HYDROLOGIC INFLUENCE OF WETLAND RESTORATION:

THE STORY MILL CASE STUDY

by

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DEDICATION

I would like to thank many people for their patience, thoughtfulness, and honest, un-patronizing guidance during this process, and when I questioned my path and identity often. You know who you are. I would also like to thank my advising committee - what an opportunity to work with each of you - challenging me in different ways, technically and creatively. But ultimately I would like to dedicate this to my Grandfather, Daniel Warrick Colhoun Jr., for his amazing capacity to be truly interested in my endeavors, challenge me to think past my project, and engage in what I have to say as though I am a colleague. You make me feel respected and proud.

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ABSTRACT

The Story Mill Wetland is a 20 hectare restoration project in Bozeman Montana, intended to help improve the quality of surface water that leaves the city. The streams that border the property, Bozeman Creek and the East Gallatin River, exceed the Montana Department of Environmental Quality's (MTDEQ) water quality standards for nitrogen (0.27 mg/L) and phosphorus (0.08 mg/L). Wetlands in the landscape have become more intriguing in the advent of MTDEQs adoption of Circular DEQ-13, a legal framework for nutrient trading to achieve improved watershed water quality. Earth-work took place in Summer/Fall 2014, including excavating 5,800 m^2 of disconnected floodplain, and filling a surface drain. The research objectives were to quantify the impacts of the restoration so as to make inferences about the short-term changes in groundwater/surface water interaction, wetland volume and area, and the wetland's impact on the water quality in the bordering streams. Measurements of groundwater levels and surface water flow rates, and water chemistry analyses for both water sources, were recorded weekly from 30 shallow wells and 5 stream gauging stations from August 2014 through September 2015. Groundwater velocity and hydraulic residence time were estimated by performing slug tests in several groundwater wells. Spatially normalized wetland area and volume were calculated based on interpolated groundwater surfaces. Throughout the monitoring period, in all surface and groundwater samples, total nitrogen never exceeded 3 mg/L, averaged 0.76 mg/L, and almost always exceeded the target standard for the East Gallatin River. Total phosphorus was below the detection limit in 97% of all samples and never exceeded 0.22 mg/L. Neither average nutrient concentrations nor pH showed significant general temporal trends, while dissolved oxygen decreased over time. Changes in hydrology were generally localized near earth moving activity. Overall, wetland volume decreased slightly and wetland area increased slightly. Hydraulic gradients showed the primary flow of groundwater to be out of the wetland, with an average soil water velocity of 0.11 m/day. The slow moving groundwater in the wetland system appears to limit the extent of groundwater/surface water interaction, and hinders the role of the wetland in enhancing the water quality in the receiving creeks.

INTRODUCTION

The Missouri River, collecting water from the largest watershed in the United States, begins in the mountains of Montana, as tiny tributaries to the Gallatin, Madison, and Jefferson Rivers (Missouri Department of Natural Resources, 2015). These three rivers join and flow southeast as the Missouri makes its way through countless acres of farm and range land, forests, and urban areas in Montana, the Dakotas, Nebraska, Iowa, Kansas, Missouri, and Illinois. Of these eight states, all but Iowa, make the top ten list for cattle inventory (United States Department of Agriculture, 2016). These 29 million cows deposit about 140 thousand tons of manure annually in the Missouri watershed (Chastain et al., 2016), in addition to the millions of tons of nitrogen and phosphorus fertilizer that are spread all across America's breadbasket (USDA Economic Research Service, 2013). The Missouri River flows through many major cities like Omaha, Kansas City, and St Louis – collecting and carrying all of their stormwater and municipal waste. But it all starts in Montana, as pristine mountain streams.

Along the length of American rivers, there are opportunities to alleviate the compounding effects of anthropogenic contamination. This can be achieved either by limiting or remediating pollution. The City of Bozeman is located in the fertile Gallatin Valley, between the Bridger and Gallatin Mountain Ranges, and some of those tiny tributaries that feed the Gallatin River run through the streets of Bozeman. Although Bozeman's population is a mere 41,660, compared to Kansas City's 459,787 (United States Census Beureau, 2014), Bozeman still struggles to meet EPA's water quality standards, and passes along water with elevated levels of nitrogen and phosphorus. In fact, 17,159 streams and rivers across Montana are considered polluted (US EPA Office of Water, 2015). Natural riparian wetlands are a possible water quality improvement measure that could prove to be a low impact and effective tool to ensure that Montana's waters remain pristine. Wetlands have become an even more intriguing solution in light of Montana DEQ's adoption of Circular DEQ-13, a legal framework for nutrient trading to achieve improved water quality. It is understood that in many instances technology and cost can be limiting for point source polluters to improve their effluent quality. This document provides an avenue to effectively allocate resources and consider a watershed as a whole.

Regulations exist to "protect against adverse effects of nutrient over-enrichment from cultural eutrophication (US EPA Office of Water, 2000)." In accordance with the Clean Water Act, the EPA created a document in 2000 called the Ambient Water Quality Criteria Recommendations. The intent of this document was to encourage and guide states and tribes to create numeric nitrogen and phosphorus water quality standards, whereas before, water quality laws have been qualitative. Montana DEQ was tasked with the responsibility to interpret and refine the guidelines, based on local water quality data and knowledge (US EPA Office of Water, 2000). Subsequently, the MT-DEQ Circular 12: Montana Base Numeric Nutrient Standards was written and adopted. The target values were determined so as to protect each water body's most sensitive use from harm, be it drinking water, fish and aquatic life, and/or recreation (Suplee et al., 2008).

Extensive research was compiled and careful consideration was taken to categorize water bodies and assign specific standards. It was determined that each water body would be classified by ecoregion so as to best take into account regional geology, soils, climate, and vegetation. Specific attention was also paid to temperature and flow patterns because of their influence on algae growth and dissolved oxygen levels. Many stressor-response studies were reviewed from Montana and around the world, on how varying levels of nitrogen and phosphorus impact beneficial use (Suplee et al., 2008). Although both Bozeman Creek and the East Gallatin River - the water bodies of concern for this research - fall into the Middle Rockies Ecoregion, each have their own specific targets, and are shown in Table 1 (Montana Department of Environmental Quality, 2014). Note that developed criteria only apply during the summer months because this is when eutrophication is most likely to occur (Suplee et al., 2008). Also note the absence of nitrate+nitrite target values, as they have not yet been adapted into law (Suplee, 2013), although DEQ has made unofficial recommendations, which are shown in Table 2 (Montana Department of Environmental Quality, 2013).

Individual Stream or Reach Description[2]	Period When Criteria Apply[3]	Numeric Nutrient Standard[4] Total Phosphorus (µg/L)	Numeric Nutrient Standard[4] Total Nitrogen (µg/L)
Wadeable Streams: Clark Fork River basin			
Flint Creek, from Georgetown Lake outlet to the ecoregion 17ak boundary (46.4002, -113.3055)	July 1 to September 30	72	500
Wadeable Streams: Gallatin River basin			
Bozeman Creek, from headwaters to Forest Service Boundary (45.5833, -111.0184)	July 1 to September 30	105	250
Bozeman Creek, from Forest Service Boundary (45.5833, -111.0184) to mouth at East Gallatin River	July 1 to September 30	76	270
Hyalite Creek, from headwaters to Forest Service Boundary (45.5833,-111.0835)	July 1 to September 30	105	250
Hyalalite Creek, from Forest Service Boundary (45.5833,-111.0835) to mouth at East Gallatin River	July 1 to September 30	90	260
East Gallatin River between Bozeman Creek and Bridger Creek confluences	July 1 to September 30	50	290
East Gallatin River between Bridger Creek and Hyalite Creek confluences	July 1 to September 30	40	300
East Gallatin River between Hyalite Creek and Smith Creek confluences	July 1 to September 30	60	290
East Gallatin River from Smith Creek confluence mouth (Gallatin River)	July 1 to September 30	40	300
Large Rivers[6]:			
Yellowstone River (Bighorn River confluence to Powder River confluence)	August 1 -October 31	55	655
Yellowstone River (Powder River confluence to stateline)	August 1 -October 31	95	815

Table 1. Numeric surface water quality standards. Developed by the Montana Department of Environmental Quality for specific streams in Gallatin County, Montana (taken from Montana Department of Environmental Quality (2014)). Rivers of interest for this project are Bozeman Creek, from the Forest Service Boundary to mouth at East Gallatin River, and the East Gallatin River between Bozeman Creek and Bridger Creek confluences.

Parameter	Criteria
Nitrate+Nitrite Nitrogen	≤ 0.100 mg/L
Total Nitrogen	≤ 0.300 mg/L
Total Phosphorus	≤ 0.030 mg/L
Benthic Algae	≤ 129 mg/m²

Table 2. Suggested nutrient standards. Includes Department of Environmental Quality recommendation for nitate+nitrate for streams in the Middle Rockies Ecoregion in Montana (taken from Suplee (2013)).

The Story Mill Wetland is a restoration project in Bozeman Montana which may help improve water quality in the waters that leave city limits. The site of the restoration is a 20 hectare parcel, ideally situated at the confluence of Bozeman Creek and the East Gallatin River, just downstream of the City. Both creeks tumble down from the Gallatin Range, gathering water from a 33,000 ha watershed, through agricultural fields, past subdivisions, alongside, and under the streets of Bozeman - each a potential source of nutrients. The Story Mill wetland is located in a low sloping alluvial fan, where river worn, rounded gravel is topped by centuries of loess deposits (Montagne et al., 1982). This rich soil, and a shallow groundwater table support an array of water loving grasses and the occasional mixed stand of aspens, cottonwoods and willows.

This parcel is speculated to have historically been a wetland that was dredged and/or filled and dewatered for agricultural production late in the 19th century (US Department of the Interior National Park Service, 1996). Industrial and office buildings have encroached its western and southern borders, and a large storage complex was built in the middle of the site, but remains outside of the project property boundary. The area is now recognized by the City of Bozeman and the Gallatin Local Water Quality District as a “high priority site for implementing wetland restoration activities to improve water quality in the East Gallatin River and Bozeman Creek as well as enhance wetland function (Pope et al., 2014).”

Figure 1. Aerial view of the Story Mill site before construction.

The Trust for Public Land bought the property in December 2012, and is partnering with the City of Bozeman to convert the abandoned land to a public park. The design for the park is multifaceted, and is intended to include playgrounds, sports fields, and walking trails. As discussions and plans progressed, it was realized how valuable this location was for a wetland restoration, and extensive ecological goals for the park's development began to take form. These goals are stated in the Story Mill Ecological Restoration Selected Alternative Conceptual Design Report (RESPEC Consulting and Services, 2013), and are as follows:

> The overarching ecological goal for the project is:
> ***In consideration of site constraints and other project goals, restore and protect on-site natural processes necessary for a functioning riparian and wetland system.***
> This goal is supported by the following five ecological objectives:
>
> - E-1 Provide hydrologic connectivity between stream floodplain and wetlands to maximize riverine and wetlands habitat diversity.
> - E-2 Remove river process constraints and non-natural features to the extent possible in the context of land ownership and access.
> - E-3 Remove or modify drainage and excavated features that disrupt and diminish groundwater-dependent wetland extent and functioning to restore wetland functions to the extent site constraints allow.
> - E-4 Demonstrate improved water quality (temperature, nutrients and sediment measures).
> - E-5 Restore native plant diversity (upland, wetland and riparian communities) and minimize invasive plants.

The major earth work and construction for the restoration took place in two stages, and was intended to reconnect the hydrology of the rivers to the wetland, and increase the area and function of the wetland (RESPEC Consulting and Services, 2013). In September and November 2014, efforts were focused on the Southern and Triangle Parcels, and the streams. Activities included excavation of fill from floodplains, re-contouring of the pond, removing rip-rap from the stream banks, and eliminating a ditch that may have once dewatered the site. Then in September 2015, restoration of the Northern Parcel took place where more fill was removed from the floodplain along the East Gallatin River.

Research on the Story Mill wetland began in August 2014, and continued until September 2015. Measurements of groundwater levels and quality, and surface water flow rate and quality, were taken before and after restoration construction activities. Seven water quality parameters were measured: total nitrogen, nitrate, total phosphorus, oprthophosphate, sulfate, dissolved oxygen, and pH. Groundwater velocity was also estimated. The objectives of this research were to assess the impacts of the restoration in the Southern Parcel (including excavating 5,800 m^2 area to be accessible by a two year flood in Bozeman Creek, filling a drainage ditch, and recontouring various aspects of the site) on groundwater levels and groundwater quality, up to one year after construction. These results were then used to make preliminary inferences about groundwater/surface water interaction, both flow direction and magnitude, wetland volume and area, and the wetland's impact on water quality in the groundwater, Bozeman Creek, and the East Gallatin River.

With the parameters measured, the desired response to the restoration would be to observe an increase in wetland area perpetuate a decrease in dissolved oxygen levels, and a decrease in groundwater nutrient levels. Ideally, an increase in wetland area would also cause any observed pollution entering the wetland to be more effectively removed after the restoration.

The research presented here is a localized and detailed look at water quality and processes at Story Mill, but it fits into the larger context of water quality monitoring and concerns, and nutrient trading in the East Gallatin River watershed. Just downstream of Story Mill, the Bozeman Water Reclamation Facility discharges to the East Gallatin River an average of 8.5 MGD, 33% of the river's summer flow. This state of the art facility produces some of the best effluent quality in the state of Montana (Water & Waste Digest, 2015) with 2015 average total nitrogen concentrations of 6.3 mg/L and total phosphorus concentrations of 0.23 mg/L (Bartle, 2016). The Greater Gallatin Watershed Council, with the help of the Local Gallatin Water Quality District has been collecting water quality data from a collective of 17 locations within the watershed since 2008. The map and table below (Figure 2 and Table 3) describe sampling locations within the East Gallatin watershed, and summarize historic water quality data.

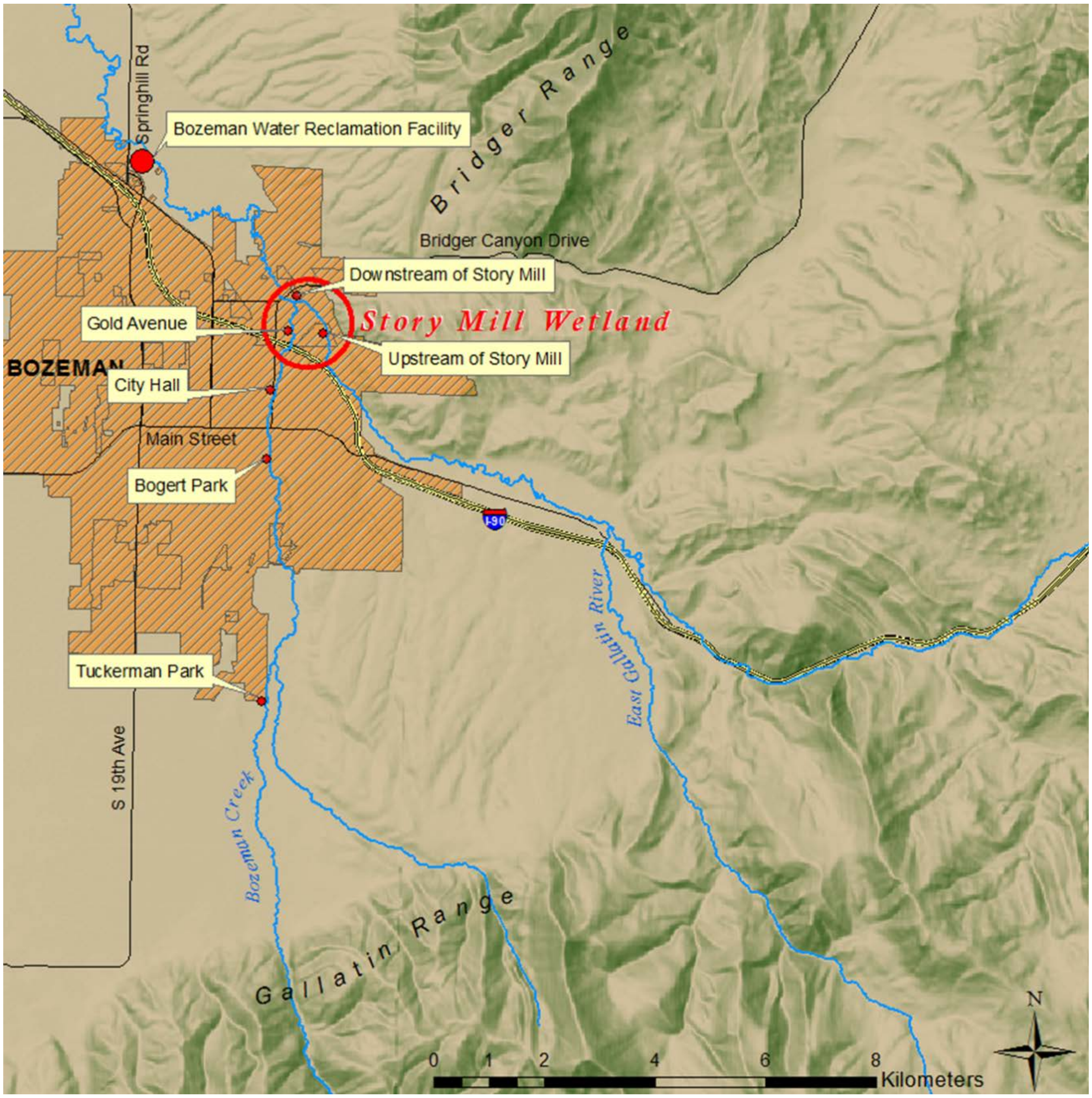

Figure 2. Locations of historic water quality data in the East Gallatin River watershed, relative to the Story Mill wetland. All locations have been monitored by The Greater Gallatin Watershed Council, with the help of the Local Gallatin Water Quality District, except for the Bozeman Water Reclamation Facility.

Bozeman Creek[1]		
	Total Nitrogen (mg/L as N)	Total Phosphorus (mg/L as P)
Target[2]	0.27	0.076
Tuckerman Park	0.55-1.57	0.03-0.06
Bogert Park	0.5-1	0.02-0.06
City Hall	0.05-1	0.03-0.07
Gold Avenue	0.04-1	0.02-0.06
East Gallatin River[1]		
Target[2]	0.3	0.03
Upstream of Story Mill	0.07-0.2	0.01-0.035
Target[2]	0.29	0.05
Downstream of Story Mill	0.06-0.21	0.01-0.04
Bozeman Water Reclamation Facility[3]		
Effluent standards[3]	7.5	1
Average annual (2015)[4]	6.3	0.227

Table 3. Historic water quality data for the East Gallatin River watershed. Locations correspond to those detailed by the map in Figure 2. Nutrient concentration ranges encompass all available data for a particular site. Sources are as follows: (1) Greater Gallatin Watershed Council (2) Montana Department of Environmental Quality (2012) (3) Revis (2012) (4) Bartle (2016)

LITERATURE REVIEW

Wetlands are known to be vital to ecosystem health, providing a range of benefits and functions. When considering wetland restoration design, it is important to understand that many factors determine if, and how well, a wetland will perform its intended purpose. This literature review focuses on a wetland's role in affecting water quality, namely nitrogen and phosphorus cycling, with the intention of understanding the Story Mill wetland restoration, its immediate effects, and long-term potential on water quality.

First and foremost, it is worth becoming familiar with the definition of a wetland and how they are delineated. Next, hydrology will be discussed. Hydrology is a unifying theme throughout the thesis, and it will become evident that it is the cornerstone driving wetland function. Then, each nutrient's forms and sources and basic cycling and transformation mechanisms will be covered. Creating this base of understanding is important to realizing the significance of the interplay between hydrology and water quality. Within each nutrient section, case studies will be discussed, in the context of deciphering what has worked elsewhere, and what has not worked, all with the goal of framing the potential of these unassuming boggy depressions.

Wetland Delineation

A wetland is defined by hydrology that creates hydric soils and supports hydrophytic vegetation, and is delineated by finding sufficient indications of each: wetland hydrology, soil, and vegetation (Environmental Laboratory, 1987). Wetland delineation has been a contentious political issue since the inception of the Clean Water Act in 1972, involving many supplements, proposed revisions, reversals, and adoptions. Although technical specifications may vary, the concepts remain the same throughout all documents (National Research Council, 1995). Montana refers to the 1987 Corps of Engineers Wetlands Delineation Manual (National Center for Manufacturing Sciences, n.d.). Wetland delineations are intended to represent average conditions of the site, both temporally and spatially. Length of monitoring is determined on a case-by-case basis depending on the complexity and sensitivity of the potential wetland. Rarely is a comprehensive approach deemed necessary, but when it is, a minimum of 10 years of observation and data must be analyzed to determine average site conditions. Ultimately, it comes down to the user of the manual to "be familiar with wetlands of the area and use his or her training, experience, and good judgment in making wetland determinations" (Environmental Laboratory, 1987).

The structural indicators used to delineate a wetland are intended to be evidence of function: they are to ensure that a particular area is indeed performing its intended ecosystem services. When a conscientious design is used to restore and leverage a wetland for water quality enhancement, many other ecosystem services are also provided by default. These additional functions are diverse and numerous ranging from providing flood control, habitat, and a sink for carbon (W. J. Mitsch et al., 2007; C. J. Richardson, 1994; Zedler, 2003). Perhaps most importantly, a wetland provides ***resilience*** within an ecosystem (Tanner, 2015): the ability to dampen the fluctuating moods of Mother Nature, and more recently, human folly.

Hydrology

Hydrology is considered "the overriding influence" responsible for the occurrence of the other delineation indicators: the formation of wetland soils, and for the occurrence of wetland vegetation (Environmental Laboratory, 1987; Faulkner et al., 1992). When water is slowed or captured by soils with low hydraulic conductivity and by depressions in the topography, it inundates the landscape, and saturates the soil. Water sources for a wetland may be from groundwater, surface water, precipitation, and/or tidal flow (C. J. Richardson, 1994). Under these saturated conditions, dissolved oxygen becomes a scarce commodity, causing the formation of indicators used to categorize hydric soils (W. J. Mitsch, Zhang, et al., 2005; Natural Resources Conservation Service, 2010) and selecting for uniquely adapted hydrophytic vegetation (Environmental Laboratory, 1987; W. J. Mitsch, Zhang, et al., 2005; National Research Council, 1995; Reed, 1988). Where a water level data point may tell something about right now, a soil profile and vegetation assemblage tells a more long term story of site conditions: in a sense, the soil and plants have a memory. Delineation requires that field data and/or recorded data show evidence of inundation or saturation within 30.5 cm of the soil surface (rooting zone) for at least 12.5% of the growing season, or 5-12% of the growing season with the observation of other indicators. Other hydraulic indicators include water marks, drift lines, sediment deposits, and drainage patterns within wetlands (Environmental Laboratory, 1987).

Just as hydraulics and hydrology are crucial themes in creating the indicators for wetland delineation, so are they in fulfilling function. A wetland's hydrologic connectivity within a landscape is key to this structure-function relationship. Mitigated or restored wetlands without connectivity to the potential source of flooding or pollution cannot act as an emergency overflow, or improve water quality (C. J. Richardson, 1994; Zedler, 2003). With urbanization and anthropogenic pressures, natural wetlands can become isolated and hydrologic interactions are prevented by incision and containment (Walling, 1999). Andersen (2004) examined a system similar to the Story Mill wetland, and its role in nitrogen cycling. The Danish floodplain wetland studied was typically inundated by floodwaters 4 times a year, more frequently than the 2-year flood events for which the Story Mill restoration was designed. Dr. Andersen determined with a hydraulic model that, other than these flood events, there was very little interaction between the surface water and groundwater in the wetland. He concluded this is the reason for the removal of only a small mass of nutrients, and that the wetland is being underutilized based on its potential. In a watershed scale wetland restoration in Durham, NC, C. J. Richardson et al. (2011) attributed the significant improvements measured in water quality to re-establishing the hydraulic connection between the stream and its adjacent floodplains. This type of research conducted by both Anderson and Richardson et al. emphasizes the value of understanding site hydrology in order to quantify the effect of wetlands on water quality (Andersen, 2004; C. J. Richardson et al., 2011).

Hydrology also plays a major role in oxygen availability, a key factor in nutrient cycling and removal mechanisms. Certain stages of nitrogen and phosphorus transformations require oxygen, while others will not proceed if oxygen is available; each will be discussed in further detail in following sections. Undisturbed soil typically has 30-40% void spaces (Geotechdata.info), which in unsaturated soil, are filled with air. But persistent saturation fills those voids with water and decreases oxygen transfer into the soil column, creating anaerobic conditions (Natural Resources Conservation Service, 2010). Often in natural systems, parts or all of a wetland will go through varying levels of inundation due to flood or draught conditions, and riparian wetlands often get pulsed with storm water flooding. Varying levels of saturation and diverse environmental niches caused by the hydrology unique to wetlands are crucial to resulting water quality (Corstanje et al., 2004; Lockaby et al., 1996; W. J. Mitsch et al., 2007).

