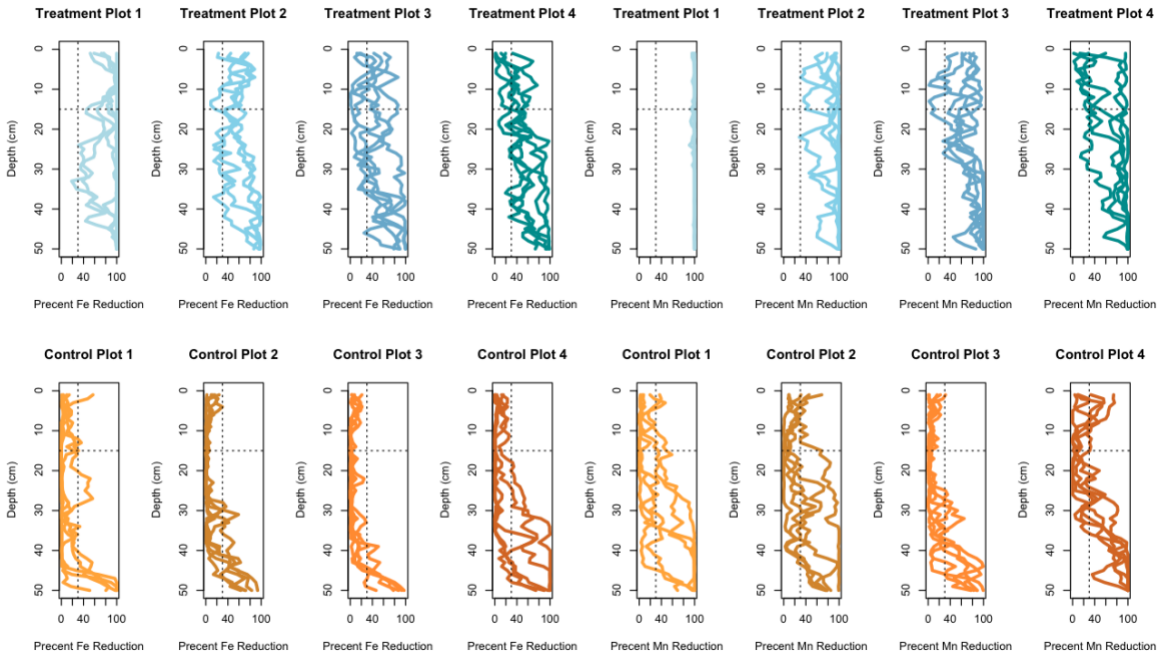


Figure A1: 2018 June plot #1-4 individual films (five per plot) reductions by depth.

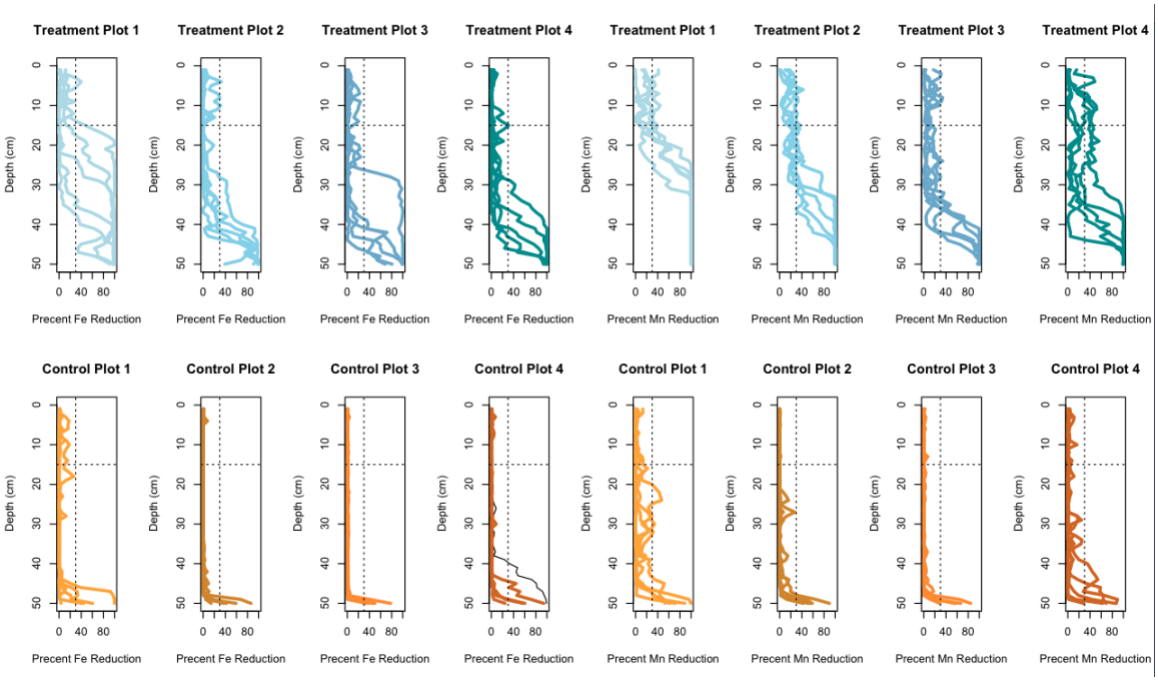


Figure A2: 2018 July plot # 1-4 individual film (five per plot) reductions by depth.

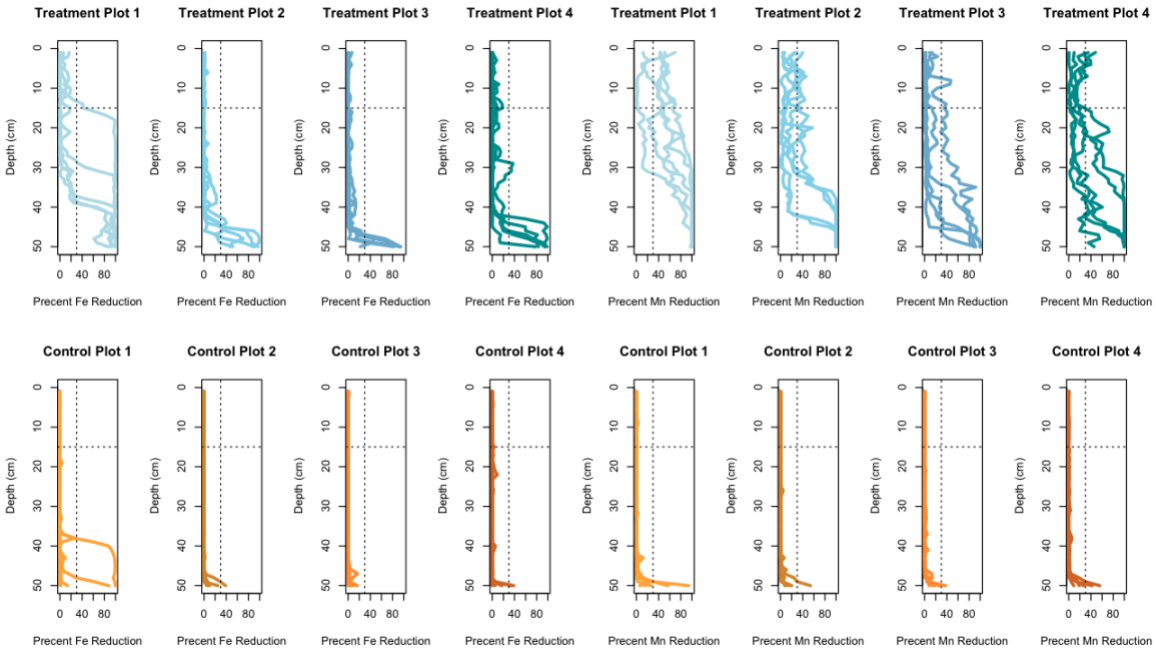


Figure A3: 2018 August plot #1-4 individual film (five per plot) reduction by depth.