Hydrology and hydraulics not only dictate water levels and saturation, they control water flow paths and velocity too, parameters that also influence a wetland's ability to remove nitrogen and phosphorus (Spieles et al., 1998). Removal mechanisms, such as plant uptake, denitrification, and settling (all of which will be discussed in greater detail in following sections), are processes that take time. The time it takes for water to flow through a bioreactor, including a wetland, is called hydraulic residence time (HRT), and is a function of volume contained in the system and the flow rate through the system:

$$HRT = \frac{V}{Q}$$

where V = volume of water within a wetland, and Q = flow rate through the wetland. HRT can be increased by either an increase of the wetland volume or a decrease of inflow (Persson and Wittgren, 2003). The longer pollutants are retained in a wetland bioreactor, the longer and more thorough the exposure is, and the greater chance they will be removed by soil particles, plants, or microbes (Persson, 2000). In a system where HRT is too short, it can be the limiting factor for treatment as nutrients remain in the water system, and risk being flushed back out into the water source. In systems where HRT is not limiting, and that are instead limited by nutrient supply, an increase in HRT will have no effect on treatment capacity (Andersen, 2004).

The velocity of surface water flowing through a riparian zone or wetland is a major factor in particle dynamics. Nitrogen, and especially phosphorus, are often associated with particulates: bound as organic matter, sorbed to soil particles, or precipitated as a solid (Sparks, 2003). Slow velocities allow for sediment to settle out of solution, whereas high velocities risk scouring wetland soils, re-suspending sediments and any associated nitrogen and phosphorus with it (Johnston, 1991). Stream velocity is typically estimated using Manning's Equation:

$$V = R^{2/3} * S^{1/2} * {}^{1}\!/_{n}$$

where V = flow velocity, R = hydraulic radius, S = stream gradient, and n = roughness coefficient. Manning's Equation makes several assumptions, and its value is limited in calculating flow through a natural wetland, namely because of irregularities in topography. Fundamentally, the general relationships hold true, and it can be seen by this equation that slow velocities in wetlands are obtained by spreading shallow flow out over a large area with a gentle gradient, and a robust plant community (Johnston, 1991; Knutson, 1987). It has consistently been found that a surface water flow regime with high velocities and a short HRT results in a wetland actually becoming a source of nutrients to the receiving water body (J. G. Cooke, 1994; Raisin et al., 1995; Spieles et al., 1998). Spieles et al. (1998) found that during extreme flooding events, removal efficiency negatively spiked to -400%. If they ignored these events, the wetland's removal efficiency improved from 35% to 56%. This research emphasizes the value of considering site hydrology and incorporating surface water hydraulics in wetland design.

Nitrogen Removal

Forms and Sources

Nitrogen exists in many different forms, some of which are benign, and some of which are harmful to an ecosystem at certain concentrations. It also has many different avenues for entering water systems, and undergoes transformations between forms in a process generally referred to as the nitrogen cycle. Ammonium, nitrite, and nitrate are all inorganic forms of nitrogen that are essential nutrients for plants and microbes. Therefore when these compounds are in water at moderate concentrations, they can be the culprits responsible for eutrophication, and at high concentrations some are toxic to plants, animals, and humans (Montana Department of Environmental Quality, 2014; Section, 2012). Natural sources of inorganic nitrogen in water begin first as either organic N, bound up with carbon in all living, dead, and decaying things (plants, microbes, manure…), or as ammonium fixed in the soil directly from the atmosphere from lightning strikes and through certain plant rhizomes (Lamb et al., 2014). Organic N is initially harmless to a waterbody, but as the organic matter decays, it is mineralized to inorganic forms, the first being ammonium (Kadlec et al., 2009). However, most nitrogen in waterbodies comes from anthropogenic sources: organic forms coming from livestock operations, pets, and outdated or un-serviced septic systems, or inorganic forms such as synthetic fertilizer used in agriculture and lawn care or point sources such as wastewater treatment plants (United States Environmental Protection Agency). Once in an inorganic form, nitrogen can be removed from water by immobilization, or nitrification of ammonium to nitrate and subsequent denitrification to gasses that escape to the atmosphere (Kadlec et al., 2009).

Nitrogen concentrations in water vary widely depending on the source. Nitrogen in rainwater is primarily nitrate and ammonium, and in Montana, averaged 0.23 mg N/L total nitrogen in 2014 (National Atmospheric Deposition Program 2015). Nitrogen in agricultural runoff is mostly nitrate, and concentrations depends on several management, soil, and environmental factors, but has been reported to vary between 3 mg/L (Hoffmann et al., 2006) and 20 mg/L (Moreno et al., 2007). EPA speculates that non-point source runoff from agriculture is the largest polluter of streams in Montana (United States Environmental Protection Agency). Nitrogen in wastewater can be in many different forms and is therefore typically reported as total nitrogen. Total nitrogen in domestic wastewater also depends on many factors, but is approximately 50 mg/L (Fan et al., 2013; Tuncsiper, 2009).

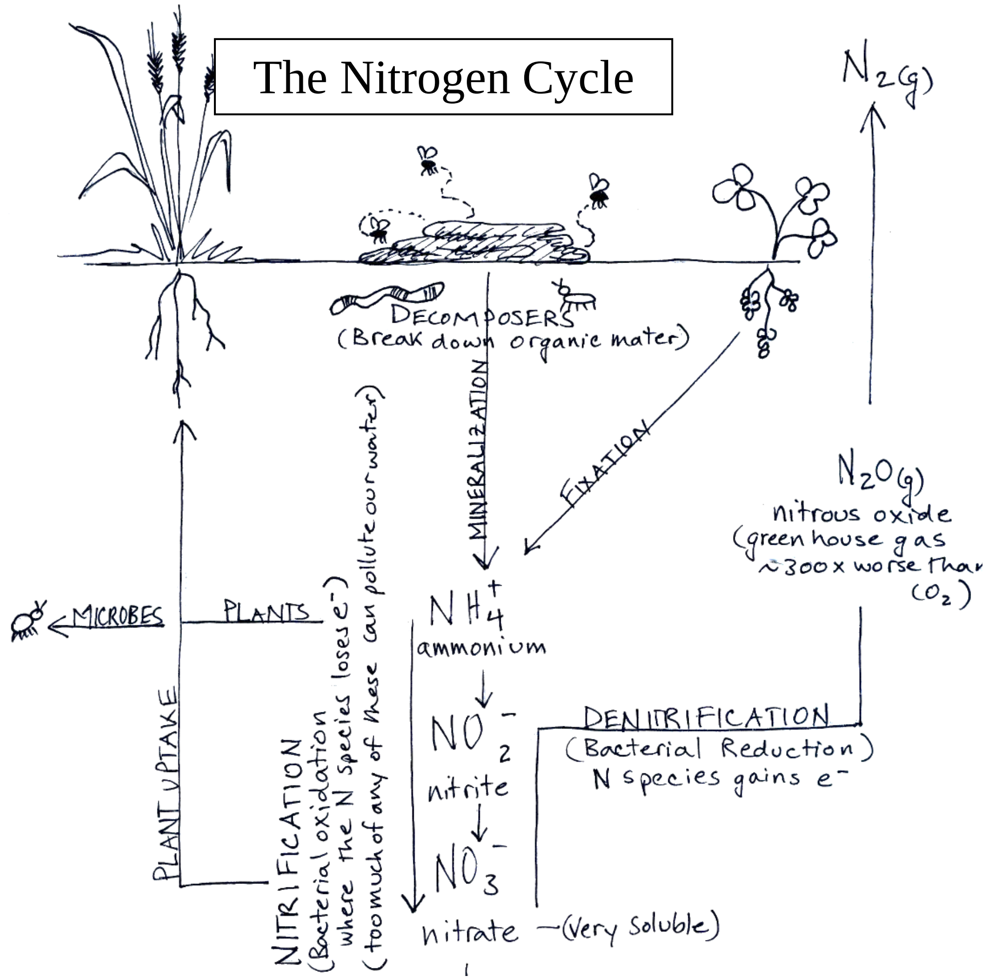

Figure 3. The nitrogen cycle is, for the most part, driven by plant and microbial processes.

<u>Plant and Microbial Assimilation</u>

Ammonium and nitrate can be used by plants and microbes as food - like a fertilizer - and immobilized from an inorganic form back into a complex organic form as biomass, removing it from the water. Uptake of nitrogen by plants can be a significant contributor to nitrogen removal, but it depends on many factors. Hydrology plays a major role in nitrogen uptake, because if the groundwater table is below the rooting zone, any nitrogen in the groundwater cannot be accessed by the vegetation: as the depth to groundwater fluctuates, so too does the role of plants in water quality (Hoffmann et al., 2006). Plant type and vigor also dictate the amount of nitrogen uptake (Hermanson et al., 2000). Vegetation typical to wetlands, especially emergent macrophytes, produce large amounts of biomass annually, and uptake lots of nitrogen (Craft, 1997). Vegetation typical of Story Mill, such as reed canary grass (*Phalaris arundinacea*), have been reported to store between 48 and 60 g N/m^2 in a growing season (Vymazal, 2005). The timing of nitrogen application is another consideration, since plants use nitrogen primarily during the spring and summer for growth, but little to none during their dormancy in late fall through the winter (Hermanson et al., 2000). Furthermore, it has been shown that the amount of nitrogen applied can impact the part plants play: biomass creation and nitrogen uptake is maximized when there are no other limiting factors to growth, such as other essential nutrients, water and sunlight. In other words, the more polluted with nitrogen the influent is, the more the plant community will remove, up to an asymptote (different for each plant type). A generalized relationship is shown in Figure 4 (Hermanson et al., 2000).

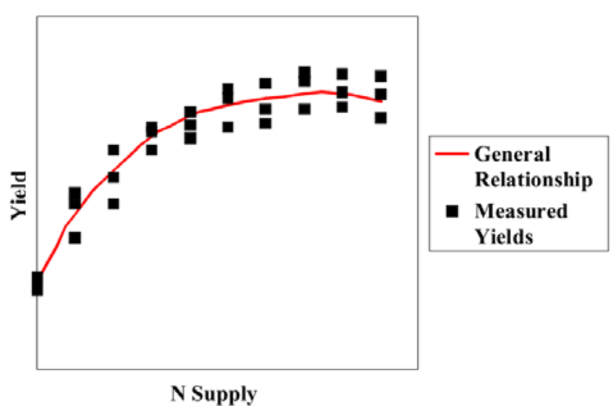

Figure 4. A yield response curve showing the relationship between available nitrogen and crop yield, demonstrating that yield increases with an increase in available nitrogen, up to a point (taken from Hermanson et al. (2000)).

After restoration activity, where the ecosystem has been significantly altered, it takes time for vegetation to re-establish (Craft, 1997). Once vegetation is established, plants' annual contribution to water quality improvement levels off to a more consistent removal rate. In mature, verdant systems that are unlimited by nitrogen, plant removal rates can improve from 6 to 14 g $N/m^2/yr$ (Craft, 1997; Craft et al., 1993). In summary, to maximize nitrogen removal by plant uptake, during the growing season it is important to have an elevated water table in the horizon that plant roots occupy, with high levels of nutrient to support a healthy and robust plant community.

Plants also play an important role in below-ground ecology, providing a vast network of roots for microbial attachment, and root exudates of oxygen and carbon assisting in the removal of nitrogen through nitrification and subsequent denitrification (Weisner et al., 1994). In a study of fluctuations in microbial biomass in wetland soils, X. L. Wang et al. (2016) found a strong positive correlation between microbial biomass and plant biomass, likely due to increased carbon availability providing food for the microbial community. In a wetland soil, they found that microbial biomass fluctuated from 4 – 700 mg/kg of soil throughout the growing season.

This positive correlation between plant and microbial biomass is also reflected in water quality. Of the masses of microbes found in wetland soils, all are responsible for immobilizing nitrogen into their biomass, and many of them are nitrifiers and denitrifiers. Wen et al. (2010) observed that the difference between planted and unplanted constructed wetlands could not be completely explained by biomass removal, and that improved treatment in planted wetlands was apparent year round - not just during the growing season. This relationship has also been observed in a treatment wetland in Montana. Planted wetland cells performed significantly better than unplanted wetland cells throughout the winter when the wetland was buried beneath up to 5ft of snow (Moss, 2016).

Mineralization

The transformation of nitrogen into plant and microbial biomass is not permanent. Each year, much of the plant community's hard work - its stems and leaves, parts of its root system, its seeds - all return back to the earth, as do the bodies of microbes, and the recycling process continues. This biomass is now fodder for a host of new microbes, dismantling and decomposing, taking what was once inorganic nitrogen, and became complex organic nitrogen, right back to inorganic nitrogen in the form of ammonium. This process is called mineralization.

It is debated how fast nitrogen is turned over in a wetland, and if wetlands can actually accumulate organic nitrogen. Kadlec et al. (2009), in their Treatment Wetlands text, argue that in wetlands designed specifically for the treatment of wastewater, there is near complete recycling of any nitrogen accumulated in plant biomass within a year. Craft et al. (1988), on the other hand, showed that in a more natural wetland ecosystem, production can exceed decomposition, resulting in a net storage of organic N, where N accumulation rates ranged as high as 12 g $N/m^2/yr$ in a new wetland. Accumulation rates seemed to slow and stabilize to 2 g $N/m^2/yr$ as the observed wetlands aged and vegetation succession became constant (Craft, 1997; Craft et al., 1988; Craft et al., 1993). Lockaby et al. (1996) found that in ideal conditions for mineralization, a well oxygenated system that is not carbon or water limited, 80-85% of organic N was mineralized to inorganic N in 106 weeks. They go on to theorize that mineralization rates should decrease with less oxygenated conditions. Their research also found that in treatments receiving influent with elevated levels of nitrogen - 10 mg/L - decomposition rates were significantly depressed. They concluded this is due to the interference between nitrogen and lignin degradation, citing the work of Berg et al. (1991). Since inevitably a portion, and as some would argue, all nitrogen taken up by plants is re-mineralized, many constructed wetland systems for treating waste water opt to harvest plant biomass or assume the mechanism of plant uptake is inconsequential (Vymazal, 2005).

Nitrification

Once nitrogen in a wetland is in the form of ammonium, it can be removed by first being nitrified, or transformed to nitrate, and subsequently denitrified. Nitrification is microbial oxidation of ammonium to nitrite, and then to nitrate. This reaction occurs in the presence of oxygen, but only proceeds slowly, if at all, in the presence of organic carbon, and is influenced by temperature:

1. $$2\ NH_4^+ + 3\ O_2 \rightarrow 2\ NO_2^- + 2\ H_2O + 4\ H^+$$
2. $$2\ NO_2^- + O_2 \rightarrow 2\ NO_3^-$$

Nitrifiers are autotrophs, meaning they synthesize their own carbon from inorganic sources, and are therefore typically slower growing and get outcompeted by heterotrophs in the presences of organic carbon (Ling et al., 2005). Nitrification is not as important in natural systems, as opposed to waste water systems, because ammonium concentrations in natural waters are typically very low, and constitute a small fraction of the influent nitrogen (Hinkle et al., 2014). Surface water and most groundwater are usually exposed to plenty of oxygen, and nitrification has already taken place before the water source comes into contact with the wetland. Nitrogen removal rates have been shown to improve when the influent is comprised of nitrate, so as to skip the nitrification process (Boustany et al., 1997).

Denitrification

Nitrate-nitrogen can be completely removed from the wetland system through denitrification, or microbial reduction in the absence of oxygen with the help of organic carbon. Nitrate is first transformed to nitrite, then, to nitrous oxide, and finally to dinitrogen gas. Dinitrogen gas is benign and naturally makes up 80% of our atmosphere, but nitrous oxide is a potent greenhouse gas, 300 times worse than carbon dioxide in its capacity to warm the earth's atmosphere (United States Environmental Protection Agency, 2016b). Research is being done to better understand denitrification to hopefully improve wastewater treatment so as to encourage complete denitrification to dinitrogen gas, instead of impartial denitrification to nitrous oxide (W. J. Mitsch, Zhang, et al., 2005).

Denitrification and plant uptake are the two primary mechanisms of nitrogen removal, but their relative contributions depend on influent strength. Kadlec et al. (2009) compiled several literature sources and clearly showed a correlation between influent N concentration and the proportional role of plants: although the plants removed approximately consistent amounts of nitrogen, their percent of total removal decreased from 60% to 4%, as applied nitrogen increased from 8 g N/m^2/yr to 1183 g N/m^2/yr. It can be assumed that denitrification was responsible for the remaining removal of nitrogen although other, more minor pathways exist. Craft (1997) illustrate this relationship nicely in Figure 5.

Several factors affect the efficiency of denitrification within a system. Essentially, the healthier the denitrifying microbes, the better: a wetland system that is oxygen limited, within an optimal temperature range, and that has plenty of organic carbon and nitrate should support a healthy denitrifying community. Each factor affecting denitrification rates will be discussed below.

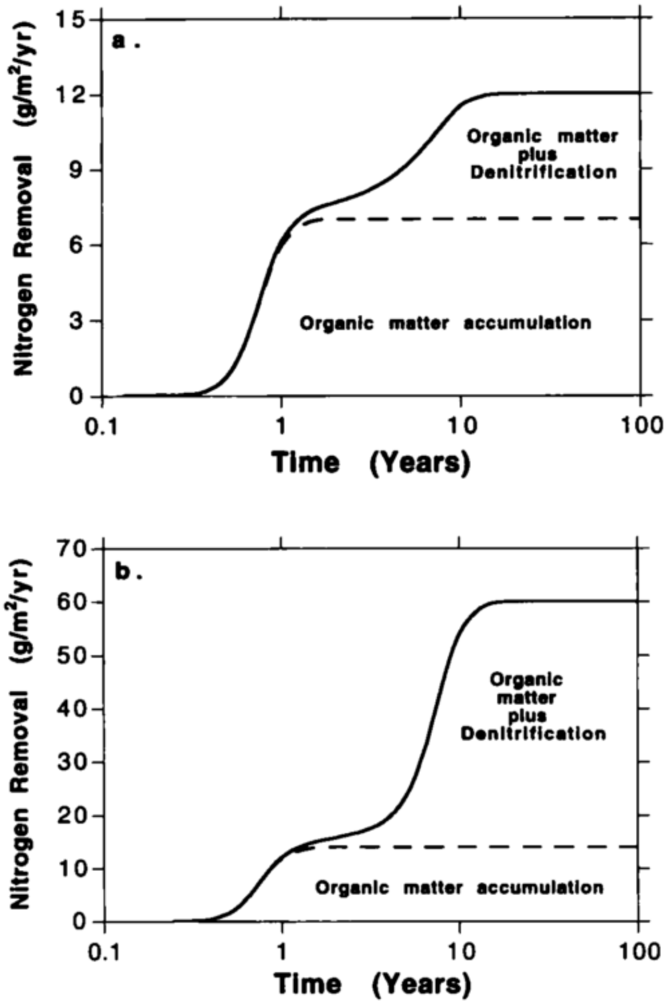

Figure 5. Trends in nitrogen removal over time in constructed wetlands under low (a) and high (b) nutrient loadings. These graphs demonstrate the relative roles of assimilation and denitrification based on available nitrogen in influent (taken from Craft (1997)).

Oxygen: Denitrification is a microbial process that needs organic carbon, nitrate, trace minerals, little to no oxygen and ideally, temperatures ranging from 20-25 ^{o}C (Crites et al., 1998; Reddy et al., 1994; Sutton et al., 1975). Wetlands are ideal settings for high denitrification rates because their defining hydraulic regimes cause persistent saturation, limiting oxygen transfer into the soil, and creating an environment ripe for denitrifying bacteria (Groffman et al., 1989). Dissolved oxygen levels have been correlated to percent soil saturation and redox potential, each fluctuating with water levels within a wetland. Sufficiently reduced soils conducive for denitrification occur at about 40% saturation and at redox potentials less than 300Eh (Hunter et al., 2008). Interestingly, in a successful wetland restoration that improved nitrogen levels in Sandy Creek, NC, C. J. Richardson et al. (2011) saw a trend in increased percent soil saturation, but no trends in groundwater dissolved oxygen levels. This research emphasizes the complexity of natural systems, especially ones that have been thoroughly disturbed in the restoration process.

Temperature: Denitrification rates are dependent on soil temperature. Cold temperatures also depress the mineralization of organic nitrogen and nitrification rates, decreasing the supply of nitrate for denitrification (Kadlec, 1999; Kadlec et al., 2009). Optimal denitrification in wetlands occurs between 20-25 ^{o}C (Crites et al., 1998; Reddy et al., 1994; Sutton et al., 1975), but has been shown to proceed at temperatures as low as 5 ^{o}C in laboratory tests (Brodrick et al., 1988). Natural and wastewater treatment wetlands have also been shown to continue to be effective in cold climates such as Minnesota (Kadlec et al., 2003), Montana (Moss, 2016), and Ohio (Spieles et al., 1998). Seasonal variations in wetland performance have been observed, showing higher denitrification rates during the summer than in cooler months (Phipps et al., 1994), and temperature has often been used in nitrogen removal models, but arguably its effects are minimal compared to seasonal variations in hydrology (Fink et al., 2004; Spieles et al., 1998).

Organic Carbon: The influence of available organic carbon on denitrification rates has been thoroughly researched. Hoffmann et al. (2000) correlated soil carbon content with denitrification rates, ranging from no denitrification in soils with less than 3% carbon content, and up to 8 g N/m^2/day in soils with 9% or more carbon content. Low soil carbon content has been blamed for relatively slow nitrogen removal rates in natural wetlands (Craft, 1997; Hoffmann et al., 2006; Spieles et al., 1998). In several cases with constructed wetlands receiving waste water, it has been shown that adding a carbon amendment to the wetlands, such as methanol or mulched plant biomass, can increase denitrification rates by 25-95% (Chen et al., 2014; Gersberg et al., 1983, 1984; Zhu et al., 1995). Wen et al. (2010) compared several carbon amendment schemes in constructed wetlands, and found a distinct positive relationship between available carbon and nitrate removal rates (R^2=0.93).

In natural systems, this same hydraulic scheme responsible for low dissolved oxygen concentrations is also responsible for storing carbon, so it is not typically in short supply: persistent soil saturation causes wetlands to be carbon sinks. Growth rates of the microbial community in the soil slow as the microbes are forced to use less energetic electron acceptors than oxygen, such as nitrate, iron, and sulfur (J. L. Richardson et al., 2000). With decomposition rates retarded (Craft et al., 1988; W. J. Mitsch et al., 2007), wetland soils are typically high in organic matter and carbon (Environmental Laboratory, 1987). Swamps and marshes average 72.3 kg/m^2 soil carbon content, with an average turnover rate of 520 years, compared to 18.9 kg/m^2 and 61 years in temperate grasslands (Raich et al., 1992).

This is not always the case in restored or created wetlands. Excavation for a restoration can literally remove tons of organic matter and carbon stores per hectare (Unghire et al., 2011), and created wetlands may be starting from a completely different ecosystem such as a grassland or forest, with soils that can have up to 80% less carbon (Raich et al., 1992). In a comparison of 12 created and 14 natural wetlands, Campbell et al. (2002) found that created wetlands had significantly less organic matter content than natural wetlands, and that there was no correlation between constructed wetland age and improvement in soil formation. Campbell et al. (2002) commented that this was likely due to the poor landscape position of created wetlands that did not promote persistent saturation. In contrast W. J. Mitsch, Zhang, et al. (2005) studied a created wetland in a natural floodplain that had managed hydraulics to ensure saturation. In this study they found that soil organic matter increased by 63% in the upper 8 cm of soil, and by 8.6% on average, 10 years after creating the wetland. Craft (1997) theorized, also through a comparison of natural mature wetlands to constructed wetlands of various ages, that denitrification may be limited by "available carbon during the early years of ecosystem succession." He estimated that after a restoration it may take 5-10 years to accumulate appreciable organic matter sufficient to support maximum denitrification.

<u>Nitrate:</u> In natural wetland systems that have not been disturbed, nitrogen removal is actually often limited by the availability of nitrogen itself (Ambus et al., 1991; Johnston, 1991). Nitrate's role is to provide denitrifying microbes with a source of energy, and without it, this particular bacterial community cannot exist and thrive (Maier et al., 2009). Several studies have shown that with an increase in nitrogen loading, the

microbial population also increases (Drury et al., 1998), along with denitrification rates, and wetlands are actually able to remove more nitrogen by mass (Ambus et al., 1991; Boustany et al., 1997; Craft, 1997; Drury et al., 1998). Johnston (1991) reported 300 times greater denitrification rates when comparing N-enriched wetland receiving 60 $gN/m^2/yr$ to wetlands only receiving 0.19 $gN/m^2/yr$.

As the water flows through a wetland, it is exposed to diverse niche environments ideal for supporting both nitrifying and denitrifying bacteria. Often in natural systems, parts or all of a wetland will go through varying levels of inundation due to flood or drought conditions. Riparian wetlands often get pulsed with storm water flooding. Even in a persistently saturated soil, wetland plant roots can convey oxygen belowground (Allen et al., 2002). The plants, and the varying water levels, contribute to varying levels of oxygen and carbon, turning on and off different processes, as is illustrated in Figure 6.