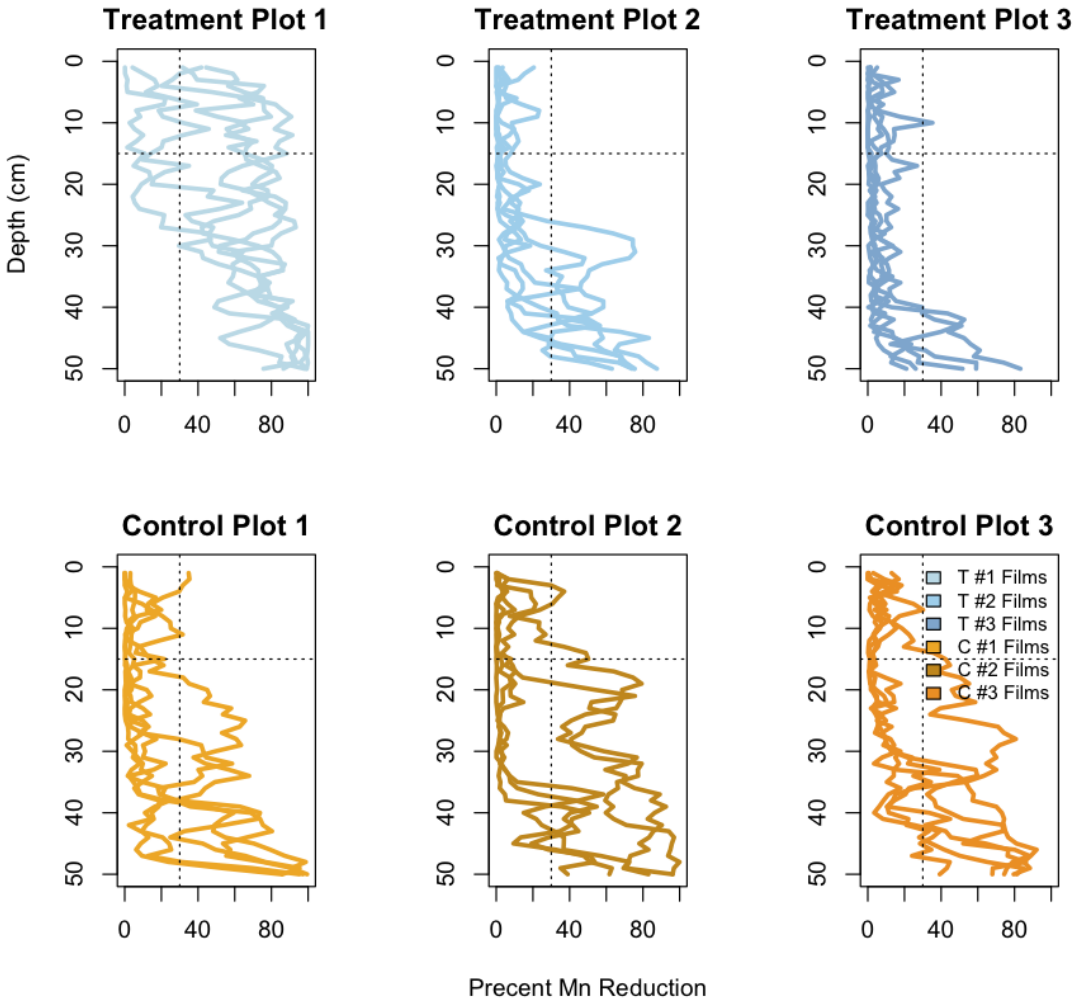


Figure A4: 2019 June plot# 1-3 individual film (five per plot) reductions by depth.

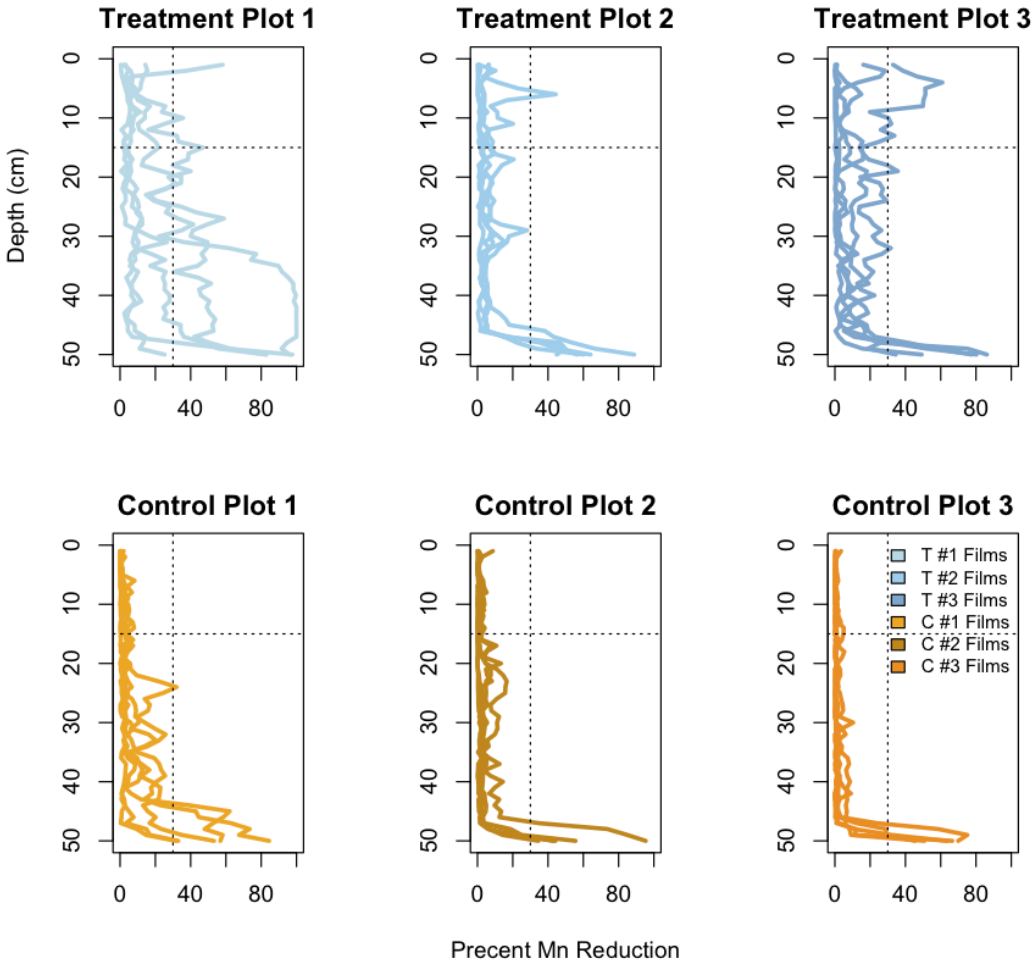


Figure A5: 2019 July plot# 1-3 individual film (five per plot) reductions by depth.