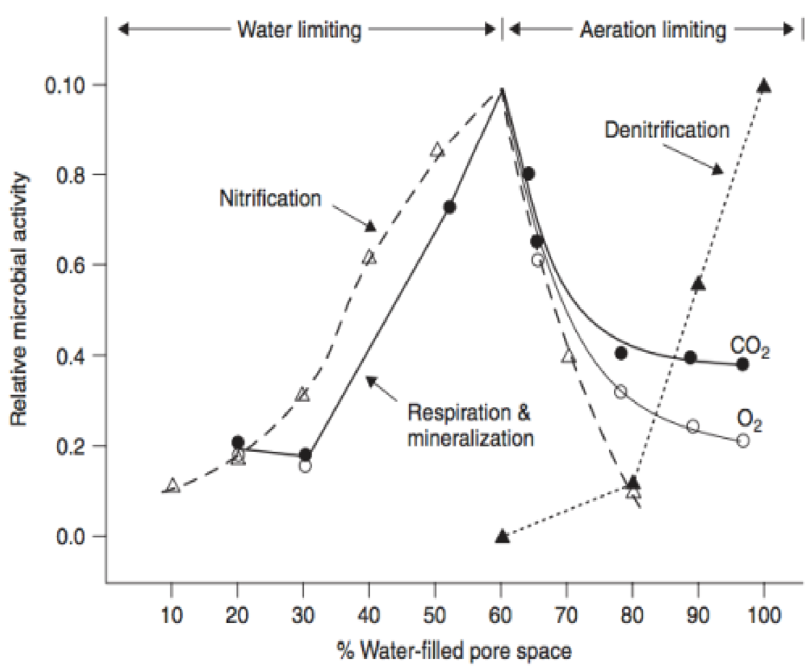

Figure 6. The relationship between water-filled pore space (a measure of soil moisture availability) and relative amount of microbial activity (re-drawn from Linn et al. (1984)).

Nitrogen Summary

Several factors have been discussed that affect nutrient uptake and denitrification in wetlands, and ultimately if leveraged correctly, wetlands can remove considerable amounts of nitrogen. To summarize, desirable features include: a robust plant community, plenty of carbon and nitrogen, niche environments that are both aerobic and anaerobic, surface water flow with slow velocities, and a hydraulic residence time that allows enough time for removal mechanisms to take place. All of these factors depend on hydrology to drive plant community selection, dissolved oxygen concentrations, and hydraulic schemes that dictate nutrient transport and flow through hydraulics.

For a functioning wetland, in general, as nutrient loading increases, so does the total mass of nitrogen removed, but removal efficiency decreases. W. J. Mitsch, Day, et al. (2005) compiled 47 years' worth of data collected from several wetlands in the Mississippi River Basin (Figure 7) to illustrate this phenomenon. In this compilation, the highest observed removal rate was 90 $gN/m^2/yr$, and that on average, natural wetlands can be expected to remove approximately 45% of influent nitrogen. Constructed wetlands that are engineered specifically to treat high strength wastewater, tend to be able to more effectively remove higher loads of nitrogen than natural wetlands. A $47.6m^2$ system designed in Montana, only 19 km from the Story Mill wetland, removes over 3 $g/m^2/day$ total nitrogen, even during the depths of winter with temperatures dropping to -20^oC (Moss, 2016). These wastewater treatment wetlands are highly engineered to manage hydraulics as well as oxygen, carbon, and nitrogen resources to maximize nitrogen removal. Natural wetland systems, instead, often rely on highly variable, environmentally driven flows and loading rates (David Moreno 2006).

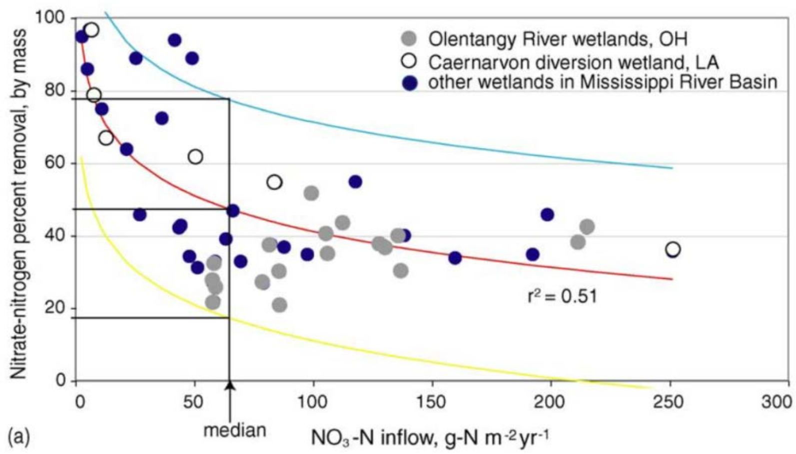

Figure 7. Removal of nitrogen by percent mass in created wetlands in the Mississippi River Basin. Each data point represents data for a complete year for a wetland. Figure taken from (taken from W. J. Mitsch, Day, et al. (2005)).

P removal

Forms and Sources

Phosphorus, like nitrogen, is an essential nutrient for life and can be harmful to an ecosystem at certain concentrations by causing eutrophication. Phosphorus nutrient cycling is simpler than the nitrogen cycle in that it has much fewer forms, sources, and pathways for removal. Phosphorus exists in water in two main forms, dissolved as orthophosphate, or as particulate, either sorbed to or precipitated as a solid, or associated with organic material (Kadlec et al., 2009). Only in very particular situations can phosphorus become gaseous, such as very intense wild fires (Gray et al., 2006). Although this is an important ecosystem process, it is not particularly relevant to a discussion of wetlands role in water quality. The discussion to follow will focus on dissolved and particulate phosphorus forms.

Phosphorus is a mineral found in many rocks, and is mined for fertilizers, detergents, and soaps. Phosphorus is also found in all living things, and is thus recycled through excretion and decomposition. Sources of phosphorus in surface and groundwater can be from local geology, agriculture, wastewater, and rain where concentrations very considerably. A study of a wetland receiving agricultural runoff in Denmark found that groundwater phosphorus concentrations were only 0.01 mg/L, whereas precipitation contained greater concentrations at 0.063 mg/L (Hoffmann et al., 2006). A study in Spain found higher phosphorus concentrations in agricultural runoff, ranging from 0.04 – 0.37 mg/L (Moreno et al., 2007). Phosphorus concentrations in runoff from poultry litter were recorded to be as high as 90 mg/L (Elliott et al., 2002), and phosphorus concentrations in waste water typically range from 4 – 15 mg/L (Bitton, 2005).

Mechanisms of Removal

There are four major pathways of phosphorus removal in wetlands: sedimentation, sorption, precipitation, and immobilization. Hydrology is also a common theme in phosphorus removal, driving system hydraulics and redox potential. Independent of hydrology, the soil environment pH is also very important.

Sorption and Precipitation: The mechanisms of sorption and precipitation are essential to long term storage of phosphorus in wetland soils (Craft, 1997). Sorption occurs when negatively charged phosphate is attracted to positively charged iron and aluminum oxides. Precipitation occurs when chemical binding of phosphorus primarily with iron, aluminum, and calcium ions (Sparks, 2003). Sorption depends heavily on the availability of sorption sites; once they are full, phosphate can no longer be removed this way. Soils high in clay content typically have high cation exchange capacities, and are effective at removing phosphorus from solution. Sandy soils on the other hand, have low cation exchange capacities, and tend to not sorb phosphorus well (Martin et al., 2013; Sanchez et al., 1980). The efficiency of phosphorus removal by sorption is also dependent on phosphorus concentrations in groundwater and surface water within a wetland. As phosphorus concentrations decrease, diffusion rates decrease, and it becomes less likely for phosphorus to come into contact with sorption sites, especially as more sorption sites become full (Kadlec et al., 2009). Typically, phosphorus removal rates are higher in young wetlands, but as sorption sites become full, removal rates taper quickly (Craft, 1997; Kadlec et al., 2009). A study was performed on a wetland in New Zealand receiving sewage effluent, where the inflow was also high in iron and aluminum. After 10 years, the phosphorus removal capacity remained high, and the authors theorized that with a continues supply of sorption sites, the high performance could be maintained indefinitely (J. G. Cooke, 1994).

Sorption reactions are also influenced by pH. The pH of the wetland environment dictates the strength of the positive charge of iron and aluminum oxides. As pH decreases, the concentration of positively charged hydrogen ions in solution increases, which encourages more hydrogen ions to be associated with the iron and aluminum oxides present. This causes the charge differential between phosphate and the oxides to increase, strengthening the attraction. As pH increases, these oxides become progressively more negatively charged, thus repelling phosphate, and returning the nutrient to solution. Phosphorus is most soluble, and therefore both plant available and susceptible to leaching, at pHs ranging from 6-6.5 (Pierzynski et al., 2005).

Phosphorus can also be removed by precipitating with calcium, another reaction dictated by pH. At higher pH ranges, calcium carbonate dissolves, releasing calcium into the soil solution, thus making it available to bind with phosphate (Pierzynski et al., 2005). In a study of a tidal wetland, it was observed that nearby subtidal oyster reefs supplied constant calcium inputs, resulting in a sustained phosphorus removal rates of 6-8 $gP/m^2/yr$ (Craft, 1997).

Dissolved oxygen concentrations in the wetland environment are instrumental in the permanence of phosphorus removal by sorption, as well as in precipitation reactions. When anaerobic conditions persist, iron oxides are reduced, releasing any sorbed phosphorus. At this point, iron and phosphate ions in solution are available to react and form precipitate, but this reaction is predicted to be slower than desorption, potentially allowing for the phosphorus to leach (Chacon et al., 2006; Dangelo et al., 1994). Craft (1997) concluded in his investigation of ecosystem succession in wetlands, that as created wetlands age and transition from oxidizing to reducing, phosphate becomes more mobile and susceptible to being exported. The work of C. J. Richardson et al. (2011) supports this claim, finding that exchangeable phosphorus increased when wetland hydrology was re-established in the floodplains of a watershed in North Carolina.

Sedimentation: Sedimentation can be a major pathway for particulate phosphate suspended in surface water to be removed. Sedimentation is largely a function of surface water velocity, as was discussed in the Hydraulics section, occurring when velocities are slow enough for solids to settle out of solution. The role of sedimentation in the removal of phosphorus also depends on the type of sediment entering the wetland: Kronvang et al. (2007) found that depositions rates were much higher in loamy catchments, transporting parent material high in phosphorus, versus sandy catchments depositing coarser sediment low in phosphorus. In a review of 20 inundated floodplains in the UK, Walling (1999) reported phosphorus deposition associated with sedimentation to be as high as 11.6 $gP/m^2/yr$.

Immobilization and Mineralization: Biotic immobilization and mineralization of phosphorus can play vital roles in whether a wetland leaches or sequesters phosphorus. Just as for organic nitrogen, mineralization rates and the recycling of organic phosphorus are slowed in anaerobic conditions. But unlike organic nitrogen, some organic forms of phosphorus are more recalcitrant, and phosphorus is more likely to build up over time (Dunne et al., 2005). Johnston compiled the work of several scientists and found that plant-uptake ranged from 0.12 gP/m2/yr to 5.33 gP/m2/yr, in different natural wetland systems. Hoffmann et al. (2006) measured that plants removed 11.9 mgP/m2/d during the growing season, in a wetland receiving agricultural runoff in Denmark. It has been found that much of this phosphorus is subsequently recycled, and that long-term retention of can only be expected to amount to approximately 1 $gP/m^2/yr$ (Craft, 1997; Johnston, 1991). Kadlec et al. (2009) summarized several studies of wetlands receiving wastewater, where phosphorus levels were substantially higher than those seen in natural systems. They concluded that as phosphorus loading rates increased, so too did plant uptake and phosphorus storage, from accretion rates of 0.73 g $P/m^2/yr$ in nutrient poor wetlands, to as high as 3.6 g $P/m^2/yr$ in nutrient rich wetlands. As with nitrogen, harvesting plant biomass is a practice used in constructed wetland systems for treating waste water (N. M. Wang et al., 2000).

Phosphorus Summary: A wide range of phosphorus removal rates have been observed, from net exports (Comin et al., 2001; Hoffmann et al., 2006) to the removal of 20 g $P/m^2/yr$ (Craft, 1997). The relative importance of each removal mechanism on water quality depends heavily on site characteristics. Sedimentation can play a large role if the influent water is high in particulate phosphorus, and hydraulics facilitate settling. Precipitation may be instrumental in neutral to basic calcareous soil, or in extremely reduced soils rich in iron. Sorption contributes most significantly in acidic soils that contain abundant iron and aluminum. Plant uptake and phosphorus accretion rates are greatest in anaerobic conditions where decomposition rates are slow, or when plant matter is harvested. For all mechanisms, removal by mass has been shown to improve with increased phosphorus loading.

Conclusion

Nitrogen and phosphorus cycling are vital to an ecosystem, but anthropogenic pressures have tipped the scales. No longer is it just the wild herds of elk, schools of salmon, and flocks of geese defecating in the woods, fields, rivers and lakes; it is huge confined animal feeding operations, and millions of acres of fertilized crops. It is septic systems that are outdated, under designed, and un-serviced, and it is green lawns, from the back yard to the golf course. It is the use of laundry and dish detergents, shampoos and soaps. With urbanization, wetlands become more crucial to our ecosystems. The changing landscape has decreased resilience while simultaneously increased pressure. With streams straightened, constricted, and deeply incised, unable to interact with their floodplains, and wetlands drained and filled, the increased nutrient and sediment load is compounded down the length of rivers.

Much has been done to improve water quality since the clean water act was passed in 1972. Water quality objectives have moved from qualitative to quantitative, and the EPA, instead of the FDA, now regulates agricultural waste management (United States Environmental Protection Agency, 2016a). Awareness of the importance of wetlands has increased significantly, with grants available for restoration and protection. With the help of programs like the Wetlands Reserve program over 11,000 private land owners have preserved 2.3 million acres of wetlands in the US (Service). Despite these strides, total wetland area continues to decline (Dahl, 2009). Careful wetland design, monitoring, and re-evaluation needs to persist to help evolve how waterways are managed to keep them clean and ensure efficient use of funding and effort.

MATERIALS AND METHODS

The Story Mill site is a complex wetland at a location that has endured anthropogenic alterations over time during the homesteading process that converted the area from a natural setting to an agricultural operation (US Department of the Interior National Park Service, 1996). The site has been recently remediated with the goal of reestablishing the historic wetland function. To understand the extent to which the restored wetland shows traits of a natural treatment system, field observations were undertaken from summer 2014 to fall 2015. Direct access to the surface water in streams that border much of the site and to shallow groundwater at spatially distributed locations throughout the site provided information used to develop estimates of water movement and implications concerning water quality. Materials and methods are described here to first provide an overview of the restoration and a site description. This section will also document the groundwater and surface water field observations, laboratory analyses of water sampled from the site, and the resulting data analysis and modeling concerning both water quality and quantity.

For the ease of discussion, the property is divided into three main sections, the Northern, Southern, and Triangle parcels. Monitoring was focused on the Southern Parcel, but effects on surface water quality from the restoration as a whole were monitored in a downstream location.

Site Description

The Story Mill wetland is located in the Gallatin Valley, Montana, which is part of the Middle Rockies Ecoregion III (Woods 2002), characteristic of high alpine mountains, with glacial lakes and high gradient streams and rivers (Wiken et al., 2011). Bozeman sits at an elevation of approximately 1,500 m, in a large valley between mountain ranges. This area is known for its long cold winters, with only a few brief, warm and beautiful summer months, and receives an average of 50 cm of precipitation annually (National Resources Conservation Service, 2016), mostly in the form of snow. During this study, from August 1st 2014, to August 31st 2015, average daily air temperatures ranged from -27 ^{o}C to 26 ^{o}C, with 2,371 growing degree days above a base of 5 ^{o}C (Shaw, Joseph).

The Gallatin Valley is also known for its soil fertility: deep, loess-rich soils sit on top of a complex of gravelly alluvial fans that flow north from the Gallatin Range. The underlying parent material consists of Archaean gneisses, Flathead sandstones, weathered remnants of Wolsey shale, Meagher limestones, and Absaroka volcanics—all rock types comprising the Gallatin Range (Hackett et al., 1960). The uppermost parent material is loess, sourced either locally from the Gallatin Valley’s many braided rivers or more distally from other river systems (Bourne et al., 1962; Hackett et al., 1960). An example of a typical soil profile is shown in Figure 8, where the bank of the East Gallatin River at the site has exposed the soil layers. Here it can be seen that the A horizon is rich in plant roots and dark from high organic content, and that with depth, silty clay transitions first to sandy clay and then to alluvial gravel and cobbles. A typical Story Mill wetland soil profile, as mapped by the NRCS, consists dominantly of three soil series: Enbar-Nythar loams, Blossberg loam, and the Brandy-Riverwash-Bonebasin complex. All of these soils are classified at the highest taxonomic level as Mollisols, which are prairie-like soils, rich in organic matter (A horizons contain 2-5% OM). At the second highest taxonomic level, nearly all of these soil are Aquolls, Mollisols formed under an aquic (very wet) soil moisture regime; the exception is the Enbar series, an Ustoll formed under drier ustic soil moisture regime more typical of upland soils periodically wetted by rain events instead of persistently saturated conditions. Textures range from fine-loamy to sandy and sandy-skeletal (California Soil Resource Lab).

Figure 8. Exposed soil profiles on opposing banks of the East Gallatin River, displaying the variety of depth in the loess deposits on top of alluvial gravel and cobbles. The profile on the left is approximately two feet deep, while the profile on the right is approximately five feet deep.

Dr. Stephanie Ewing's ENSC454 Landscape Pedology MSU classes (Ewing et al., 2013-2015) have performed the most rigorous soil characterization efforts of Story Mill wetland soils to date. They have tentatively classified all but one of six soil profiles as Fluvaquentic Endoaquolls, with the other profile classified as an Aeric Calciaquolls. These soils in general, showed similar morphologic properties to those mapped by the NRCS. Several soil profiles had distinct redoximorphic features such as gleying, reduced iron, and the smell of hydrogen sulfide (rotten egg smell) (Figure 9). Soil organic carbon values were 4-20% for surface horizons and 1-4% for subsurface (mineral) horizons.

Figure 9. An example of the redoximorphic features found in the soils at Story Mill, during a site visit by Stephanie Ewing's Landscape Pedology class.

Vegetation at Story Mill consists of a diverse variety of over 100 different trees, shrubs, forbs, and gasses. The most quantitative vegetation survey for the Story Mill property was performed by A. Pipp in July 2013 (RESPEC Consulting and Services, 2013), and is summarized here. A stand of aspen (*Populus temuloides*) and cottonwood trees (*Populus balsamifera*) flanks the east portion of the site, all along the East Gallatin River, and dense willows (*Salix fragilis, Salix boothii,* and *Salix exigua*) border Bozeman Creek and the southern property boundary. Most shrubs are noxious non-natives such as Canadian thistle (*Cirsium arvense*), common tansy (*Tanacetum vulgare*), and dandelion (*Taraxacum officinale*). The majority of the site is covered in grasses, a mixture of native and non-native, mostly water-tolerant species ranging from facultative to obligate wetland plants. In more upland areas, dominant species include non-native quack grass (*Agropyron repens syn. Elymus repens*), red top (*Agrostis stolonifera*), and smooth brome (*Bromus inermis*), and native field horsetail (*Equisetum arvense*) and common timothy (*Phleum pretense*). Pre-restoration delineated wetland areas were dominated by native obligate wetland plant species, including a variety of sedges (*Carex microptera, Carex pellita*, and *Carex praegracilis)* and rushes (*Eleocharis palustris*, *Equisetum hyemale*), as well as reed canary grass (*Pharlaris arundinacea*) and cat tail (*Typha latifolia*).

Figure 10. The Story Mill wetland in the fall 2015, looking north-west from the eastern property boundary.

The Restoration

The major earth work and construction for the restoration took place in several stages in 2014 and 2015 and was intended to reconnect the hydrology of the rivers to the wetland, and increase the area and function of the wetland (RESPEC Consulting and Services, 2013). The majority of construction took place September through November in 2014, which focused on the creeks, and Southern and Triangle parcels (Figure 1). Restoration of the Northern Parcel was done in September 2015. Major restoration activities included creek restoration, floodplain excavation, filling a ditch presumably used to dewater the site, re-contouring the pond, and removing several buildings. A conceptual design map of the restoration, provided by RESPEC, is shown in Figure 11 and includes previously delineated wetland area, the anticipated area expansion, as well as many other park features. A map diagraming the actual cut and fill from the construction in the Southern parcel is also included below (Figure 12).

Several areas in the floodplain were excavated to be accessible by a 2 year flood with the hope of increasing connectivity between the creeks and the wetland. The feature in the south-west corner of the site is called the Bozeman Creek Backwater Slough. The inlet and outlet to the Slough are coincident: floodwaters are designed to flow into the Slough as a large eddy, and as water levels recede, flow goes back out to the creek using the same entrance. A similar feature was created in the Northern Parcel, but with several access points instead of just one, potentially allowing for flow-through hydraulics. In the Triangle Parcel, the East Gallatin’s severely incised banks were lowered and dug back approximately 30 m on either side, completely reconnecting about 70 m of river to its floodplain. Collectively, approximately 11,500 m^3 of fill material was removed from the Story Mill property floodplains.

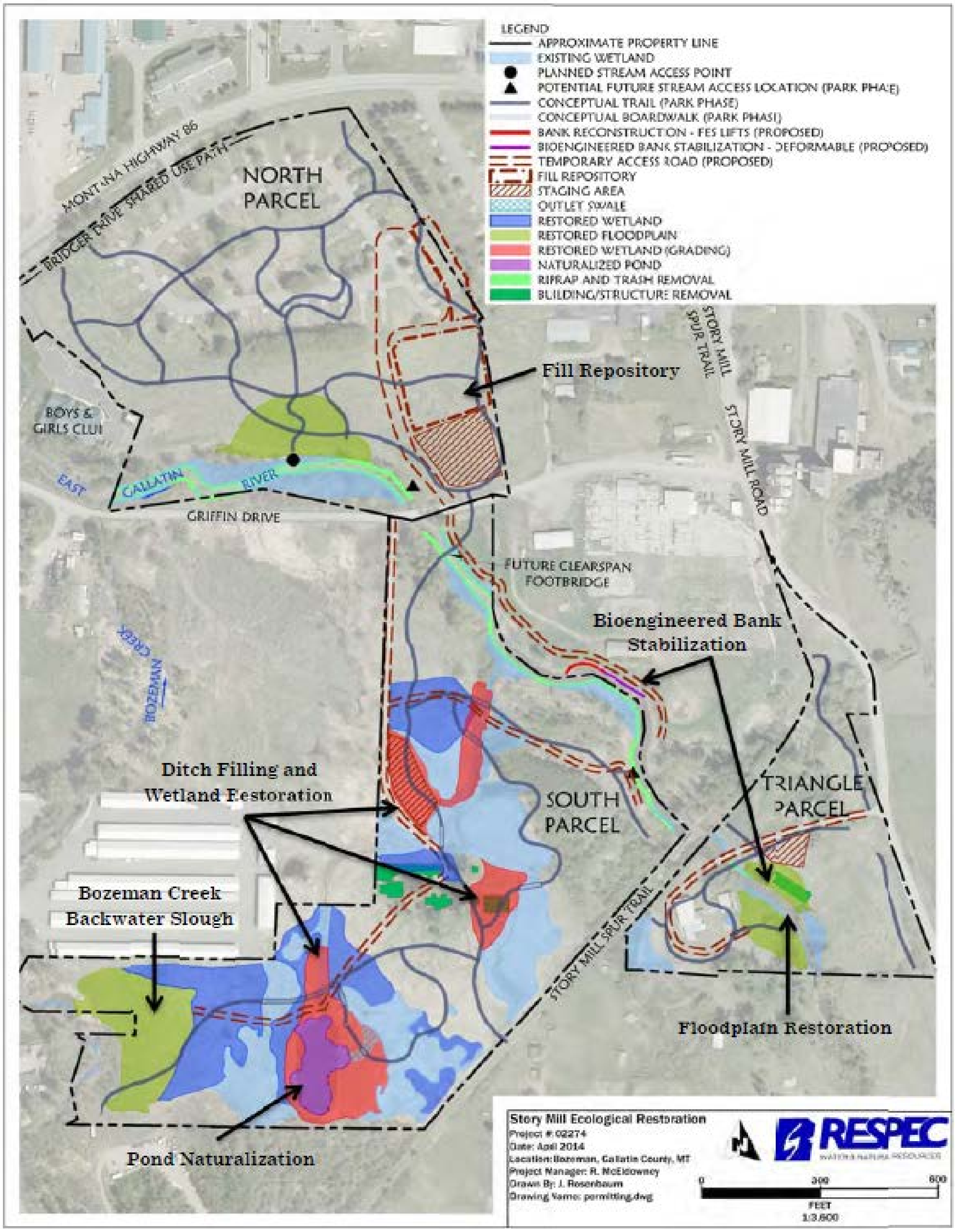

Figure 11. An aerial view of the Story Mill wetland, indicating areas delineated as wetland before the construction, and areas intended to be restored to wetland. Areas of floodplain excavation, and recontouring are also shown, along with plans for walking trails (taken from (RESPEC Consulting and Services, 2015)).

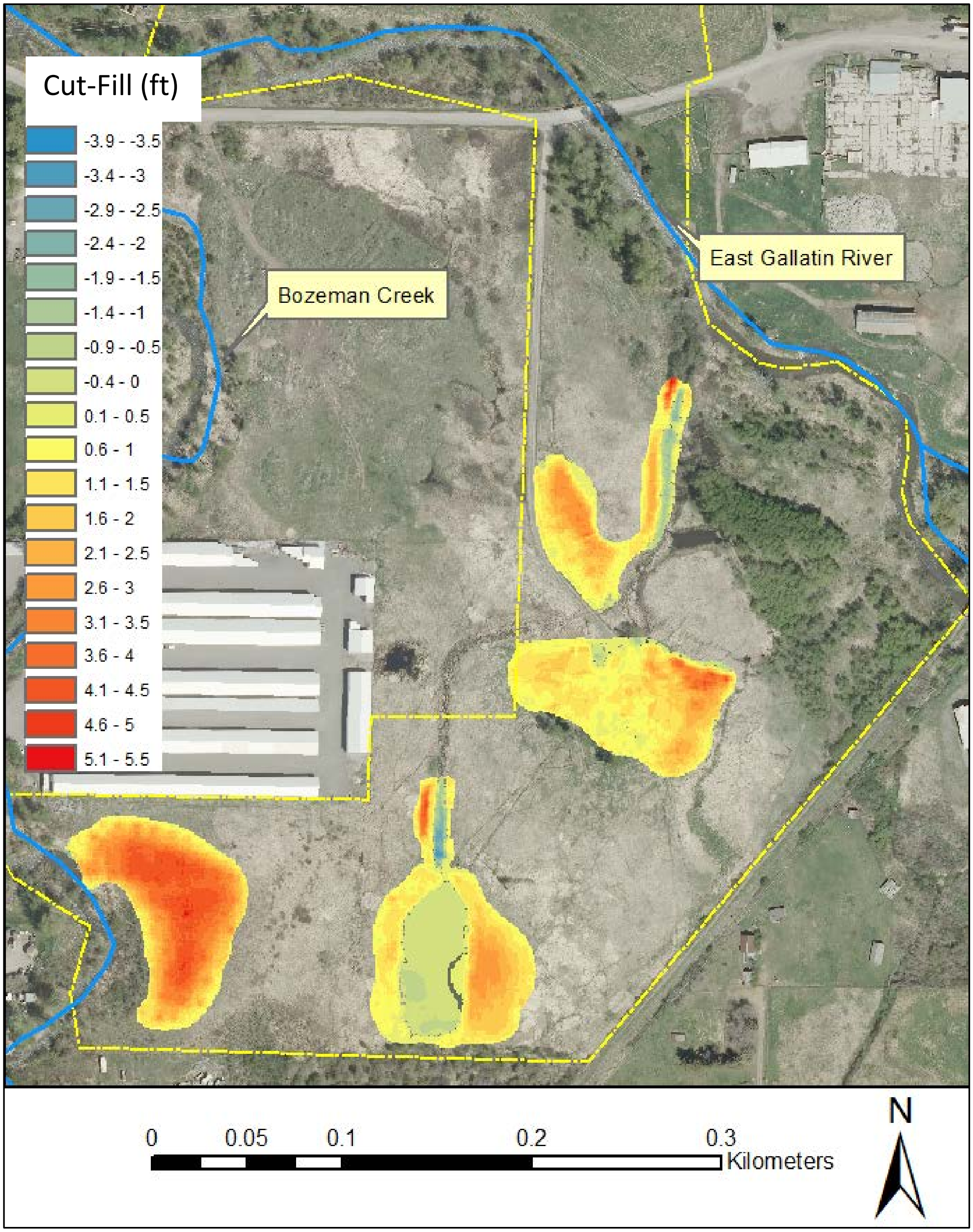

Figure 12. An aerial view of the Southern Parcel of the Story Mill wetland, indicating as-built excavation and fill around the pond, ditch, old structures, and in the floodplain. Yellow and red indicate cut, while green and blue indicate fill.

Several other re-contouring projects were executed in the Southern Parcel. The pond was reshaped, and its banks were removed, intending to encourage the pond to regularly spill out into the field to the east. A few buildings were removed from the middle of the site, and the hill they were built on was regraded. The potential drainage ditch was filled, aiming to reverse historic dewatering efforts, and slow the water moving through the site. The anticipated response was for the groundwater elevations to rise, thus increasing wetland volume, and contributing to an increase in wetland area. Topsoil was saved, and re-applied as best as was feasible, and native plants were planted in all disturbed areas.

Stream restoration efforts included dredging 6,800 kg of rip-rap from the riverbed and banks, hoping to allow for more natural hydraulics to encourage energy dissipation and prevent further incising (RESPEC Consulting and Services, 2013). Efforts to promote less confined flow patterns were limited because of property boundary constraints. A section of the East Gallatin River actually had to be re-stabilized to ensure that any changes caused in the creeks flow patterns due to the restoration did not undercut neighboring property.

Photographs of the restoration are shown below in Figures 13-18. Examples of historical compared to design cross-sections are also provided in Figures 19, 20, and 21.

Figure 13. Aerial view of the Bozeman Creek Slough during construction in September 2014 (taken from RESPEC Consulting and Services (2015)).

Figure 14. Floodplain excavation along the East Gallatin River, similar to the Bozeman Creek Slough, constructed on the Northern Parcel in September 2015.

Figure 15. Floodplain excavation along the East Gallatin River on the Triangle Parcel, completed in November 2014. Tall river banks were dug down and back to create a more connected riparian zone (taken from RESPEC Consulting and Services (2015)).

Figure 16. Recontouring the Pond. Top Left: Original pond construction with a sluice gate to drain into the ditch. The pond was contained by tall berms of soil. Top right: Pond one year after construction. Bottom: Pond directly after construction, with lowered bank and no sluice gate (taken from RESPEC Consulting and Services (2015)).

Figure 17. Before and after photographs of the ditch draining the pond in the Southern Parcel.

Figure 18. Before (left) and after (right) pictures of the East Gallatin River. Several tons of concrete and metal were removed from the river bed and banks. In the top series, where the river flows past private property, more natural bank stabilization was employed to ensure any resulting creek hydraulics did not undercut the bank (taken from RESPEC Consulting and Services (2015)).

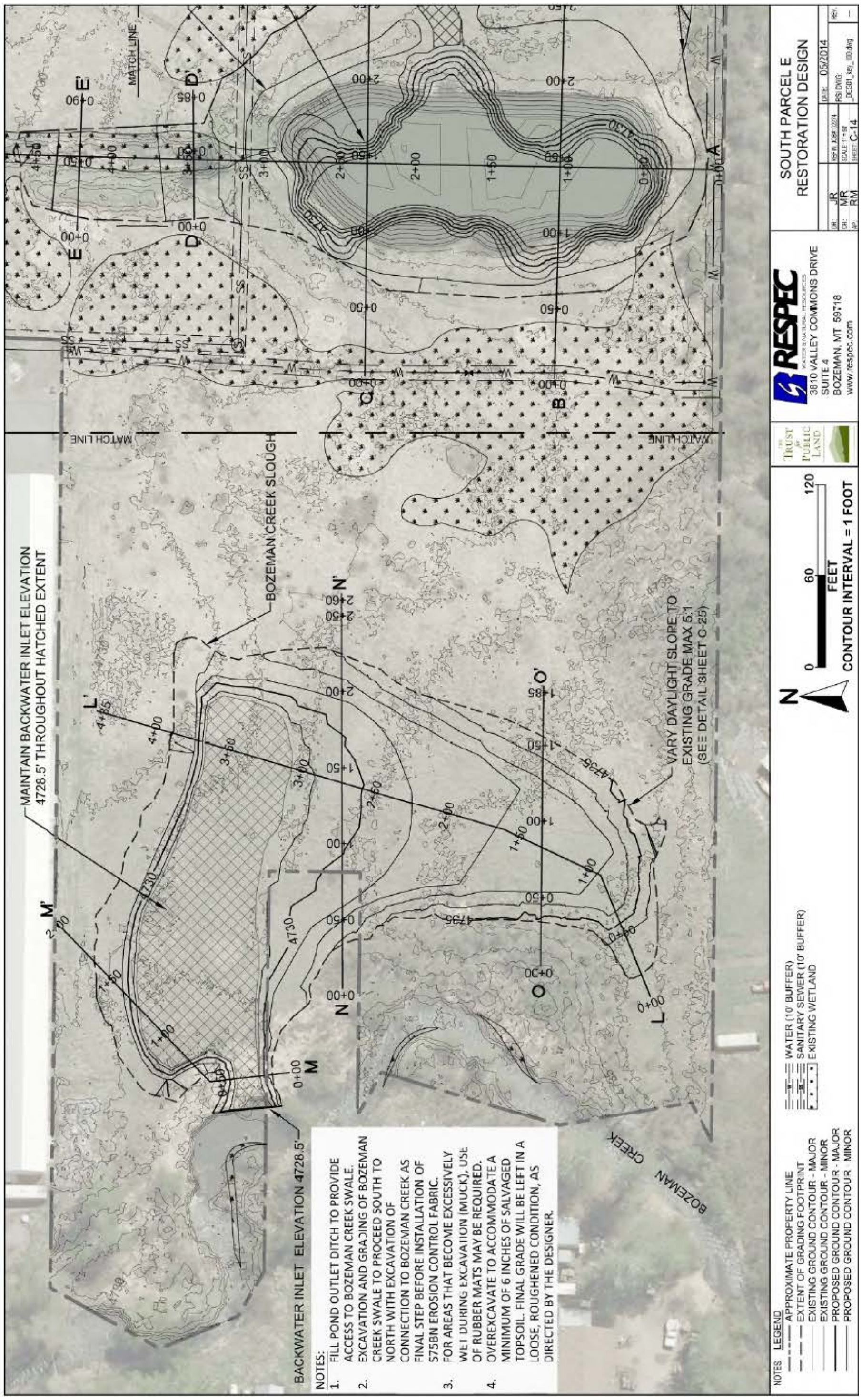

Figure 19. Design drawings for the construction of the Bozeman Creek Backwater Slough and recontouring the pond and ditch (taken from RESPEC Consulting and Services (2014)). *Figure displayed in English units, conversion factor from ft to m = 0.305

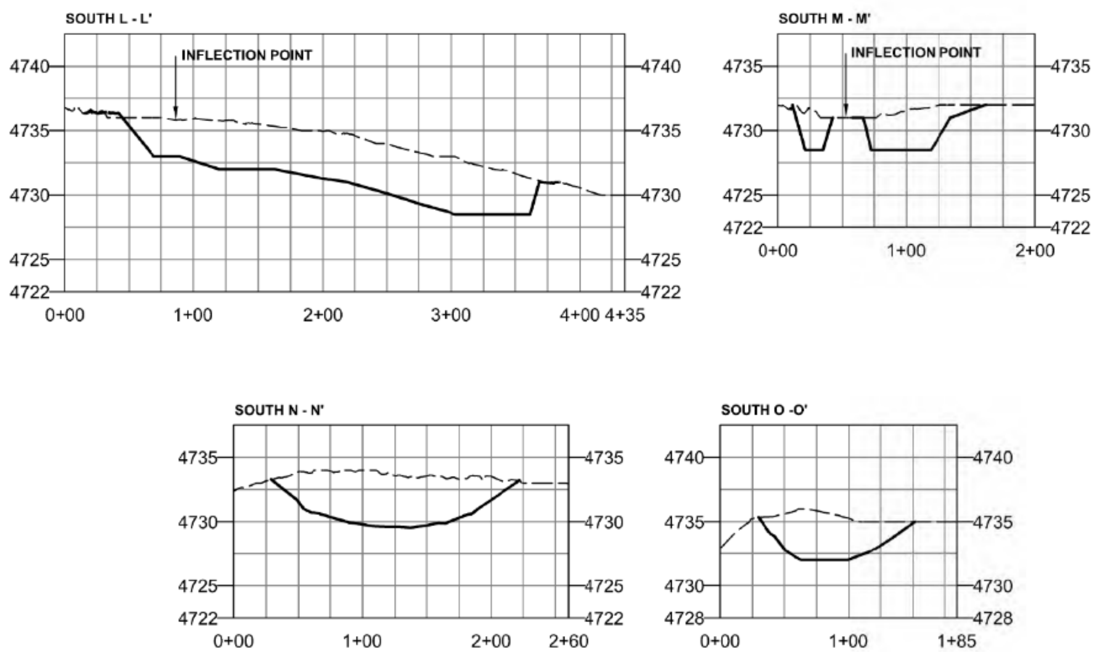

Figure 20. Crossections of the original floodplain and the restored Bozeman Creek Backwater Slough (taken from RESPEC Consulting and Services (2014)). *Figure displayed in English units, conversion factor from ft to m = 0.305.

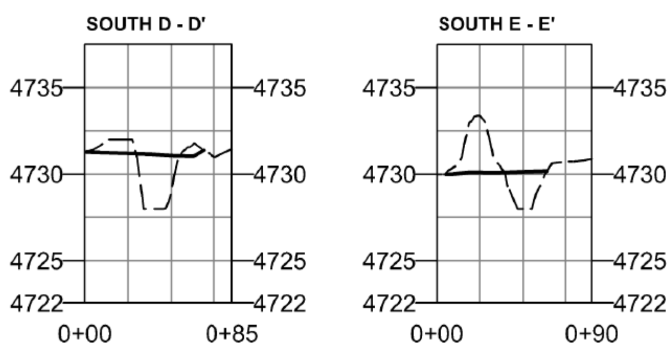

Figure 21. Crossection of the original and restored drainage ditch exiting the pond (taken from RESPEC Consulting and Services (2014)). *Figure displayed in English units, conversion factor from ft to m = 0.305.

Field Observations

Groundwater

Thirty-four groundwater wells (Figure 22) were used for data collection in the Southern Parcel. Wells were spatially arranged with the goal of creating a general picture of site-wide processes. Several constraints such as preexisting infrastructure, construction activities, and land ownership boundaries all ultimately influenced well placement. Preexisting well infrastructure was installed by two different entities over time: nine were installed in 2006 by Hyalite Engineers for a previous site investigation (Ravnaas, 2007), and fifteen were installed by TPL in 2013 as an initial effort to understand the site hydrology for the restoration (RESPEC Consulting and Services, 2013). To gain higher spatial resolution MSU installed ten additional wells in 2014, some of which replaced dry, ineffective wells.

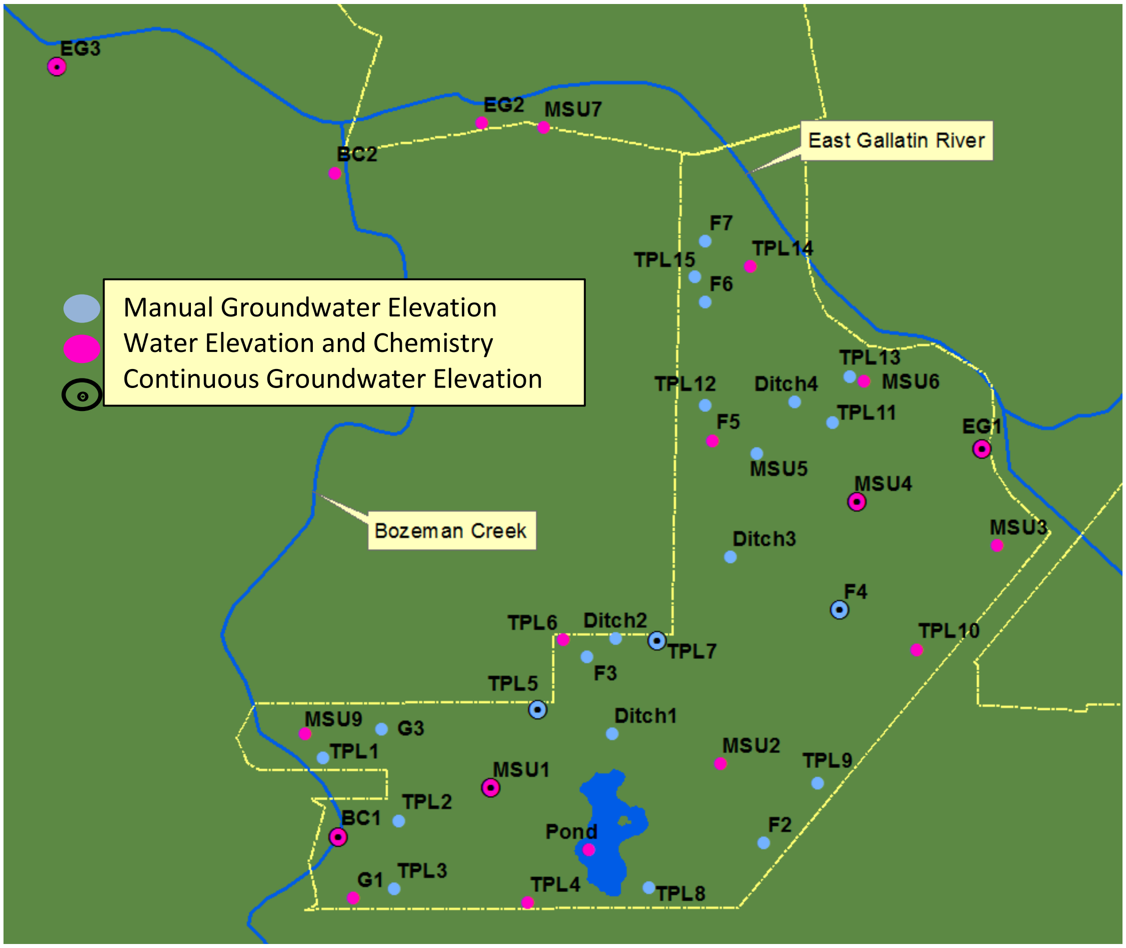

Figure 22. Monitoring map, including groundwater and surface water locations. Water level data was collected for all locations, while only a select subsets were used to collect water quality and continuous water level data.

While all wells were of functionally similar construction, the MSU wells added in 2014 consist of 5.1cm diameter PVC inserted to a maximum depth of 2.1m using a 7.6cm diameter hand auger. Installation was in accordance with USGS guidelines for groundwater wells intended for water quality studies (Lapham et al., 1997), differing only in that concrete caps were not installed around the base of the wells. Well pipes were slotted for the lower 0.6 m, and solid for the remaining length up to, and above the ground surface. Well casings were packed first with silica sand around the slotted portion of pipe, and then with bentonite around the solid portion of pipe. This design is intended to allow for groundwater to freely flow into the well, while preventing clogging and surface water contamination. Well locations and elevations were surveyed with Trimble GPS equipment to the north edge of each well at ground level. All spatial locations were referenced to the North American Datum of 1983 and are accurate to 2 cm both horizontally and vertically.

During the restoration construction, some groundwater wells were removed. Wells were replaced that were considered valuable to capturing potential hydrologic changes and necessary for adequate spatial resolution. A schedule of well owners, installations, removals, and replacements is provided in Appendix A.

Groundwater levels in the observation wells were measured manually with an electronic tape measure. Measurements were taken weekly from August 20th, 2014 to August 11th, 2015, with only a few interruptions due to freezing temperatures or absence of groundwater at the maximum well depth. When frozen or dry wells were encountered, no observations of groundwater depth were made. Both the length of pipe above ground, and the depth to groundwater from the top of the well casing were always measured, to account for the possibilty that the pipe casing had shifted in the ground (field sheet provided in Appendix B). All measurements were taken at the north edge of each well, corresponding to the surveyed location. Total groundwater level error, including instrument error and error based on reproducing the datum due to ground unevenness from tall grasses, snow, and mud was estimated to be 3 cm.

The Greater Gallatin Watershed Council (GGWC) also collected groundwater level data in the summers of 2013 – 2015, every other week from May – September. Data collected by the GGWC was only from the 15 TPL wells. On several occasions in the summers of 2014 and 2015, MSU and GGWC coordinated to visit the field and collect data together. Unique data collected by GGWC prior to MSUs monitoring efforts were included in the analysis.

Groundwater well recharge was measured in 12 wells in the summer of 2014 by performing slug tests (Hvorslev, 1951). Slug tests were performed once in each well, and wells were chosen to be spatially distributed across the site. Each well was pumped dry, and as the well refilled, depth to groundwater measurements were taken with respect to time, using an electronic tape, until water levels reached the original height. All well and casing geometry were also recorded.

Groundwater samples for water quality analysis were typically collected from 12 wells once a week, from August 19th 2014, to August 6th 2015. Sampling wells were chosen to be spatially distributed across the site, with the intention of capturing general site-wide processes (Figure 22). Samples were collected after wells were purged with a peristaltic pump and allowed to recharge, to achieve consistently representative samples of near-well groundwater. The purge volume was measured to be approximately three times the static water volume in each well, and purge water was directed away from the well head to ensure it did not drain back into the well. It was observed that in wells that were nearly dry, recovered water was turbid, likely due to disturbance of sediment settling into the bottom of the well. Samples were not collected from wells with low water levels where relatively clear groundwater could not be produced. Two 50 mL centrifuge tubes were rinsed three times with water from each well, then filled, and were promptly put in a cooler on ice. Samples were transported to the lab where they were frozen within 8 h of collection for subsequent analysis (sample field sheet provided in Appendix C).

Dissolved oxygen was measured in situ once per week in conjunction with collecting lab-destination samples. A Hach LDO Probe (IntelliCALTM LDO101) was used, which has a range of 0.1 to 20.0 mg/L with an accuracy of 0.1 mg/L from 0-8 mg/L and 0.2 mg/L from 8-20 mg/L. Initial measurements immediately after recharge were considered to be most representative of the surrounding groundwater because the water had little time to interact with the atmosphere. Measurements were taken from the middle of the water column, after they stabilized, determined when consecutive measurements were within 0.2 mg/L of each other (sampling field sheet provided in Appendix C).

The Trust for Public Land also contracted Torie Haraldson to collect samples and measure dissolved oxygen and pH on two dates prior to construction: November 11th 2013 and May 19th, 2014. Water chemistry parameters were collected from a subset of eight TPL wells, in accordance with the Story Mill Pre-Restoration Nutrient Monitoring Project Sampling and Analysis Plan (Haraldson, 2013).

Figure 23. Measuring dissolved oxygen and collecting samples from a groundwater well in the winter of 2015.

Surface Water

Surface water gauging stations were established in 2014 in the East Gallatin River, Bozeman Creek, the pond, and four locations in the ditch (Figure 22). Staff gauges in the streams were installed at the upstream and downstream extents of the property, with an additional location in the East Gallatin River, downstream of the confluence with Bozeman Creek. Locations were chosen to represent influent and effluent water properties entering and leaving the site. In order to minimize potential debris snags, ensure ease of access, and data accuracy, careful attention was paid to finding areas of calm and protected flow. Gauge locations and elevations were also surveyed with Trimble GPS equipment, referenced to the North American Datum of 1983, and are accurate to 2 cm in all dimensions (field sheet provided in Appendix B).

Surface water gauging stations were subject to disturbance by site construction activities as well as less predictable natural forces. The pond gauging station was replaced twice, once after it was removed during construction, and again after being dislodged by shifting ice in the winter. Ditch1 and Ditch4 were both removed when the ditch was filled in, and were not replaced. The gauging station labeled EG2 became useless for flow rate measurements after beavers made a dam directly downstream. Once the East Gallatin River ponded behind the dam, ice became a problem, and shifting ice took the EG2 gauging station out altogether. The staff gauge at BC2 was dislodged by a snag carried in the creek's flow, and was quickly replaced. A schedule of well owners, installations, removals, and replacements is provided in Appendix A.

Surface water elevations were measured manually from the gauging stations once a week on the same schedule as groundwater observations. Similarly, freezing water and equipment were constraints for surface water level data collection. Despite deliberate placement, all locations in the streams were in varying degrees of current, where the water surface fluctuated rapidly, or pillowed slightly on the staff gage, resulting in an estimated error of 3.4 cm.

Volumetric flow rates in Bozeman Creek and the East Gallatin River were estimated at locations BC1, BC2, EG1, EG2, and EG3 using the USGS method (Rantz, 1982). A Gurley velocimeter was used to measure velocity for at least 10 points for any given cross section, ensuring increments did not include more than 10% of the flow. Velocity measurements were taken at 60% of the depth (measured from the water surface downward) for any given point. The velocity measurements were repeated at several different stages to relate volumetric flow rate with water surface elevation over a range of the hydrograph. Once rating curves were developed, flow rates were calculated from staff gauge height. A guideline used by practicing engineers and hydrologist is to assume measured flowrate has approximately 10% error, while flow rate calculated from a staff gauge measurement and a rating curve can have approximately 20% error.

Figure 24. Measuring crossectional area and velocity of Bozeman Creek at the BC1 gauging station.

Grab samples of surface water were collected weekly from locations in the streams adjacent to each gauging station, on the same schedule as groundwater sampling. All samples were taken from roughly the middle of the water column. Special care was taken to ensure stream samples were from areas with downstream current, and not from stagnant water or eddy pools. Just as with groundwater samples, two 50 mL falcon tubes were rinsed three times with water from each site, filled, placed in a cooler on ice, transported to the lab, and frozen within 8hrs of collection (sampling field sheet provided in Appendix C).

Dissolved oxygen was measured in-situ in the streams by laying the HACH probe along the streambed, and in the pond by holding the probe in the center of the water column. Measurements were taken until they stabilized, using the same criteria as for groundwater (sampling field sheet provided in Appendix C).

Continuous Stream and Groundwater Well Observations

In addition to manual measurements, data loggers (HOBO Water Level (13ft) Data Logger U20L-04) recorded water level on five-minute intervals at 8 locations. Data collected in 2014 were from wells distributed around the site, whereas data collected in 2015 were from wells in an east-west transect across the site between BC1 and EG1 (Figure 22). Loggers remained in BC1, EG1, and EG3 throughout November 2015, except when all data loggers were removed for part of the winter to avoid potential freezing. HOBO data loggers are rated to have a 0.1% accuracy from 0-4 m, or a maximum error of 0.4 cm. A data logger was also used to measure atmospheric pressure on site, and used to adjust water level readings. Water and air temperature were also recorded, but are not included in this report.

Weather Data

Precipitation and air temperature data were also recorded during spring of 2013 through the summer of 2015. Data was collected from the MSU weather station, located on top of Cobleigh Hall, 3.8 km from the Story Mill wetland. Continuous precipitation and temperature data was distilled to daily totals and averages, respectively.