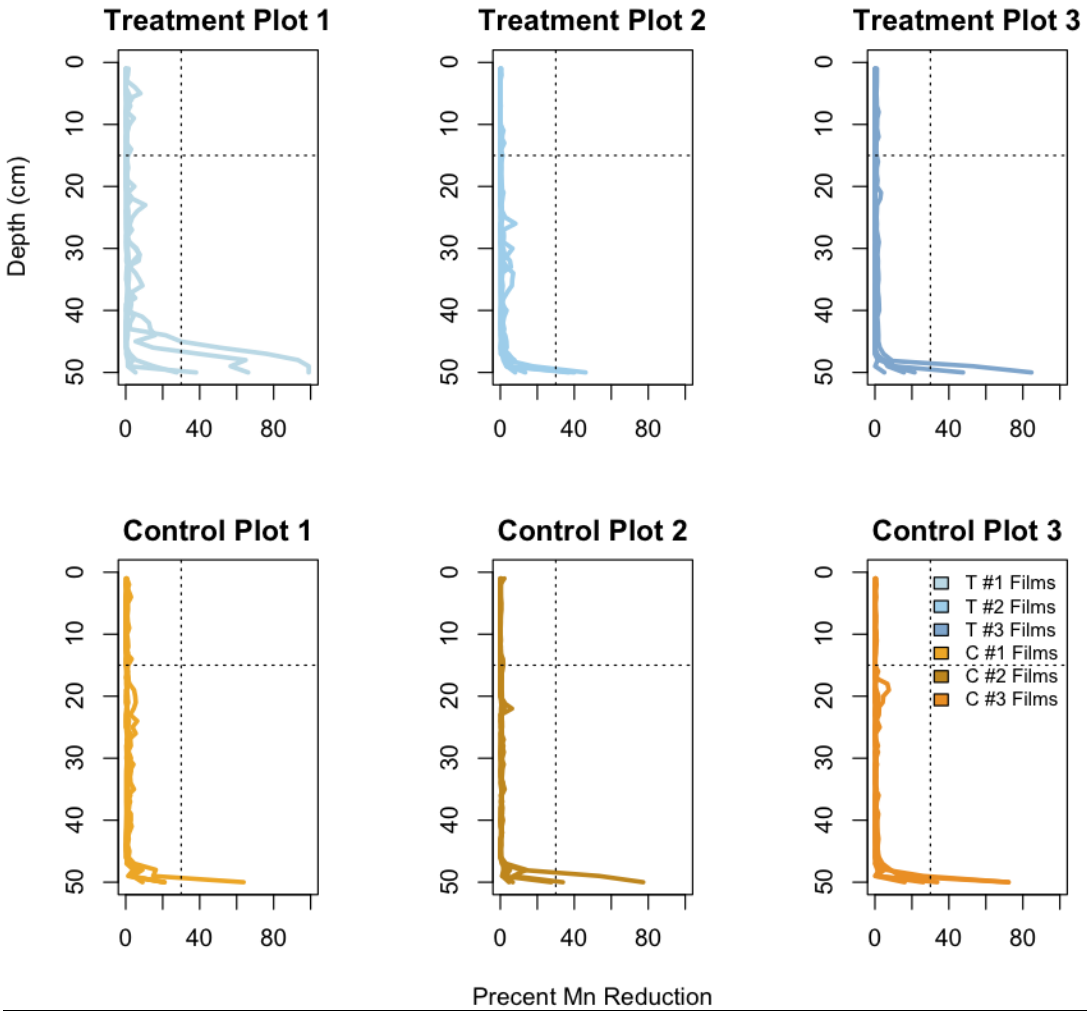


Figure A6: 2019 August plot # 1-3 individual (five per plot) reductions by depth.