Laboratory Analysis

All surface and groundwater samples were transported to the lab on ice, pH tested, and frozen for future analysis within 8 hours of collection. pH was measured using an IntelliCAL™ PHC281 pH Ultra probe. For subsequent analysis, samples were thawed and divided for either ion chromatography or for analysis of total nitrogen (TN) and total phosphorus (TP) by digestion techniques. MSU does not have an EQUIS certified lab, and some of the analytic methods used differ from those specified by the DEQ Standard Methods. Results by the methods used were validated by splitting a set of samples (twelve samples) for analysis by both MSU and Energy Labs in Helena in the winter of 2014. Results correlated strongly for nitrate, chloride, and sulfate, with $R^2 > 0.98$, and negligible y-intercepts with respect to each constituent's detection limits and range. TP, orthophosphate, and nitrite measured below MSU's detection in all split samples, and correlations were not determined for these parameters. TN correlated well, with $R^2 = 0.83$, but had a significant y-intercept of 0.63 mg/L, where Energy Labs consistently measured higher TN levels in the split samples (Appendix D).

Undigested Water Samples

Undigested samples were filtered through a 0.2 µm sieve and analyzed at MSU in accordance with EPA method 300.0 (Pfaff, 1993). In brief, this method uses ion chromatography to analyze a water sample for a panel of five ions simultaneously:

nitrate, nitrite, orthophosphate, chloride, and sulfate. For the sake of this research, only nitrate, orthophosphate, and sulfate results will be reported on. Anions were measured with a METROSEP A Supp 7-250, 250 mmL X 4.0 mmID column. Detection limits and error were determined in accordance with the method, and with the following additional considerations. Detection limits were augmented by looking closely at the lower extent of the results from the check standards, and were considered to be where the equation for the standard curve no longer clearly described samples of known concentration, corresponding to where prediction error dramatically increased. Error was based on the strength of the equations to predict a range of standard concentrations: replicate standard samples were measured at various concentrations, a standard deviation was then calculated for each set of replicates and over the complete range of standards, the 67th percentile standard deviation was calculated and used to describe error for each constituent. Error was also based on a comparison of split samples analyzed by both MSU and Energy labs for quality control. A table of determined errors and detection limits are shown in Table 4. Notice that the detection limit and error for sulfate is much higher than for the other ions. This is because the standard curve for sulfate incorporated a wide range of concentrations, from 0-50 mg/L, whereas all other ion concentrations were only expected to range from 0-5 mg/L and therefore had more precise standard curves.

<u>Digested Water Samples</u>

Samples were analyzed for TN and TP simultaneously by performing a bench top persulfate digestion on unfiltered samples. Nutrients were then measured colorimetrically on a Lachat in MSU's Environmental Analytical Lab. QuickChem Method 10-107-04-4-B was followed, where all forms of nitrogen and phosphorus are oxidized and converted to nitrate and orthophosphate, respectively. Standard curves were developed using glycine (nitrogen) and glycerophosphate (phosphorus), as specified by the USGS Water-Resources Investigations Report 03-4174. These compounds represent the most complex organics potentially found in the collected samples, and indicate how completely the digestion procedure oxidizes tightly bound nutrients. The detection limit for TN and TP were found to be 0.05 and 0.04 mg/L respectively.

Benchtop digestion is complex, and several challenges were encountered, requiring the method to be adjusted slightly. Samples boiling and losing volume was an issue. The method suggested vials with rubber gaskets to create a strong seal and prevent vapor loss. Several cap types were tested, including ones with rubber seals, and volume loss was still a consistent problem. 15mL Screw Cap Digestion Vessels, made by Environmental Express, performed the best, and were ultimately used for standard curve and sample analysis. A further recommendation would be to try screw caps with a flexible center to allow for expansion without loss (crimped caps with flexible centers were tried, but were blown off in the autoclave)(Brookshire, 2015). To further address sample volume loss, samples were autoclaved at 120^{o}C, instead of following the specified method of using a block digester at 150^{o}C. Samples were autoclaved for 30 minutes with a slow exhaust for both the nitrogen and phosphorus digestion steps. This ensured a pressurized environment until samples had cooled enough to prevent steam loss. From experimentations, it was also found that autoclaving resulted in more thorough digestion of standards, perhaps due to better thermal contact. If any losses were observed, vials were filled back to volume with DI water once all digestion steps were complete.

Another problem was nitrogen contamination in the persulfate digestion reagents. Potassium persulfate is a strong oxidizer, and will scavenge gaseous ammonia, converting it to nitrate. A low nitrogen potassium persulfate must be used, and all reagents must be stored in completely sealed containers, preferably not in a fume hood.

Parameter	Detection Limits (mg/L as element)	Error (mg/L as element)
NO3	0.05	0.02
PO4	0.1	0.05
SO4	0.5	1.48
TN	0.06	0.05
TP	0.05	0.01

Table 4. Detection limits and error for each compound measured in water samples. Values are reported as mg/L as the element of interest, either nitrogen, phosphorus, or sulfur.

Data Analysis

Once the data were collected, various tools were employed to visualize and synthesize the data in order to understand patterns and make conclusions. Data analysis falls into two major categories: hydrologic, including all water level and well recharge measurements, and water quality, including all dissolved oxygen, pH, and sample analysis results.

Several parameters, both measured and calculated, were assessed for changing trends in time to help shed light on whether conditions at the site were different before and after the construction. In all cases, simple linear regression was used, and statistical significance was considered to be any slope resulting in a $p\text{-value} < 0.05$ (Formation Environmental, 2010). When changes over time were assessed for the site as a whole, the regression was based on data from the entire site averaged for each sampling date, this way eliminating the need for a mixed effects model (Kristensen et al., 2003). At this stage in the research, one year after construction, disjointed and short data sets prevent any strong conclusions from being made because of compounding variables such as weather, seasonality, and activity on neighboring properties. Statistics were used primarily to speculate about possible trends, but further years of data collection are needed for these trends to be decisively identified.

Hydrology

Depth to Groundwater and Groundwater Elevation.

Depth to groundwater was calculated as the measurement from the ground surface to the water surface, and takes into account any shift in the well casing, or earthmoving from the construction. Groundwater elevations were calculated using depth to groundwater measurements and the surveyed datum for each well.

Depth to groundwater and groundwater elevations were graphed over time for each well, including the Pond and all Ditch gauging stations, and trends were assessed with linear regression. The fifteen wells installed by the Trust for Public Land were monitored as early as the spring of 2013, and therefore have the most complete data (water levels were measured on average 57 times from each well). The eleven MSU (eight groundwater wells, two ditches, one pond), and six Hyalite wells that were monitored for a span of at least ten months in 2014 and 2015 were also considered, (water levels were measured on average 34 times from each well). Any TPL wells that were not replaced, and wells that were monitored for fewer than a span of ten months were not included in the analyses (7 wells total). These were wells that were either removed by construction and never replaced, or were wells that were installed late in the assessment period.

Groundwater Elevation Surfaces. Weekly groundwater and depth to groundwater surfaces were interpolated from water level data collected from the spring of 2013 through the summer of 2015, using an ordinary kriging function (Olmedo, 2014). For each surface, all available data collected on a given date was used, including all surface and groundwater locations. Ordinary kriging was chosen as an interpolation technique by first assessing the groundwater level data for a significant geostatistical component by comparing groundwater surfaces created by ordinary kriging and trend removed kriging. Three different methods for trend removal were considered: first, second, and third order polynomial. Assessment was based on the resulting semivariograms: if, when the trend was removed, the graphed variance resulted in a semivariogram with a clear shape and sill, geostatistics were deemed relevant. If instead, removing the trend resulted in a scattered variance with no clear shape and a relatively flat semivariogram, geostatistics were deemed irrelevant. This would imply that once the general trend in the groundwater level data is removed, the variance between water levels at two given points has no clear dependence on the distance between those two points. Three different sampling dates were analyzed, and for all three dates geostatistics were deemed irrelevant. The semivariograms became progressively flatter as the order of trend removal increased. Even first order trend removal resulted in variance with no clear spatial dependence.

Next, seven interpolation models were compared to determine the best fit surface: inverse distance weighting with a power of 1, 2, and 3, local polynomial interpolation with orders 1, 2, and 3, and ordinary kriging. Average prediction error was used to determine the strongest method. Prediction error was calculated by looping each model, removing one point for each run, and calculating the difference between the models predicted water level and the actual measured value at that point. An average of the absolute value of each error was then calculated. The model with the lowest average absolute error was considered the most adequate at interpolating a groundwater surface. For all three dates, ordinary kriging resulted in the lowest average absolute error, and was determined to be the best method of interpolating groundwater levels to create a groundwater surface. Since EG3 was so far away from the rest of the wells, there was concern that it could be stretching the model and disproportionately distorting the surface. Predicted values for surfaces created with and without EG3 were compared, and the average resulting difference in predicted values with and without EG3 was less than 3 cm. Any distortion from EG3 was deemed insignificant, and it was left in the model.

Each surface is a product of a data set from a particular day. As wells were installed, removed, reinstalled, ran dry, or froze, the composite data set was often slightly different from surface to surface. In some cases, particular sampling locations proved themselves to be of more significance than others in the appearance of a surface, revealing anomalies and particular site traits. Comparisons should be made with this in mind. It should also be noted that a large section of the land between the East Gallatin River and Bozeman Creek is outside the property boundary and was inaccessible for data collection. Error in the surfaces is lowest where available data is the most dense, and increases outside of the property boundary where data is more sparse.

Depth to Groundwater Surfaces. Depth to groundwater maps were also made, similarly to groundwater elevation surfaces. A key difference is that no surface water data from the creeks were included during kriging, as they un-representatively weight the interpolated values: although depth to groundwater in the creeks is zero, these areas are not indicative of a wetland, and are instead often surrounded by steep banks where depth to groundwater abruptly increases. Contour lines in this case represent depth to groundwater, and can be interpreted as areas of varying degrees of saturation.

Wetland Volume. Storage, the volume of groundwater contained within the site, was calculated and plotted over time. Water storage was calculated from kriged groundwater surfaces, so as to aerially weight the more sparse raw data uniformly across the site. Each surface was first created with all data available for a particular day, and then clipped to a universal size, consistent between all dates. This way, each surface was as robust as possible, and trends spanning three years could be assessed, despite varying ranges and densities of available data. Figure 25 shows the boundaries used to clip all other surfaces. For each clipped surface, an average water elevation was calculated. The lowest average water elevation was then used as a datum, and heights above this datum were graphed over time. Since the aquifer depth is unknown, total volume is not calculable. When integrated over the site's area, height becomes a volume, or storage.

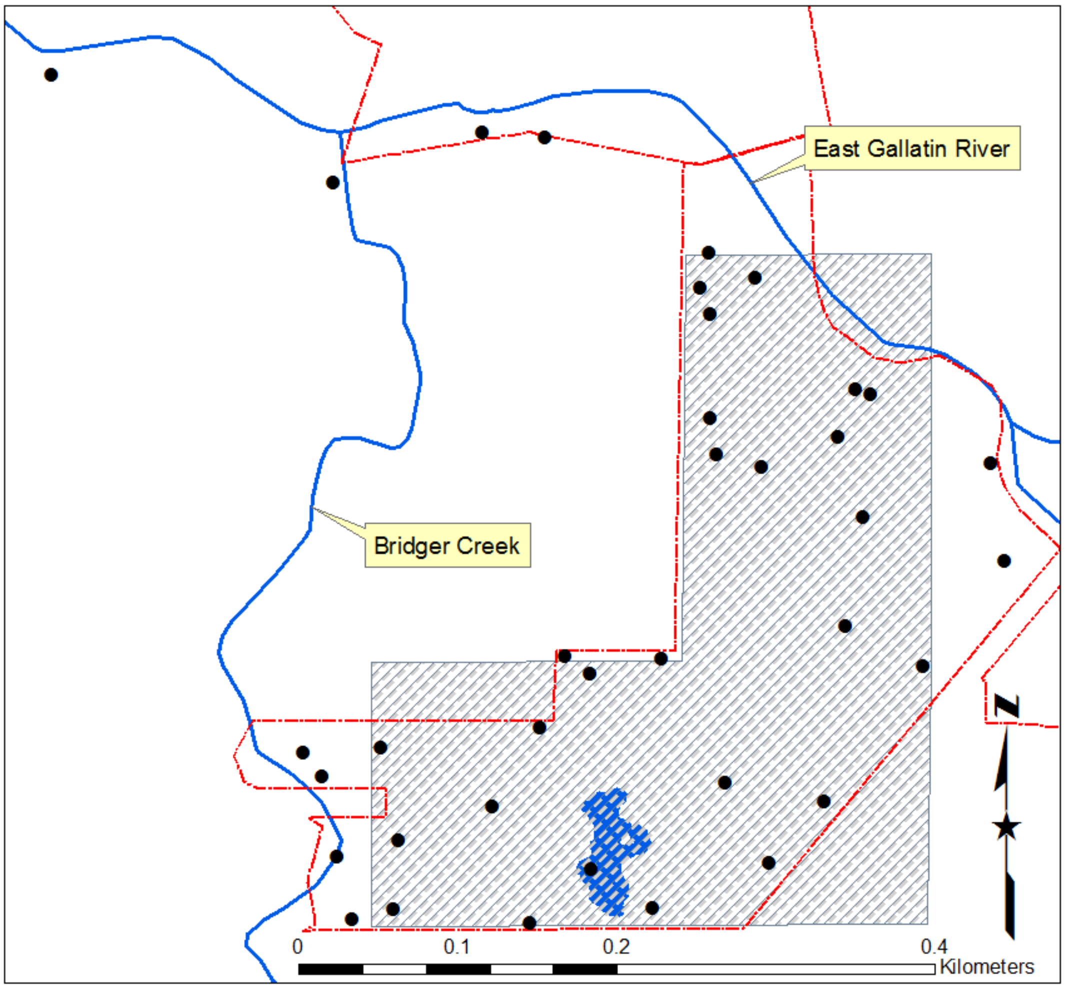

Figure 25. Cropped area used to calculate wetland volume and area.

Wetland Area. Wetland area was calculated from the depth to groundwater surfaces, and plotted over time. Wetland area was calculated as the extent of wetland where depth to groundwater was less than 30 cm. Wetland delineation should include indications not only of depth to groundwater, but also of hydric soils and hydrophytic vegetation, and should be shown to persist as average site conditions. For the sake of this analysis, just weekly snapshots of hydrology were considered, and results can only be deemed an estimate of wetland area. Just as for calculating water storage within the wetland, the kriged surfaces were cropped to a universal size after interpolation so that the most accurate map was made for each particular data set, and dates could be compared despite spatial variations in available raw data.

Continuous Water Levels. Continuously collected water level data were converted to elevation, and high-resolution stage graphs were created for three surface water gauging stations, and six groundwater wells.

Hydraulic Conductivity, Groundwater Velocity, and Hydraulic Residence Time. Groundwater recharge measurements were used to calculate hydraulic conductivity, and ultimately contributed to calculating groundwater velocity and hydraulic residence time with in the wetland. Hydraulic conductivity, or K_{sat}, was calculated from the slug tests using Hvorsleu's Method, and was averaged over time and space. K_{sat} can be expressed as:

$$K_{sat} = \frac{r^2 \ln(L/R)}{2LT_o}$$

where r = radius of the well, R = radius of the well casing, L = length of the slotted section of the well, and T_o = is the basic time lag, derived from the measured well recharge curve (Appendix E). To account for the greatest possible interaction between the wetland and the creeks, K_{sat} was conservatively increased: assuming heterogeneity in the soil and the possibility of gravel lenses, and acknowledging that water preferentially flows through the path of least resistance, the K_{sat} value used in subsequent analysis was the average observed value increased by an order of magnitude.

Groundwater velocity, or seepage velocity, is a function of hydraulic conductivity, gradient, and soil porosity. Seepage velocity differs from Darcy velocity in that it assumes water only flows through void spaces in the soil, and accounts for a decreased effective area by including porosity:

$$v_D = K_{sat} * \frac{dh}{dl}$$

$$v_s = \frac{v_D}{n}$$

where v_D= Darcy velocity, v_s= seepage velocity, K_{sat} = hydraulic conductivity, dh/dl = gradient, and n = porosity. Gradient was taken as the average difference in groundwater elevation from the southern to northern extent of the wetland, divided by the length of the flow path. Average gradient was used because the gradient was found to vary by a maximum of 0.002, and did not affect the velocity and HRT results. Porosity was estimated from the literature for a silty clay loam (Portage County Wisconsin). HRT is a function of volume contained in the system, and the flow rate through the system, which can also be expressed in terms of seepage velocity and the length of the flow path:

$$HRT = \frac{V}{Q} = \frac{L}{v_s}$$

where V = volume of water within the wetland, Q = flowrate of water through the wetland, v_s = seepage velocity, and L = length of the flow path through the wetland. Since the depth of the aquifer is unknown, length and seepage velocity were used to calculate HRT.

Water Quality

For all chemical parameters, when values were below detection limit, a value = $Detection\ Limit/\sqrt{2}$ was used (Croghan et al.). It is expected that the nutrient water quality results will follow a chi squared distribution, censored at zero. It has been found that this method of retaining information below detection limits results in the least distorted, and most accurate representation of data that has a chi squared distribution (Croghan et al.).

Water Chemistry Surfaces. Weekly surfaces were interpolated separately for each water chemistry parameter collected from the spring of 2013 through the summer of 2015, using the ordinary kriging function in R. Surfaces were created for nitrate, orthophosphate, sulfate, total nitrogen, total phosphorus, dissolved oxygen, and pH. The code and process developed for creating groundwater elevation surfaces was used, substituting chemical parameters for elevation. Surfaces with less than eight data points were not created.

Temporal Trends in Groundwater Chemistry. Temporal trends in all chemical parameters were assessed for the site as a whole, and for each groundwater well individually. Linear regression models were used, with time as the independent variable, and concentration or pH as the dependent variable. Data collected before August, 2014 was not used because it was so sparse with only two sampling dates from November 2013 to May 2014. To assess trends in the site as a whole, all data from groundwater wells and the pond were considered in the regression. Data from the creeks were not included. For trends in individual wells, data from each well was considered separately.

Parallel and Separate Lines Model. Nutrient concentrations were assessed for spatial dependence along the flow path of the groundwater at the site. Wells were grouped into north wells and south wells, and separate and parallel linear models were fitted over time. Statistical significance was considered to be any slope or factor resulting in a p-value < 0.05. Models were based on averages, where all data in the north and all data in the south was first averaged for each sampling date before applying the model. This eliminates the need to consider mixed effects, and only the fixed effect of location was considered (Kristensen et al., 2003). Trends in orthophosphate, TP, nitrate, and TN were assessed. The separate lines model was first used to determine if nutrient levels fit a trend over time, and if trends were statistically different based on being located in the north or the south. In other words, did the nutrient levels decrease along the flow path, and did the difference between the concentrations found in the north vs. the south change with time, which could suggest improved or degraded treatment. The model is represented by:

$$\mu\{[Nitrogen]|time, location\}$$
$$= \beta_o + \beta_1 * time + \beta_2 * location + \beta_3 * time * location$$

where β_o = the average nitrogen concentration in the wetland at the start of the project, β_1 = the average change in nitrogen concentration in all wells over a year, β_2 = the average change in nitrogen concentration based on if wells were located in the south or the north, β_3 = the average difference between changing nitrogen concentrations over time in wells located in the south vs wells in the north. If it was found that there was no interaction between time and location - determined by the statistical significance of β_3 - fitted lines for wells in the north and the south were deemed to have the same slope. At this point, the parallel lines model was assessed. This model determines if the fitted line for wells in the north vs. the south are statistically separate: on average, do they have different nutrient concentrations, and by how much. The parallel lines model is represented by:

$$\mu\{[Nitrogen]|time, location\} = \beta_o + \beta_1 * time + \beta_2 * location$$

where all parameters are defined the same as in the separate lines model, only β_3 is no longer included.

Surface Water Quality. The surface water environment was assessed separately. Plots of each chemical parameter were created over time. Linear regression was not used, anticipating strong seasonal patterns, and the need for larger datasets. If nutrient concentrations in locations downstream of the wetland were different than in upstream locations, speculations as to the role of the wetland could be made. It is important to note that the water quality in the creeks is far more sensitive to watershed processes than the groundwater in the wetland, due to the relative velocities of the water conveyed in the creeks vs. through the soil. Changes in farming or storm water management practices upstream may be directly measurable in the creeks at Story Mill, but may not be in the groundwater. Even just the short length of Bozeman Creek and the East Gallatin River that border the site are being influenced by other vast watersheds to the east, west, and north.

Database

All data collected, this thesis, and R analysis code will be published and made available through the Montana State University ScholarWorks database: http://scholarworks.montana.edu/.

RESULTS

Hydrology

Depth to Groundwater and Groundwater Elevation

Depths to groundwater and groundwater elevation were plotted relative to time to help understand the potential changes in site wetland characteristics. Two major construction activities potentially impacted the wetland area: filling the drainage ditch, and recontouring the ground surface. A change in depth to groundwater may have been due to either activity, whereas a change in groundwater elevation is more likely associated with filling the ditch. Trends from wells with longer data sets (TPL wells), spanning from the spring of 2013 through the summer of 2015, are more statistically robust than trends from wells that only including one year of data (MSU and Hyalite wells). Data from the TPL wells though, are discontinuous, with a gap spanning the fall/winter/spring between monitoring in the summer of 2013 and summer 2014. Conclusions based on such short and disjointed data sets are not advisable, but potential trends are still worth mentioning and merit future monitoring.

Only seven wells, out of 31, showed statistically significant trends in changing groundwater elevation over time (Figure 26), three of which increased, while four others decreased. Data from all other wells that did not show statistically significant trends were also plotted and shown in Figures 27-29. Note the coincident pattern observed in most wells, where higher water levels were measured in the winter than in the summer, indicative of a seasonal signal.

Decreasing trends in Ditch2 and 3 are very slight, approximately 0.1 and 0.2 m during the monitoring period respectively, but are definitive. This is the opposite of what might be expected if the ditch had been acting to drain the site before restoration. Decreasing trends in TPL13 and 11 also do not jibe with this notion, where wells in close proximity to the ditch were expected to recharge instead of become drier. It is possible that the ditch had in fact been acting as an infiltration gallery, and that after construction, it is cut off and no longer being fed by “upstream” water from the pond, causing it, TPL13, and TPL11 to dry up.

Increasing groundwater elevation trends were observed in F5, TPL2, and the Pond. The Pond is included here because data collection spanned 10 months, but note that there is only one pre-construction data point. Confidence in the observed increased trend is suspect. Since the outlet to the pond was a head gate prior to the restoration, the hydraulics of the pond may have been altered from filling and draining patterns expected of a naturally groundwater driven system. It is unclear as to how the head gate was manipulated, and to what extent, if any, it was damning or draining the groundwater in the pond. A rise in groundwater elevation in the wells surrounding the pond was not observed (TPL4 and TPL8), further eroding the significance of the increased trend in the in pond.

Groundwater elevation increased in TPL2. TPL2 is located between two independently fluctuating free-water surfaces, whose relative elevations dictate the steepness of the gradient into or out of the wetland to the west. This relationship would need to be modeled in more detail to understand the reason for an increased groundwater elevation in TPL2. If indeed the water surface in the Pond did increase, this mounding may have created a more preferential gradient towards Bridger Creek, also causing the water levels in TPL2 to rise. This speculation is not supported by an increasing trend in MSU1.

The reason for the increase in groundwater elevation in F5 is also not completely clear. An increase in groundwater elevation in F5, just down gradient of the ditch may suggest that the ditch was acting to intercept groundwater and shunt it northeast to the East Gallatin River. In the advent of filling the ditch, that groundwater now flows north through the soil unchecked, resulting in more saturated conditions around F5. But, as was discussed before, this theory is contradicted by a decrease in groundwater elevation observed in Ditch3. And once again, increasing trends are not reciprocated in neighboring wells such as MSU5 and TPL12.

The contradicting results in F5 and Ditch3, and increases or decreases in groundwater elevation in isolated wells leave more questions than answers. These inconsistencies may point to soil heterogeneity, causing different localized responses to the restoration. It also highlights how little was known of the system before construction, and the importance of pre-restoration monitoring. What is clear, is that at this stage - one year after restoration - is there is not a definitive and pervasive response in wetland area or volume to filling the ditch.

Four of the seven wells with significant groundwater elevation time trends also showed a significant depth to groundwater time trends (Figure 30). Data from all other wells that did not show statistically significant trends were also plotted and shown in

Figures 31-33. For wells that were located in areas where there was no construction, the groundwater elevation and depth to groundwater plots should be mirror images of each other. This is the case for TPL13 and essentially TPL11: depth to groundwater in TPL11 measured zero during April and May 2014 when water elevations were measured above the ground surface. Depth to groundwater in TPL2 exhibited the strongest trend, as would be expected because it is located in the Backwater Slough, where 1-1.5 m of soil was removed from the floodplain. TPL8 had no trend in groundwater elevation, suggesting that any change in depth to groundwater is solely due to lowering the ground surface. Although F5 was also located in an area of construction, this plot is an exact reciprocal of the groundwater elevation plot, suggesting that hydrologic changes here are primarily due to a rising groundwater table.