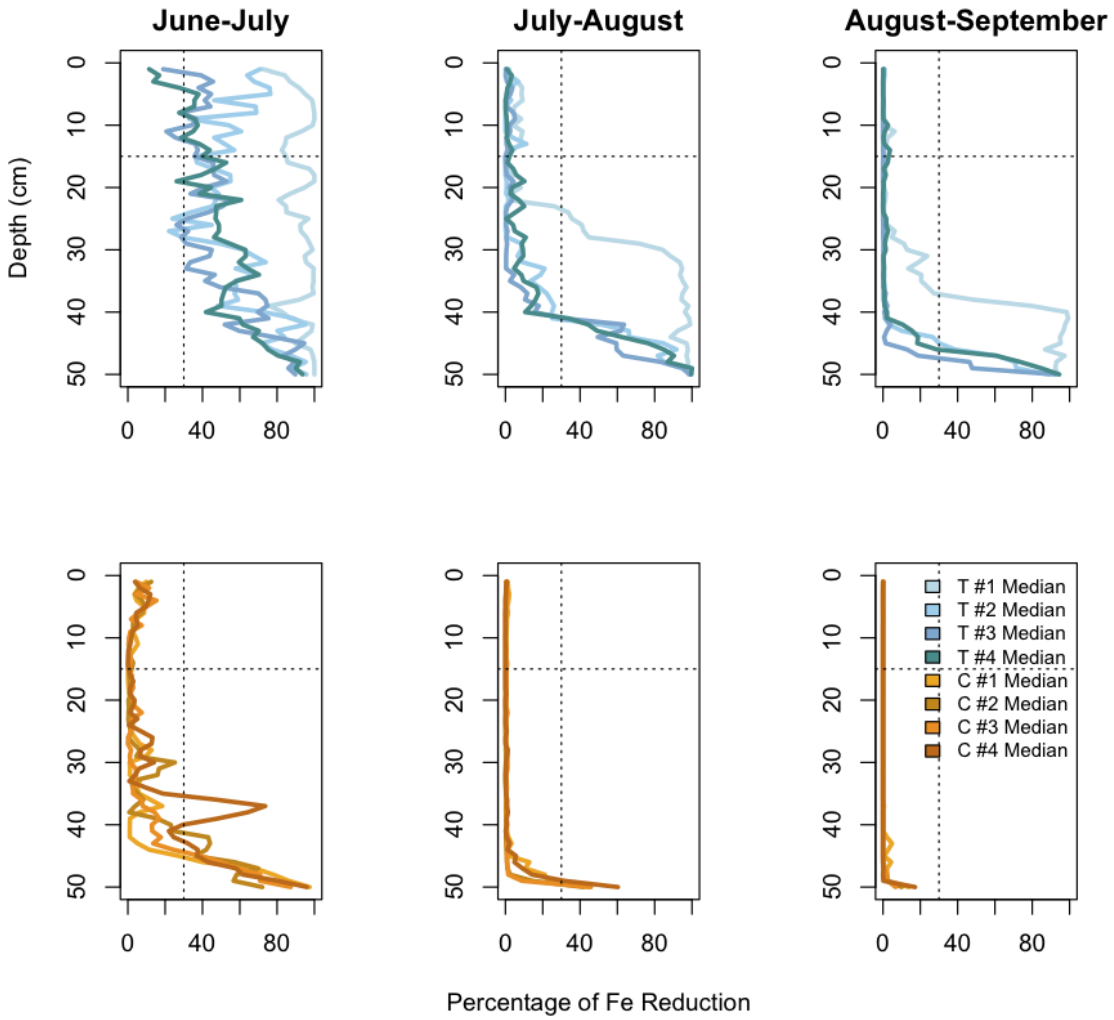


Figure A7: 2018 Median Fe IRIS film results used to assess Hydric Soil Standard. Treatment plot #1 during the June-July deployment met the standard, while no other treatment or control IRIS deployments met the standard during the entire field season.

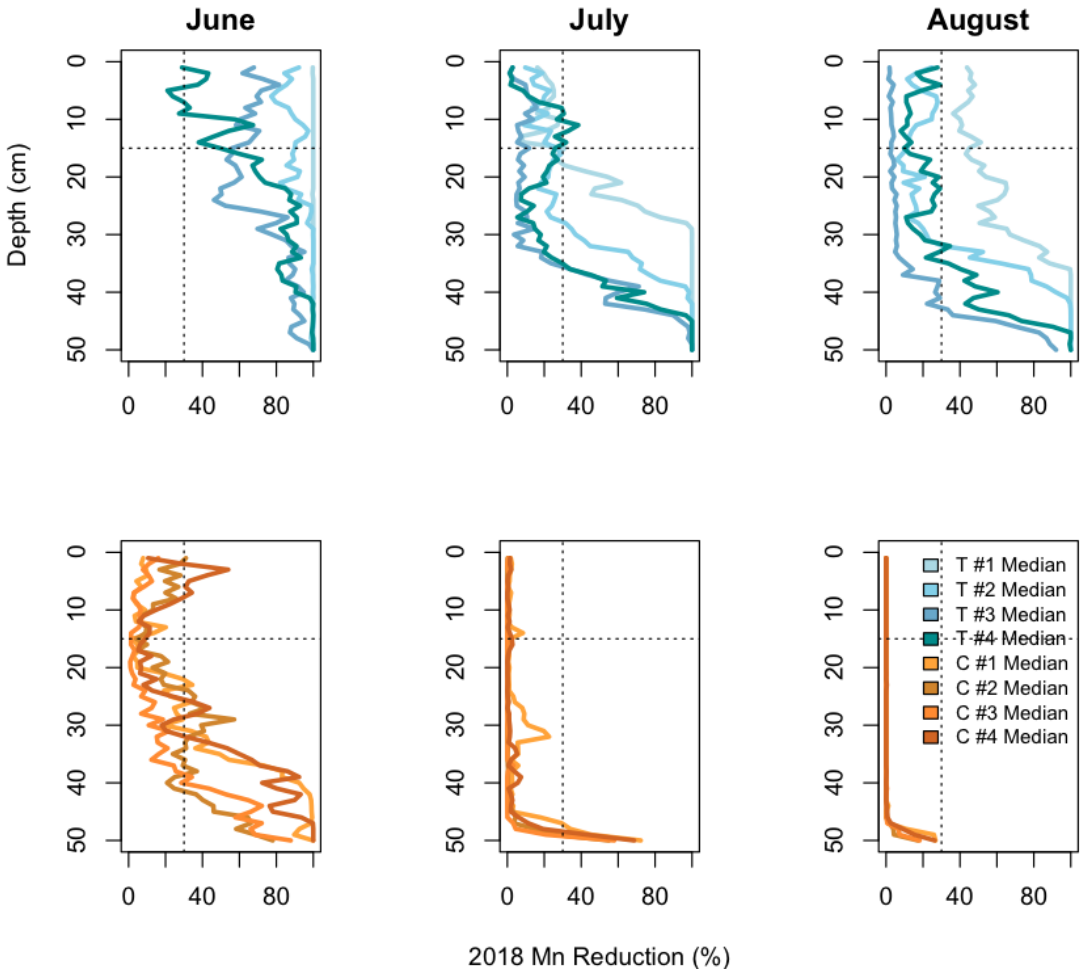


Figure A8: 2018 Median Mn IRIS film results used to assess Hydric Soil Standard. Treatment plots #1-4 all met the standard during the June-July deployment and plot #1 met the standard again during the August-September deployment. No control IRIS film deployments met the standard for the entire field season.

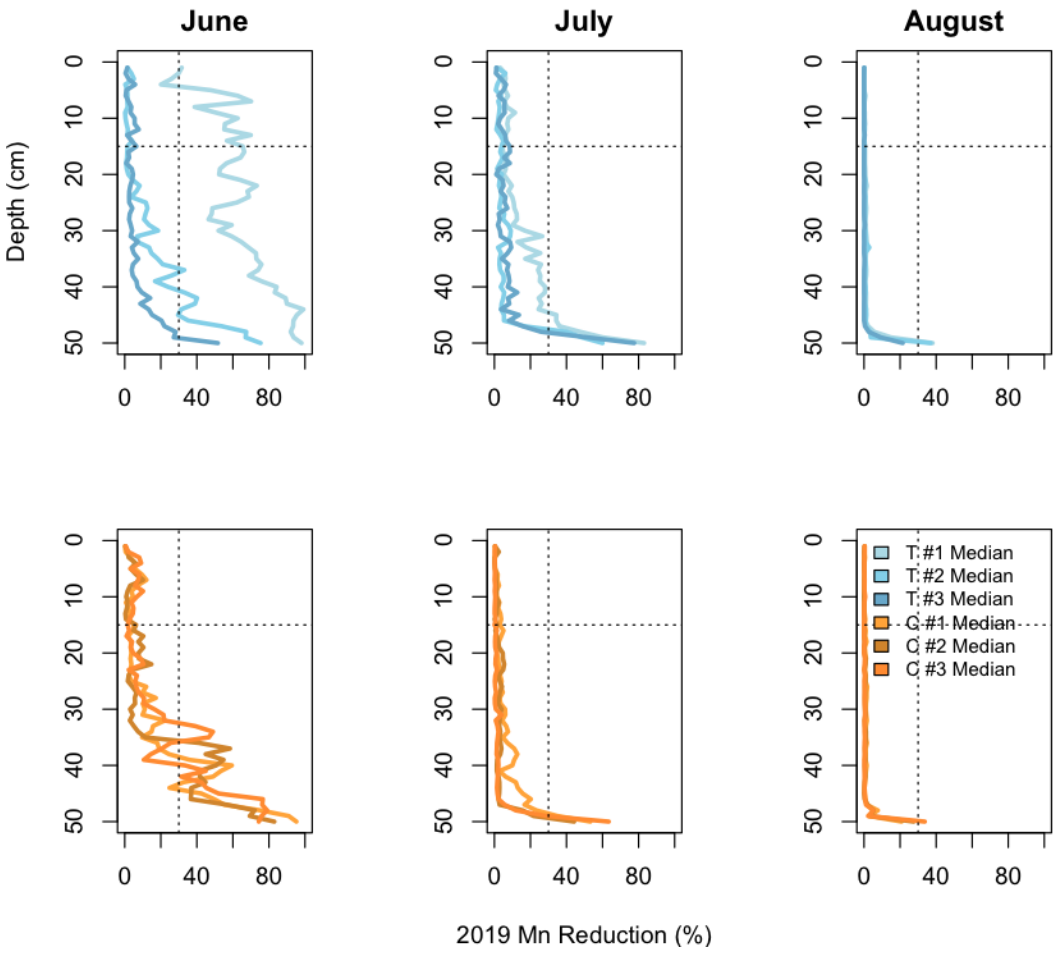


Figure A9: 2019 Median Mn IRIS results used to assess Hydric Soil Standard. Treatment plot #1 met the standard during the June-July deployment, while no other treatment or control deployments met the standard for the entire field season.



Figure A10: 2018 and 2019 field Season Total Film Reduction. Simple linear regressions were calculated for each field season by treatment and control. A linear regression for T18 (F-statistic= 80.76 on 1 and 118 DF, p-value: 1 e-14, Multiple R-squared= 0.41) and for C18 (F-statistic= 99.44 on 1 and 118 DF, p-value: < 1e-15, Multiple R-squared= 0.46) were calculated and explained approximately 40% of the variation in fit. For T19, the linear regression explained less of the variation (F-statistic= 21.13 on 1 and 43 DF, p-value: 3.75e-05, Multiple R-squared= 0.33), which differed from CT19 (F-statistic= 40.25 on 1 and 42 DF, p-value: 1.271e-07, Multiple R-squared= 0.49) which explained almost 50% of the fit.