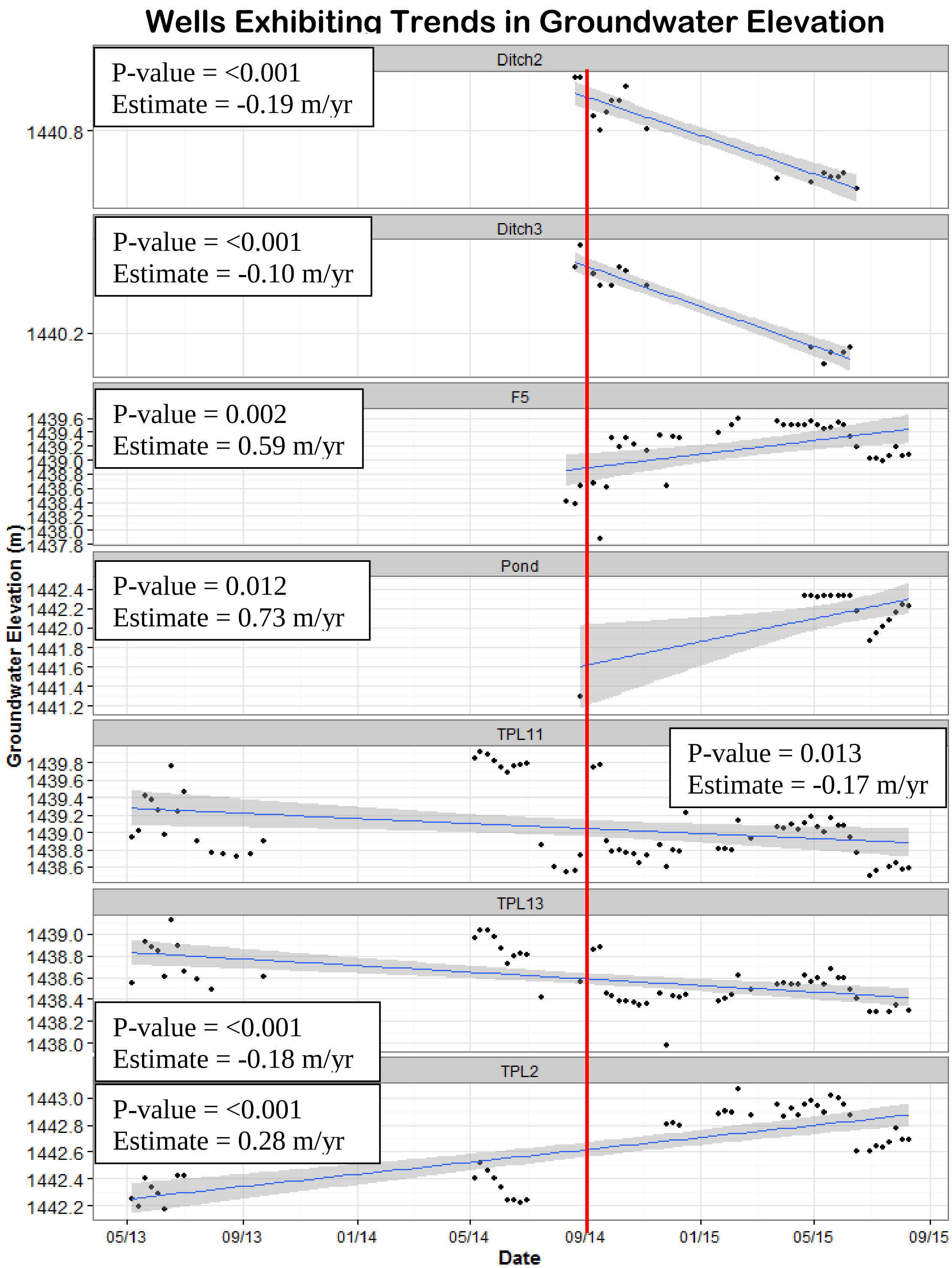

Figure 26. Plotted groundwater elevation in wells with significant trends over time, indicating localized changes in wetland volume. Blue lines represent a fitted simple linear regression, and gray areas represent each model's 95% confidence interval. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

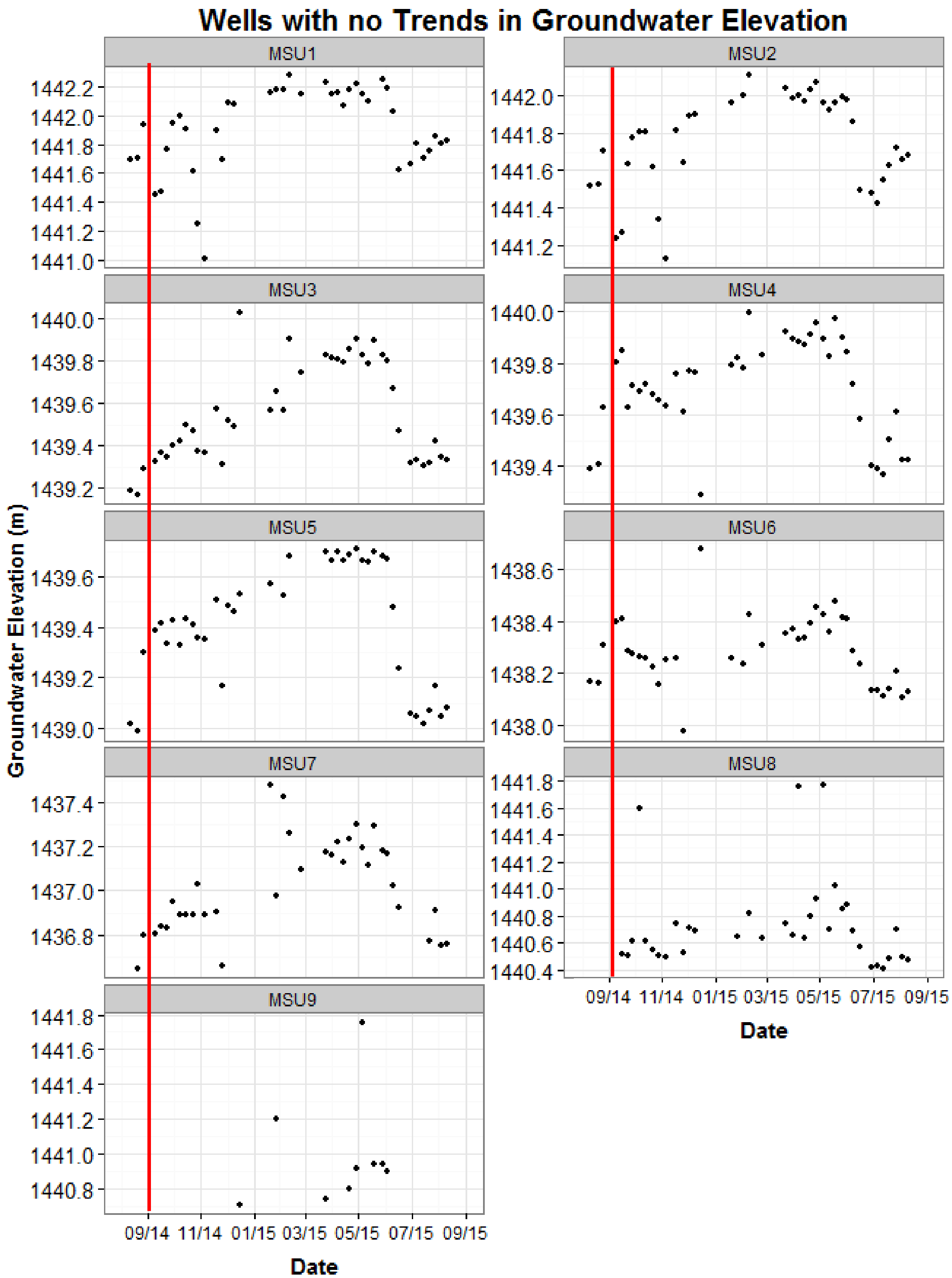

Figure 27. MSU groundwater wells with no significant trends in groundwater elevation during the monitoring period. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

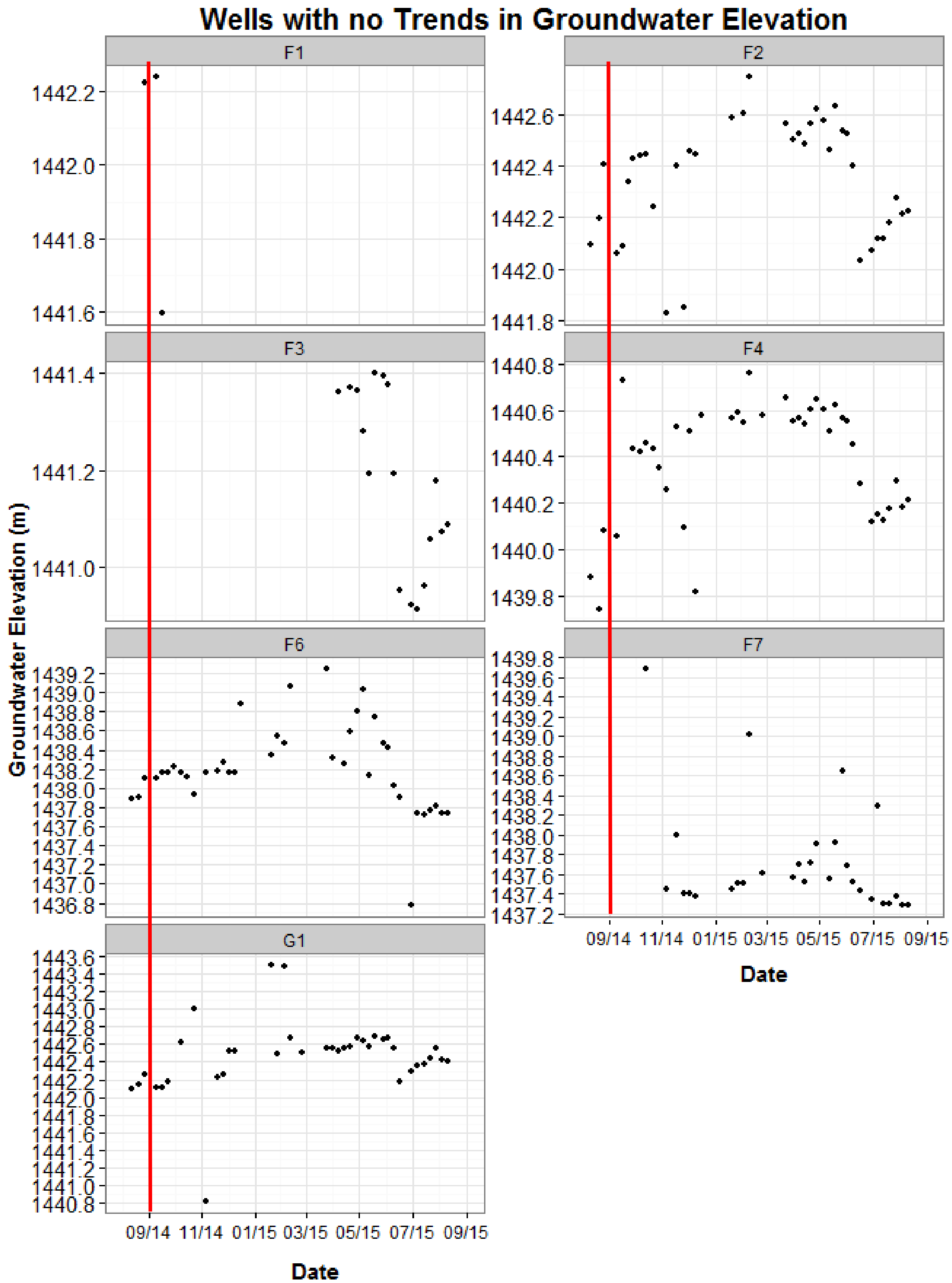

Figure 28. Hyalite Engineering groundwater wells with no significant trends in groundwater elevation during the monitoring period. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

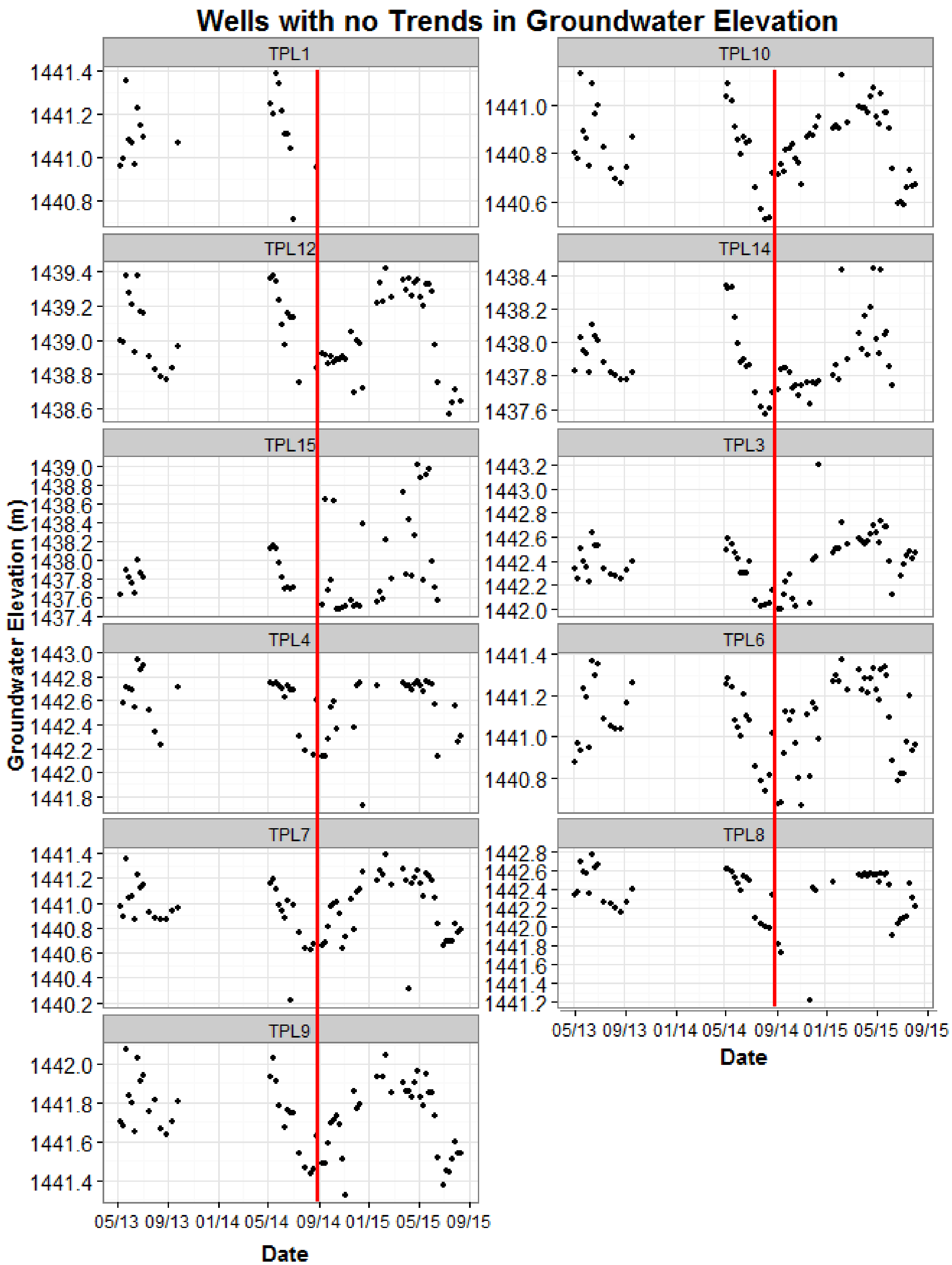

Figure 29. TPL groundwater wells with no significant trends in groundwater elevation during the monitoring period. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

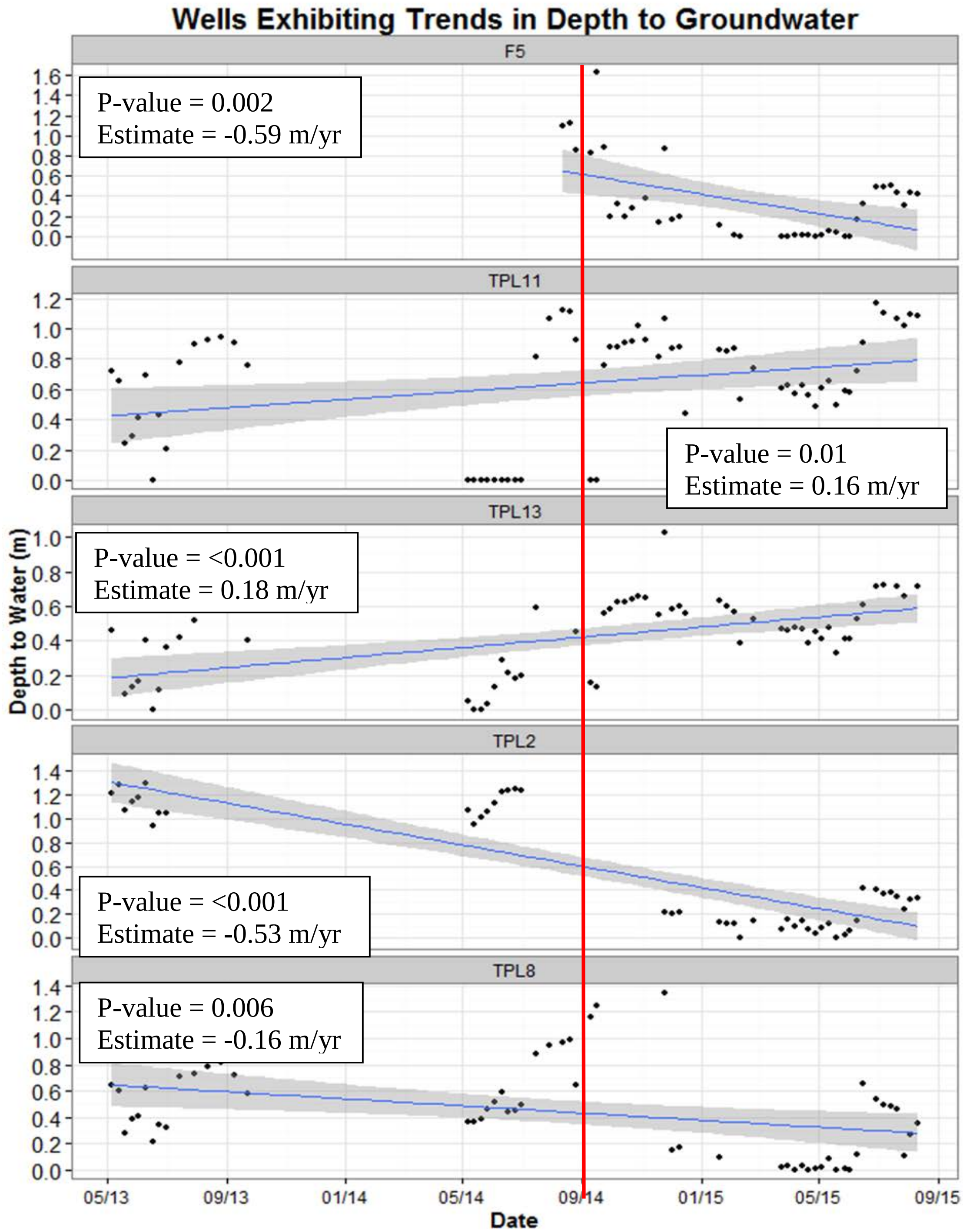

Figure 30. Plotted depth to groundwater in wells with significant trends over time, indicating localized changes in soil saturation. Blue lines represent a fitted simple linear regression, and gray areas represent each model's 95% confidence interval. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

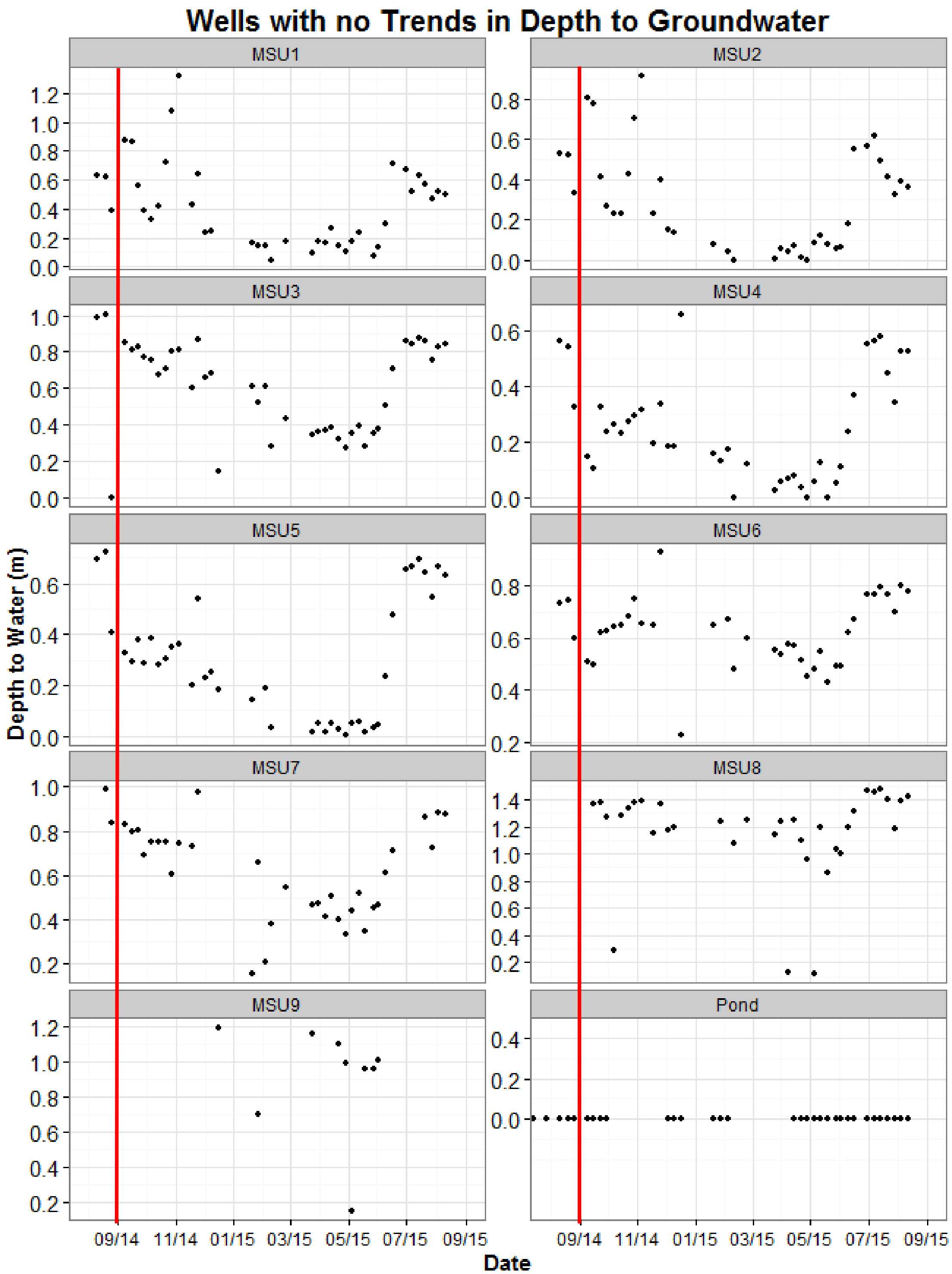

Figure 31. MSU groundwater wells with no significant trends in depth to groundwater during the monitoring period. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

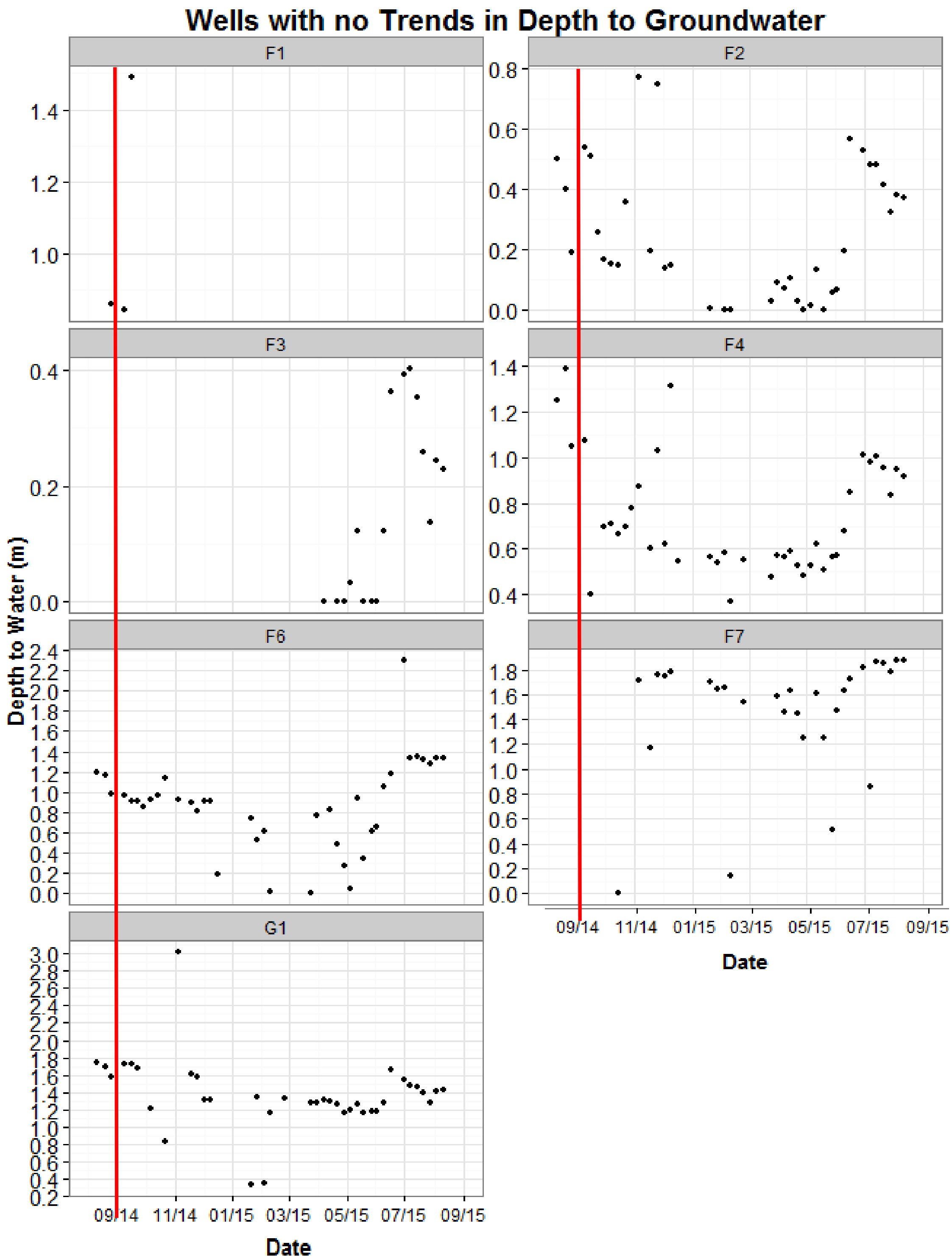

Figure 32. Hyalite Engineering groundwater wells with no significant trends in depth to groundwater during the monitoring period. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