APPENDIX B

SOIL MOISTURE AND TEMPERATURE

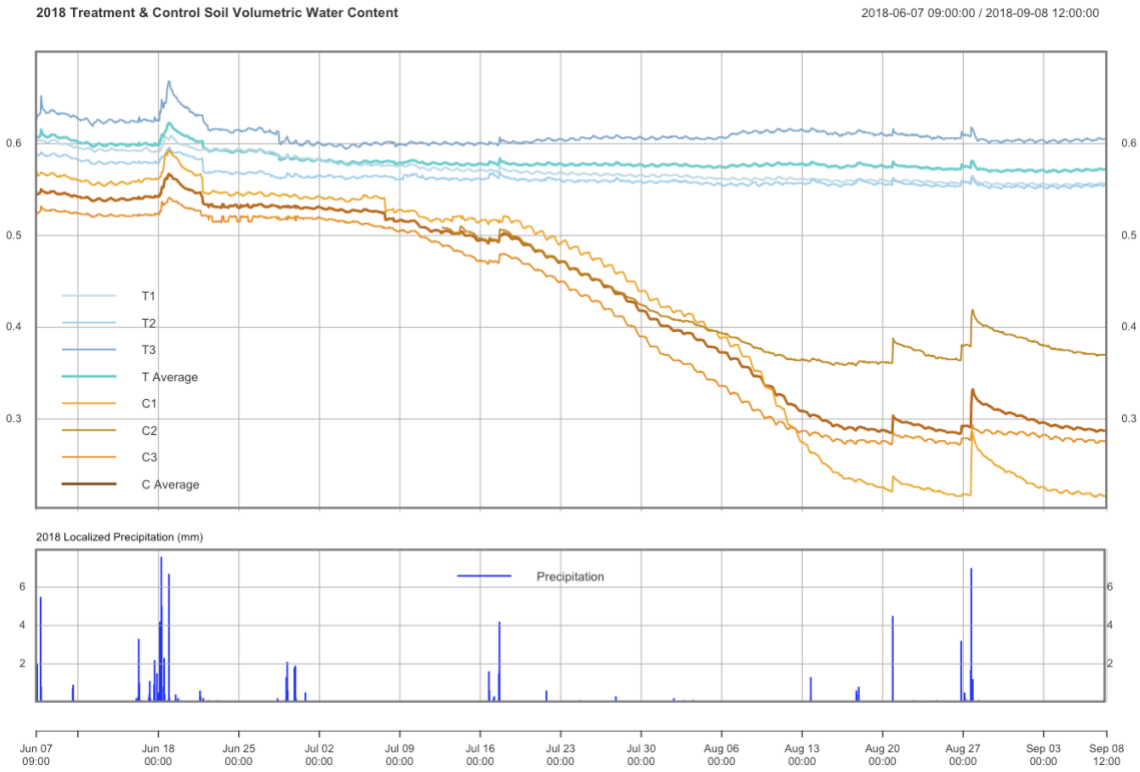


Figure B1: 2018 plot specific soil moistures over the field season with associated precipitation events.

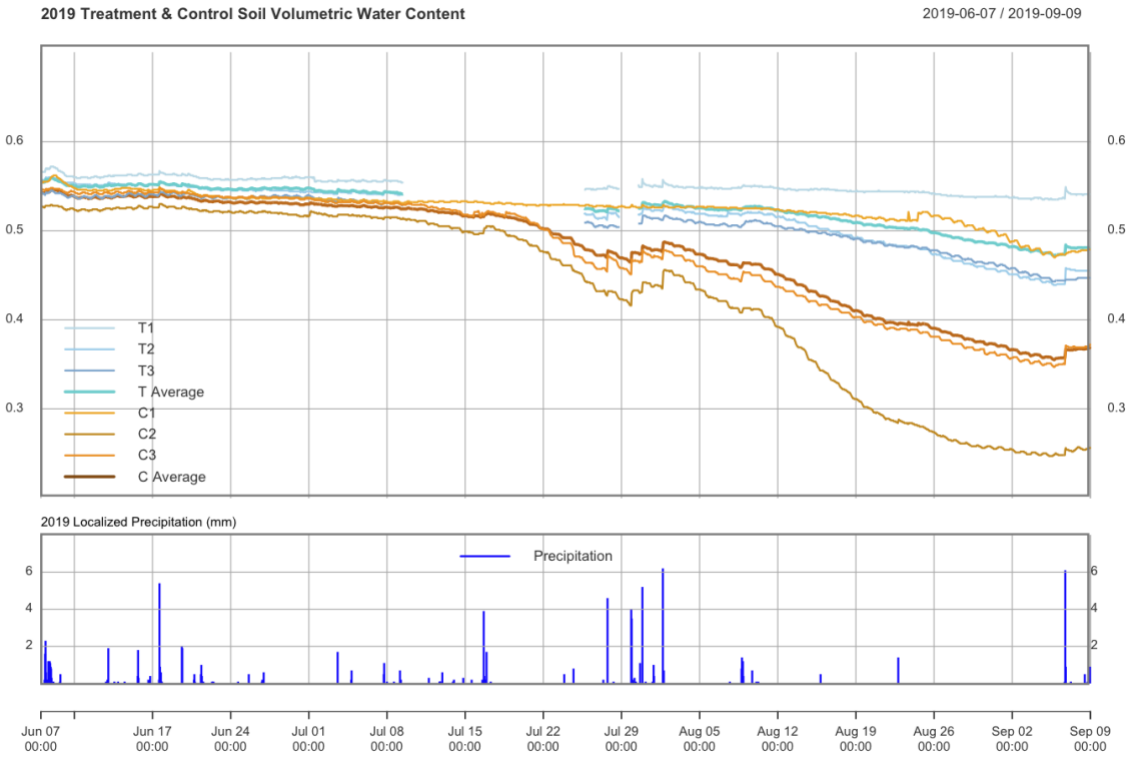


Figure B2: 2019 plot specific soil moistures.