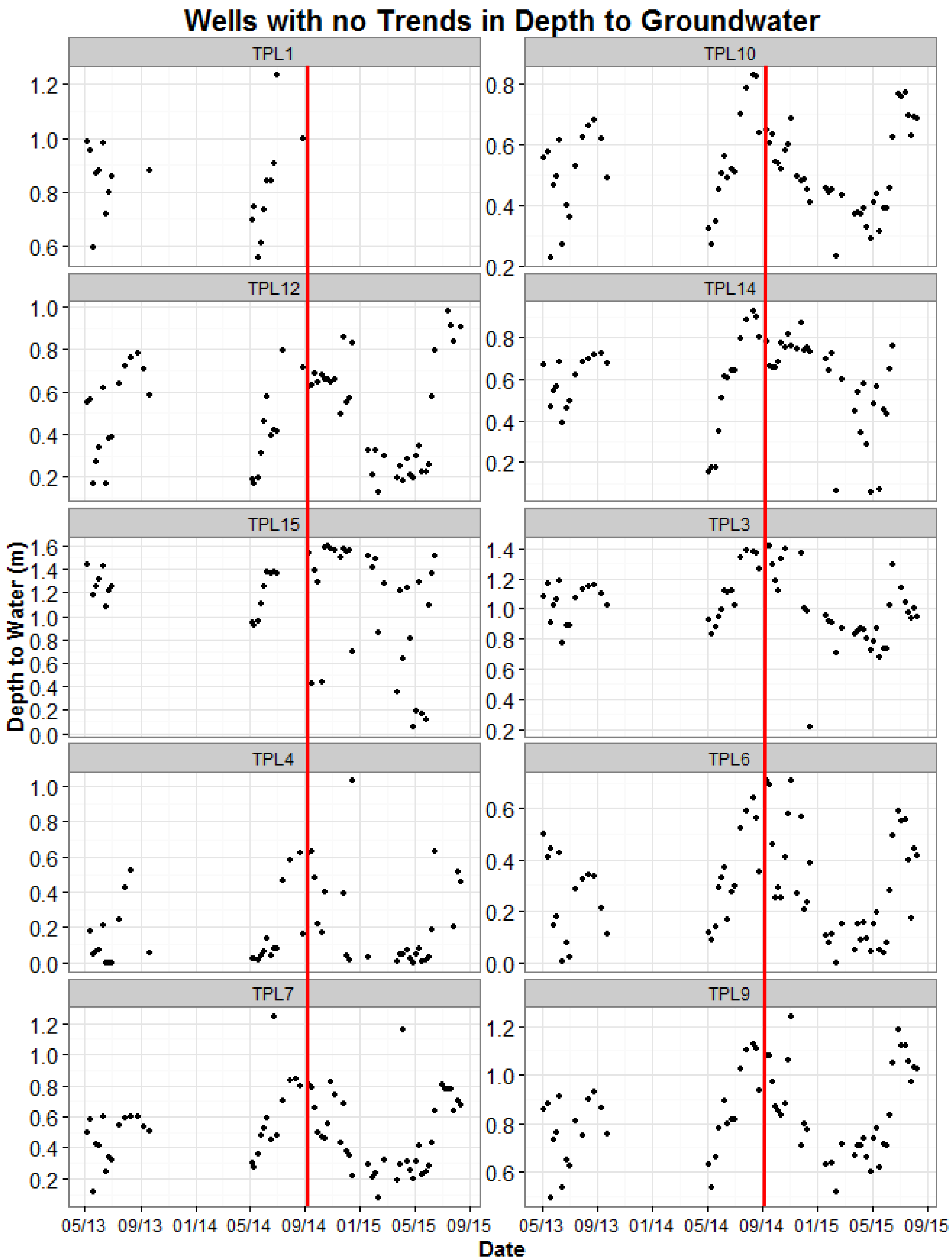

Figure 33. TPL groundwater wells with no significant trends in depth to groundwater during the monitoring period. The vertical red line marks the date when construction began. All y-axes are displayed in increments of 0.2 m.

Groundwater Surfaces

Changes in groundwater surfaces reveal general changes in hydrologic trends, groundwater/surface water interconnectivity, and wetland volume. Of the 68 groundwater surfaces created, several were of notable interest. Two separate groups of surfaces will be discussed. The first group addresses changes in groundwater flow direction and potential groundwater/surface water interaction. The second group specifically illustrates the effects of filling the ditch on wetland volume. In the images presented, higher elevations are represented by light blue, and lower elevations are red. Red dots denote groundwater measurements that contributed to creating a particular surface. Note that error is lowest where there available data is denser, and error increases outside of the property boundary where data is sparser. Flow direction is assumed to be perpendicular to groundwater surface elevation contour lines.

The possibility of hydrologic connectivity between the creeks and the wetland depends on flow paths originating near Bridger Creek in the southwestern extent of the property, and along the entire length of the East Gallatin River as it circumnavigates the site to the east and north. These are areas with the highest potential for surface water interaction with the restoration efforts. The groundwater surfaces shown in Figure 34 are representative of general patterns in groundwater flow throughout the duration of monitoring at Story Mill. Images A, B, and C, are snapshots each a year apart, from the first week of May in 2013, 2014, and 2015. Image D is from the last sampling date in August 2015. Images A and B are indicative of before construction conditions, and images C and D represent after construction.

These images indicate that the hydraulic gradient drives groundwater flow from south to north, and then west, following the general flow direction of the creeks. The images also show persistently higher groundwater elevations in the center of the wetland, both before and after construction. The contours in all images show that the water table slopes down to both creeks, implying that the wetland is losing water to the east, west and north, and that limited surface water is entering the wetland. One particular difference is the “bulls-eye” circling the northern wells in images A and B is not present in images C and D. This is most likely due to the density and location of available data, and not to differences between before and after construction. Otherwise, the hydrology displayed by all surface plots shown is remarkably similar, with little to no change in hydrologic connectivity between the creeks and the wetland associated with the restoration.

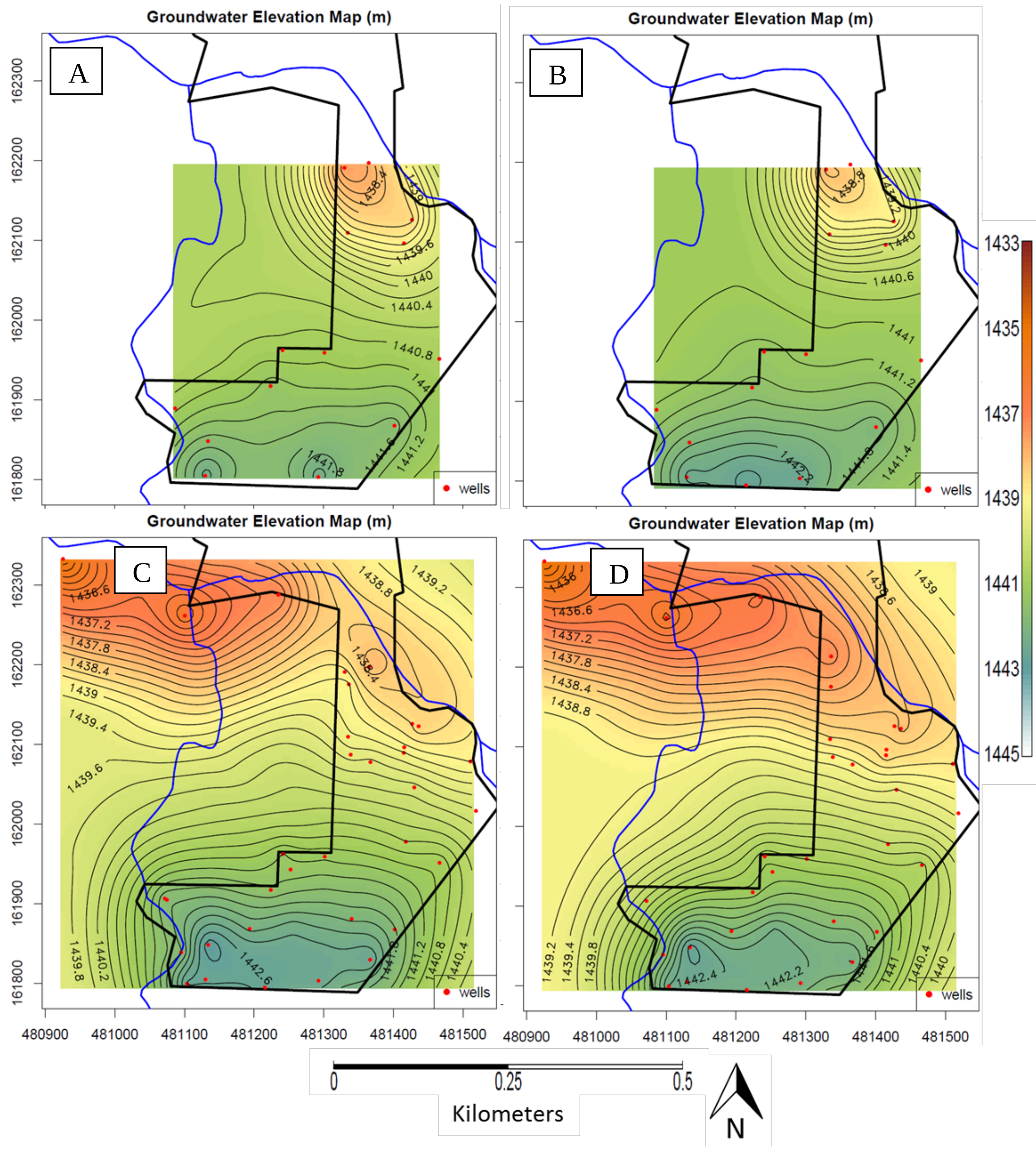

Figure 34. Groundwater Elevation surfaces. Clockwise from the top: Image A, 5/6/3013; Image B, 5/6/2014; Image C, 5/5/2015; Image D, 8/11/2015. Images A and B are indicative of before construction conditions, and images C and D represent after construction.

An inspection of groundwater elevation surfaces on dates before and after filling the ditch reveals a few patterns of interest. Limited high resolution data sets are available to illustrate for “before-restoration” hydrology: three high resolution groundwater surface maps exist from before filling the ditch (Figure 35, Images E, F, and G), and one exists between filling the southern portion of the ditch where it exists the pond (Figure 35, Image H). Figure 36 displays “after-restoration” hydrology, where Images I and J are of the two weeks following filling the ditch, and Images K and L are from several months later, in April and June of 2015. When comparing between surfaces, many things within the watershed are changing, and thus all differences cannot be directly related to filling the ditch. For example, during ditch construction, precipitation events and evapotranspiration were occurring all the while, and the pond was simultaneously being dewatered into the field just to the east.

There are several differences between these eight maps to point out. All maps from before filling the ditch show Ditch2 to be higher in elevation than its direct neighbors, TPL6 and 7. After filling the ditch, Images I, J, K, and L all show this relationship reversed, where Ditch2 then became lower in elevation than TPL6 and TPL7, illustrating the decreased trend in groundwater elevation observed in Ditch2. The perched water table in the ditch before being filled and the depressed water table after, implies that the ditch was acting as an infiltration gallery, and not a drain. Images G and H (before filling the ditch, and after filling the outlet to the pond, but before filling the norther portion of the ditch, respectively) show bunched contour lines around the northern portion of the ditch. It appears that the gradients here potentially drove water both through the ditch out towards the East Gallatin River, and out of the ditch towards

F5, MSU 5, TPL13, and MSU6. After filling the ditches, the surfaces consistently show the water spreading out and flowing in a more regular distribution across the site.

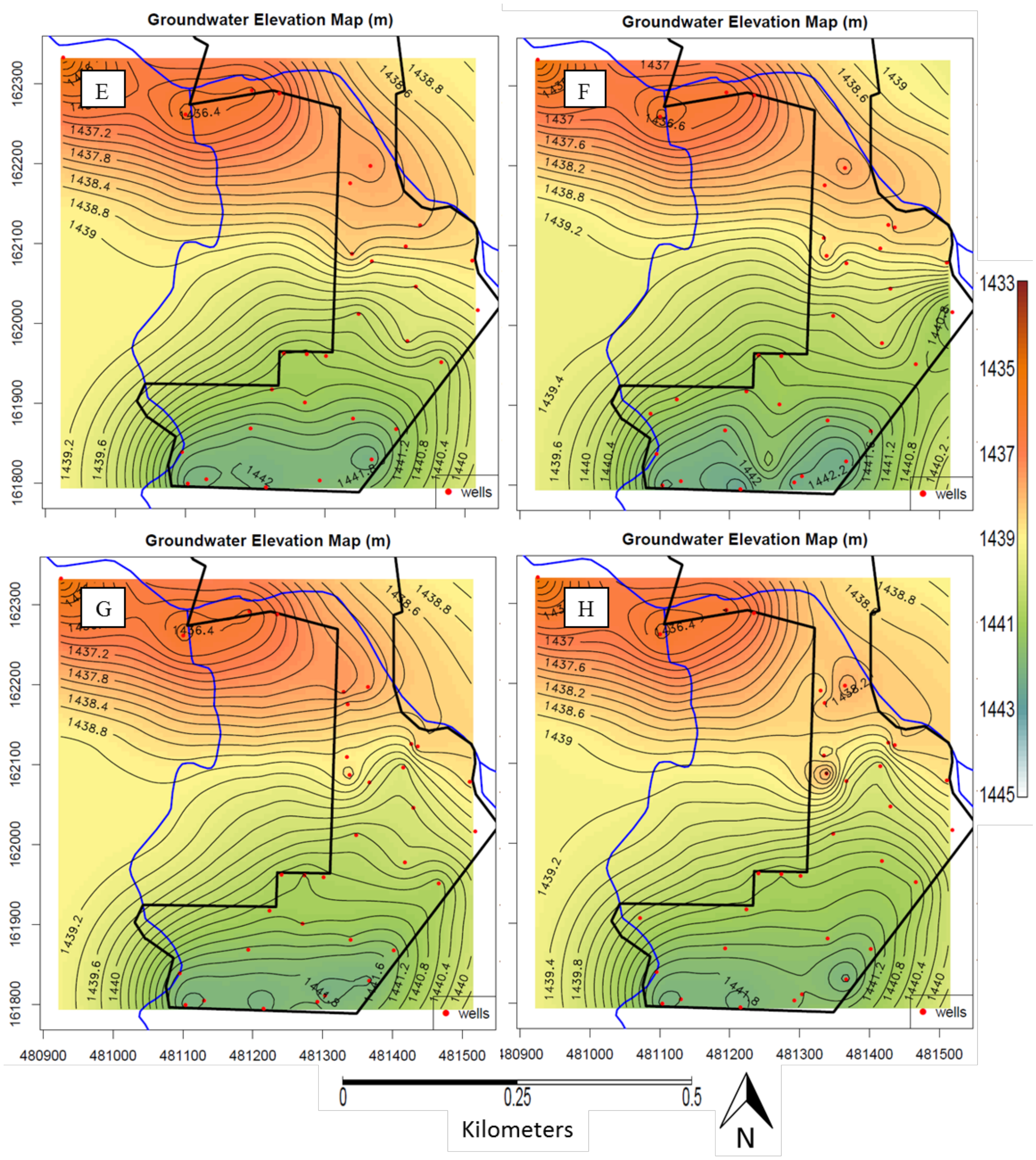

Figure 35. Groundwater elevation surfaces. Clockwise from the top: Image E, 8/20/2014, Image F, 8/26/2014, and Image G, 9/9/2014 are from before filling the ditch; Image H, 9/16/2014 is from after filling the ditch near the outlet from the pond.

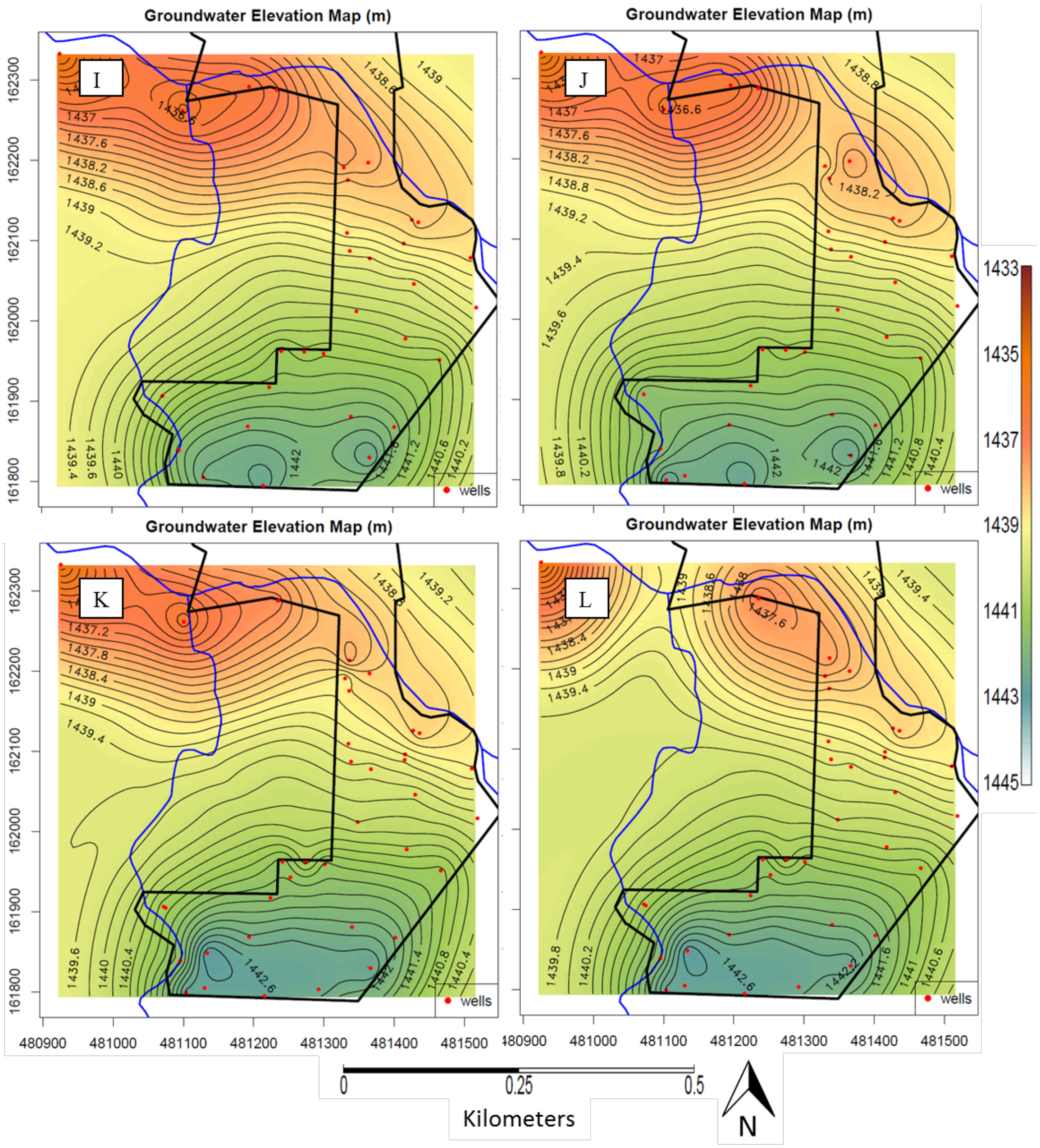

Figure 36. Groundwater elevation surfaces. Clockwise from the top: Image I, 9/29/2014, Image J, 10/7/2014, Image K, 4/28/2015, and Image L, 6/2/2015 are all from after both the southern and norther portions of the ditch were filled.

Depth to Groundwater Surfaces

Maps of depth to groundwater are shown in Figure 37, and can be used to help spatially visualize the effects of changing depth to groundwater in individual wells on wetland area. Each image is approximately a year apart, representing high water levels in the spring of 2013 (Image H), 2014 (Image I), and 2015 (Image J). Blue indicates potential wetland areas where depth to groundwater is less than 0.3 m. As depth to groundwater increases, and the tendency for an area to be drier, and more upland-like, the map color transitions first to green, then yellow, and finally red. The last surface shown, Image J, is created from a higher resolution data set, which reveals some localized nuances that cannot be deciphered from dates before August 2014, and more clearly outlines the wetland boundary. The increased wetland area in Image J around MSU2, F2, and MSU4, is an artifact of a higher resolution data set, as none of these wells were in areas where construction occurred or exhibited trends in decreased depth to water. The increased wetland area encompassing the Backwater Slough and around the back of the pond, on the other hand, accurately displays the decreasing trends in depth to groundwater seen in TPL2 and TPL8. The maps also shows that wetland area potentially increased in the meadow down gradient of where the more northern portion of the ditch was filled, exhibiting the trend in decreased depth to water observed in F5. Also notice that the area around TPL11 and TPL13 is drier in Image J than in Images H and I, confirming the decreased groundwater elevation trends observed in these wells.

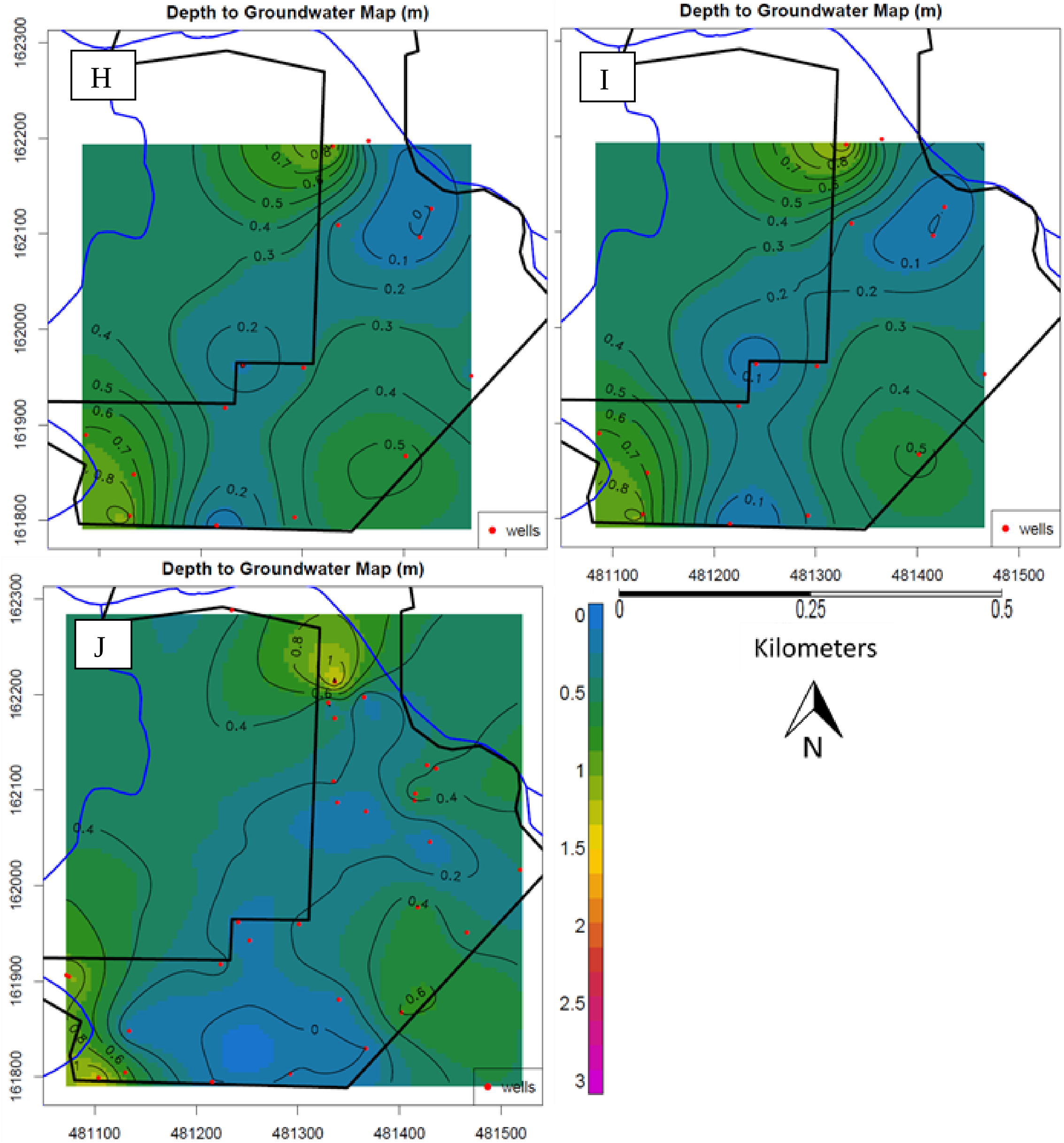

Figure 37. Depth to groundwater surfaces, indicating potential areas of wetland, considered to be where depth to groundwater is less than 0.3 m. Clockwise from the top: Image H, 6/17/2013, Image I, 5/12/2014, Image J, 5/19/2015. Each image represents high water levels measured in the spring.