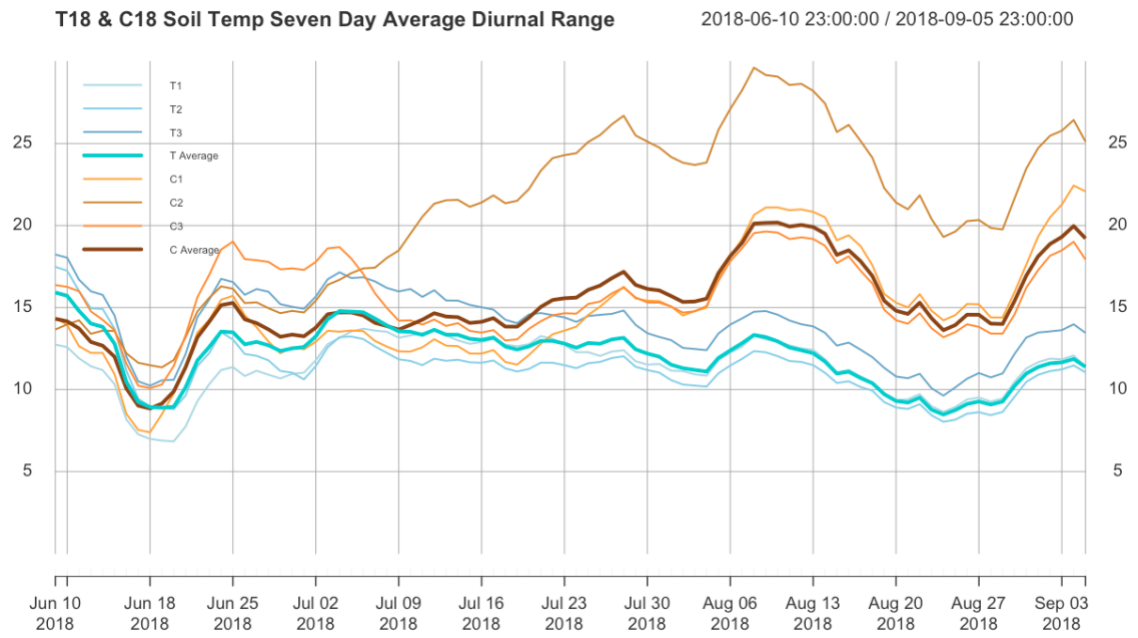


Figure B3: 2018 Plot specific soil temperatures.

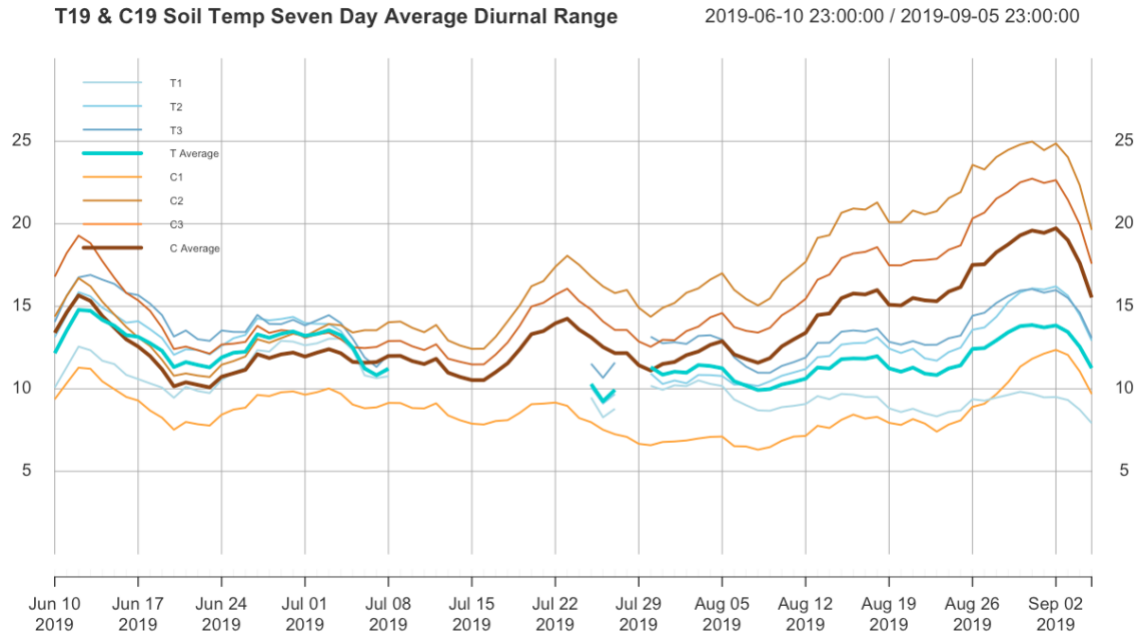


Figure B4: 2019 plot specific soil temperatures.