<u>Wetland Volume</u>

To understand the pattern of groundwater volume within the wetland over time, a graph of changing water storage within the wetland was created (Figure 38). This graph represents when water entered and left with changing seasons, and how it was affected by local processes like infiltration and evapotranspiration. Recall that the plotted values are above a datum (1436 m), below which is assumed to be the steady state volume within the wetland. Precipitation and temperature data are also shown for the duration of the project to help give context to the water height data.

Average water levels varied approximately 37 cm over three years of data collection. When applied to the area contained within the southern parcel, this change in height represents almost 39 million liters of water gained and lost. General losing trends can be observed throughout each summer, and data from 2014/2015 suggests a gaining trend throughout the winter. Peaks and valleys in the storage graph during spring/summer/fall tend to mimic rainfall events. In the winter, this relationship is less clear, but at least two of the peaks occurred during periods when temperatures were sustained above freezing, which potentially caused the snowpack to melt and increase storage. By simple linear regression, there is weak evidence to support that storage decreased from 2013 to 2015 (p-value = 0.049), but there is strong evidence to support that average precipitation remained consistent (p-value = 0.27).

Considering how slowly the groundwater moves, variation in the storage graph is unlikely due to trends in groundwater flow. The seasonality observed in water levels in individual wells and for the site as a whole is typical of losses due to evapotranspiration:

plants begin to draw water levels down in early spring and throughout the summer, senescing in the late fall, allowing storage to recharge in the winter. Evapotranspiration rates were measured to be 105 cm annually at a site with similar mixed grass makeup as the Story Mill wetland (Laczniak et al., 1999). Daily ET, as estimated by the Irrigation Guide for Montana, is approximately 0.46 cm/day for grasses, and 0.56 cm/day for orchards (Linford, 1974). By these values, ET rates could easily account for the greatest fraction of annual variability, with an estimated annual loss of 34.5 cm occurring in the spring through the fall of 2014 (approximately 0.16 cm/day). During the winter, decreases in storage are more difficult to explain, as ET is assumed not to contribute to losses.

There are several speculative reasons for the discrepancies in a direct precipitation/ET – storage relationship. Precipitation events in Montana can be highly localized, and recall that the weather station is located 3.8 km from the Story Mill wetland. Other possibilities may be neighboring development (for example, the site just to the south was dewatered for the construction of office buildings), variations in plant activity and community, different spring runoff patterns, and manipulation of the pond outlet.

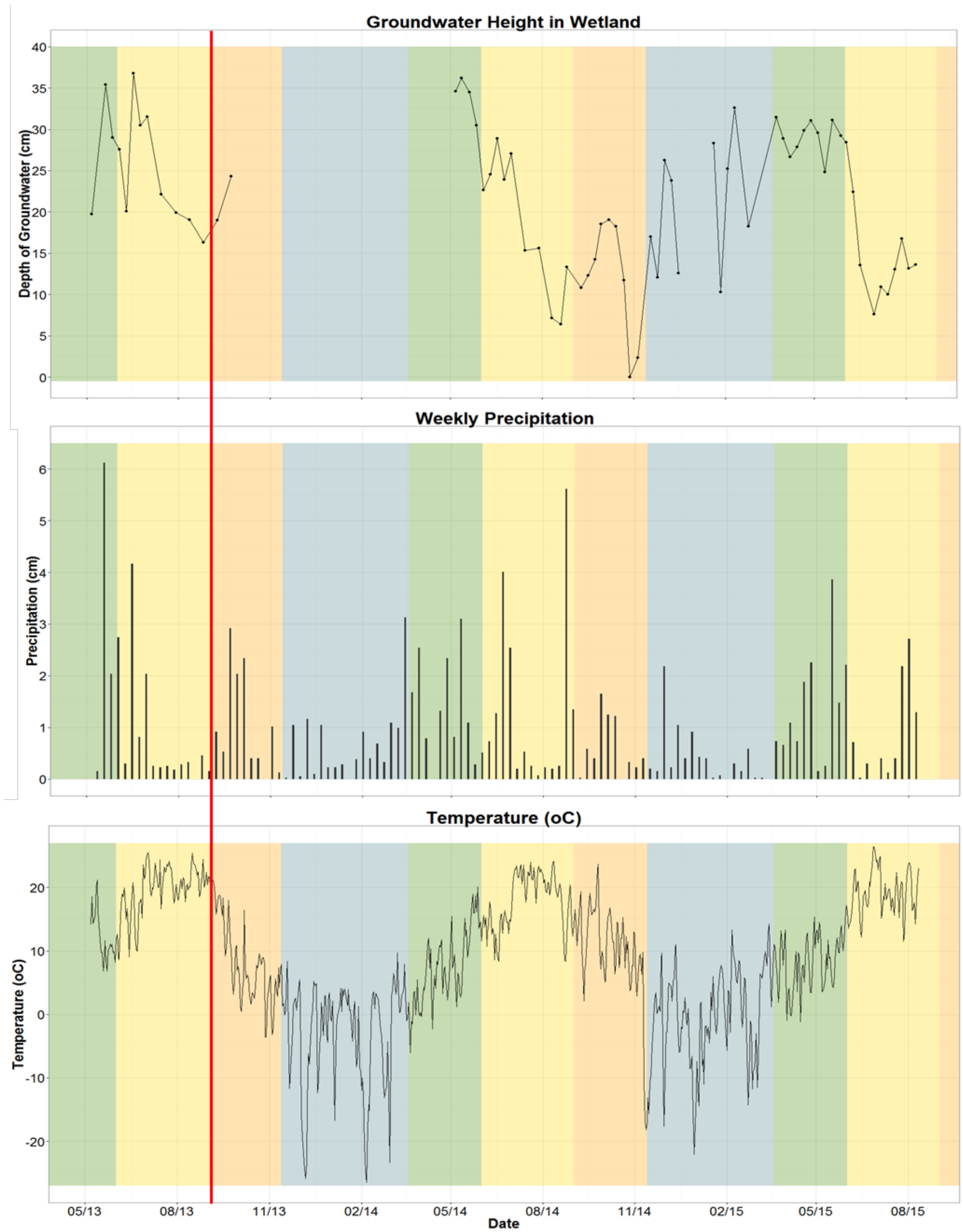

Figure 38. Average groundwater height in the wetland, over time. Each color represents a season, where green is spring, yellow is summer, orange is fall, and blue is winter. Precipitation events year round, and warm temperatures in the winter are coincident with most gains in storage, while periods of drier weather are coincident with losses. The vertical red line marks the date when construction began.

<u>Wetland Area</u>

The combined effects of floodplain excavation, recontouring, and filling the ditch on wetland area, as defined by depth to groundwater less than 0.3 m, was analyzed by looking at a plot of wetland area over time. In Figure 39, wetland area is shown in hectares, plotted against time. Keep in mind that the sparseness of groundwater wells before September 2014, and the short nature of the data set, makes it difficult to make conclusions with confidence. That being said, the weekly fluctuations in the area generally mimic the patterns seen in the storage graph, but the overall trend is different: there is moderate evidence to suggest that wetland area increased since 2013 (p-value = 0.026. Despite a decrease in groundwater storage, localized increases in groundwater elevation and lowering the ground surface in areas ultimately increased wetland area.

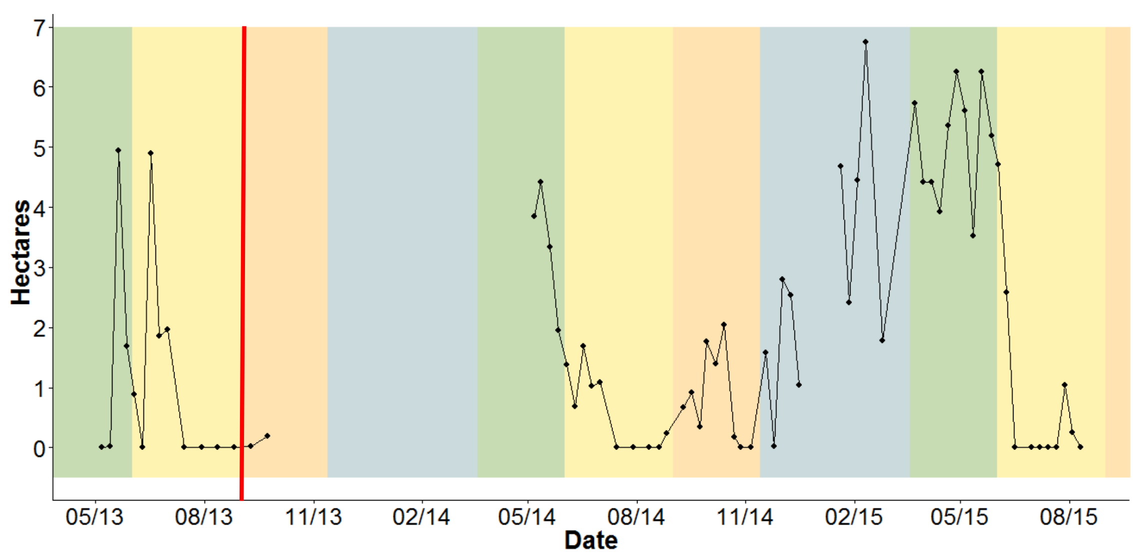

Figure 39. Total wetland area plotted over time, where wetland area is considered to be where depth to groundwater is less than 0.3 m. Each color indicates a season, where green is spring, yellow is summer, orange is fall, and blue is winter.

Continuous Water Levels

Data collected by the HOBO data loggers in 2015 were also used to understand flow direction and potential hydrologic connectivity between the wetland and the creeks. Wells were arranged in a transect between Bozeman Creek and the East Gallatin River, and in comparing their hydrographs it can be determined if any water is flowing east or west, into or out of the wetland. The graph (Figure 40) depicts water elevation above sea level, over time, making the relationships of water elevation between each well apparent. Again, it is evident that water levels in the center of the wetland are higher than in the creeks. MSU1 is consistently almost a half of a meter higher than BC1, except for a short period in June and July. From MSU1, groundwater elevation decreases monotonically moving east along the transect to the East Gallatin River. Notice also that all groundwater wells, except TPL7, begin to recharge in the fall, after about mid-September, but that the creeks do not.

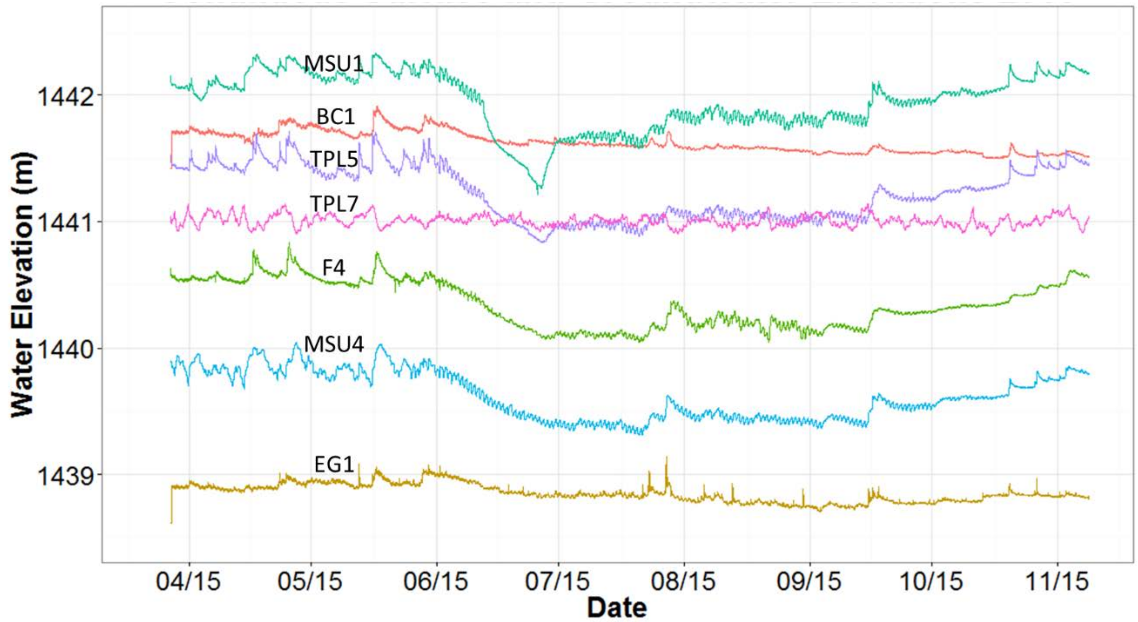

Figure 40. High resolution groundwater elevation in wells arranged in a transect between Bozeman Creek and the East Gallatin River. Water levels are almost always highest in the center of the wetland, and decrease progressively both west toward Bridger Creek, and east toward the East Gallatin River. Water in the Bozeman Creek channel is approximately 2.8 m higher than in the East Gallatin channel.

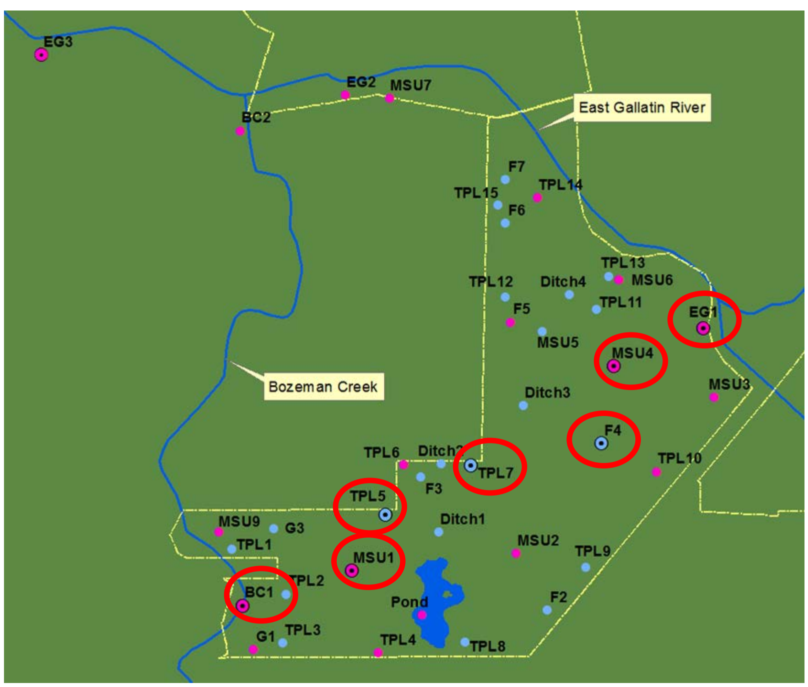

Figure 41. Locations of data loggers arranged in a transect running from Bozeman Creek to the East Gallatin River.

These high resolution data sets can also provide insight into the water balance at Story Mill. Strong diurnal patterns can be observed in the groundwater and stream levels. Figure 42 shows a period of five days with no precipitation during early September 2015. Patterns of drawdown begin approximately around 8 am, until 6 pm, and have an average magnitude of 2.4 cm (measured stage adjusted for a soil porosity of 0.4). This value is far greater than the average estimated ET of 0.5 cm/day (Linford, 1974). This discrepancy is unlikely due to influence from the creeks because results from the groundwater surfaces and the continuous water level transect indicated a gradient in the opposite direction, out of the wetland. It is possible that plants preferentially extract water from the well casing and surrounding sand grout, instead of from the adjacent native soil. Hydraulic conductivity is essentially unhindered within the silica sand surrounding the wells, as opposed to water tightly held by the wetland soils high in silt and clay and organic matter. This could essentially create a “short circuit” scenario, with ‘apparent’ ET values an order-of-magnitude greater than ET estimated from consumptive crop use. Rapid night-time recharge could be due to the cones of depression created in each well during the day by neighboring grasses, sedges, forbs, and trees, which then refill overnight. Over the six-day period shown (Figure 42), total drawdown was ~1.2 cm, or 0.2 cm/day (measured stage adjusted for a soil porosity of 0.4), supporting that general drawdown patterns at this time of year, under these conditions, are most likely due to ET but which are amplified each day by the well construction materials (sand packing).

It can also be observed that during this period, the diurnal pattern in the creeks are not as distinct as in the groundwater wells, and levels do not show the same decreasing trend as in the groundwater. Furthermore, no consistent pattern in lag time between the creeks and the groundwater wells was observed. This indicates that there may be activity upstream in the watershed influencing diurnal creek flow, as oppose to localized activity in the wetland influencing groundwater patterns.

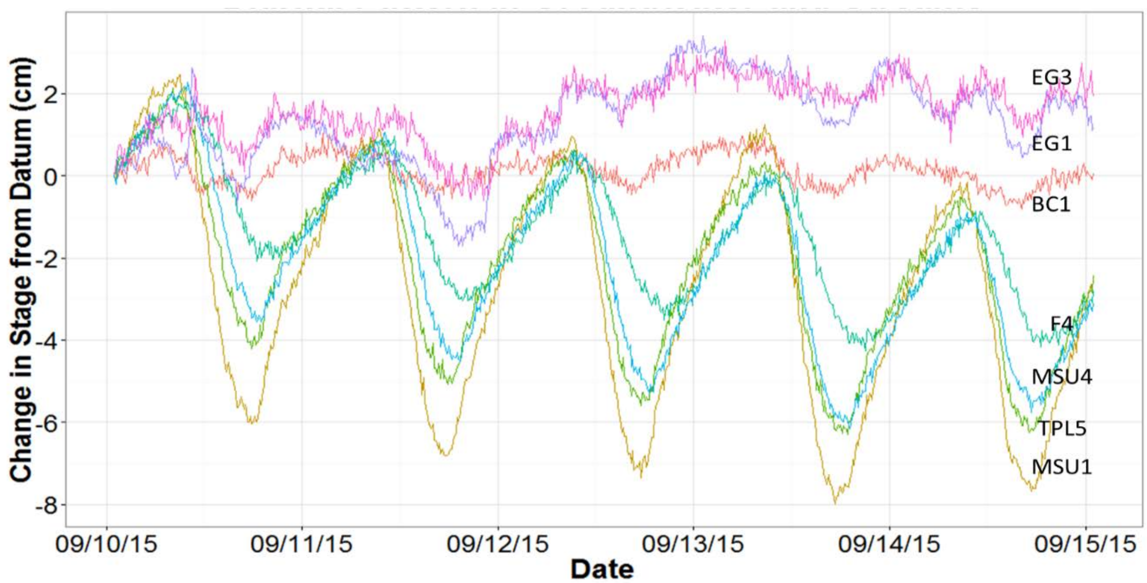

Figure 42. High-resolution water level data indicate strong diurnal patterns in the groundwater wells, which do not match patterns observed in the adjacent streams (East Gallatin (EG1 and 3); Bozeman Creek (BC1)). The datum was establish such that the first reading for all locations began at zero to clearly see relative patterns.

Further water balance analysis can be done by comparing high temporal resolution water level data to precipitation events. Figure 43 is a graph of groundwater levels above a datum (1435 m), and weekly cumulative precipitation values, represented as bars. Similar patterns were observed in most wells, and only TPL5 was graphed to serve as a representative example. Almost every spike in the water level graph is mirrored by a rain event. This direct relationship implies increased water levels in the wetland during the spring and summer are predominantly driven by precipitation. Discrepancies in this general relationship, as was discussed in the Wetland Volume results section, may be due to the fact that precipitation events in Montana can be highly localized. Rain events on the MSU campus where precipitation data were recorded may have different intensities and durations, or not happen at all, at Story Mill.

Interestingly, the magnitude of most groundwater level spikes is far greater than the amount of precipitation that fell (notice the difference in units, and that groundwater levels are reported as stage height and must multiplied by porosity = 0.4 to achieve true water height). It is possible that this is another issue of short circuiting. Wells were capped with bentonite clay, intended to be an impermeable surface above the silica sand casing. If any gaps formed between the clay cap and the well, essentially a funnel is made, collecting a greater area of rain and delivering it directly into the well. Assuming the cap has a 15 cm radius, this could potentially account for a 36 cm rise in a groundwater well, during a 1 cm storm. This artificially large spike causes a steep gradient out of the well, accounting for the immediate rapid depletion.

More general drawdown patterns in Figure 43 are also indicative of ET. The steep decrease in stage from June to July 2015 occurred during the height of the growing season, showing the system was losing approximately 0.8 cm/day - compared to the 0.2 cm/day previously discussed in late September 2015. This is only slightly greater than ET predicted by the Montana Irrigation Guide of 0.5 cm/day.

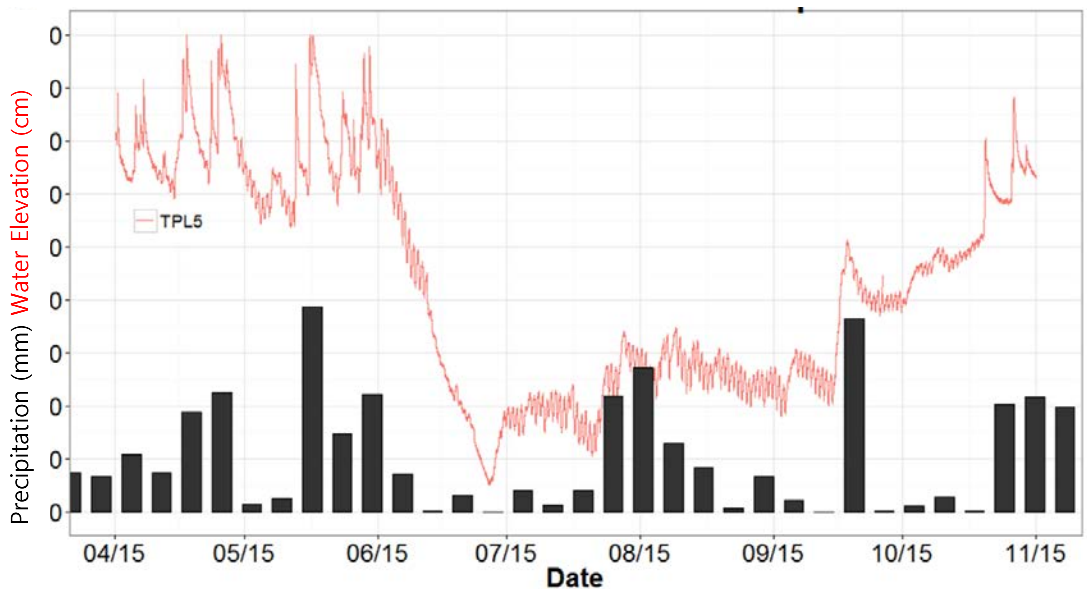

Figure 43. High resolution water level data in red, plotted with precipitation events as black bars. Precipitation was plotted as weekly totals. Water levels in TPL5 was used to represent the general relationship between patterns in water height and precipitation events. Water elevation is reported as stage height above a datum = 1435 m; to compare directly to precipitation, water elevation must be multiplied by soil porosity = 0.4.

Groundwater Velocity

Groundwater velocity (seepage velocity) was calculated to estimate the magnitude of groundwater-surface water interaction. Average site hydraulic conductivity, increased by an order of magnitude, was calculated to be 5.0E-5 m/s (4.3 m/day). This value is greater than the hydraulic conductivity estimated by the NRCS for the upper soil horizons at Story Mill (to a depth of 60 cm), and consistent with values estimated for lower, sandier horizons (60-150 cm) (California Soil Resource Lab). The value used for the analysis is over an order of magnitude faster than the hydraulic conductivity found in the literature for silty clay sands (Domenico, 1990; Heath, 1983). The site investigation of the Idaho Pole Plant showed larger hydraulic conductivities (MultiTech Services, 1992). Although the site is near to Story Mill (approximately 1.6 km to the southeast), and both have similar soil classifications, the topographic gradient at Story Mill is shallower, and soil heterogeneity is to be expected. A graph of K_{sat} values is shown in Figure 44, comparing those measured at the site to values found in the literature, from the nearby Idaho Pole Plant investigation, and reported by the NRCS. The average gradient during the sampling period was 0.01, and porosity was assumed to be 0.4. The resulting seepage velocity was very slow, at 1.3E-6 m/s (0.11 m/d). At this velocity, the hydraulic residence time, or the time it takes water entering the site from the south to travel the length of the wetland and exit into the East Gallatin River to the north, is approximately 12.5 years.

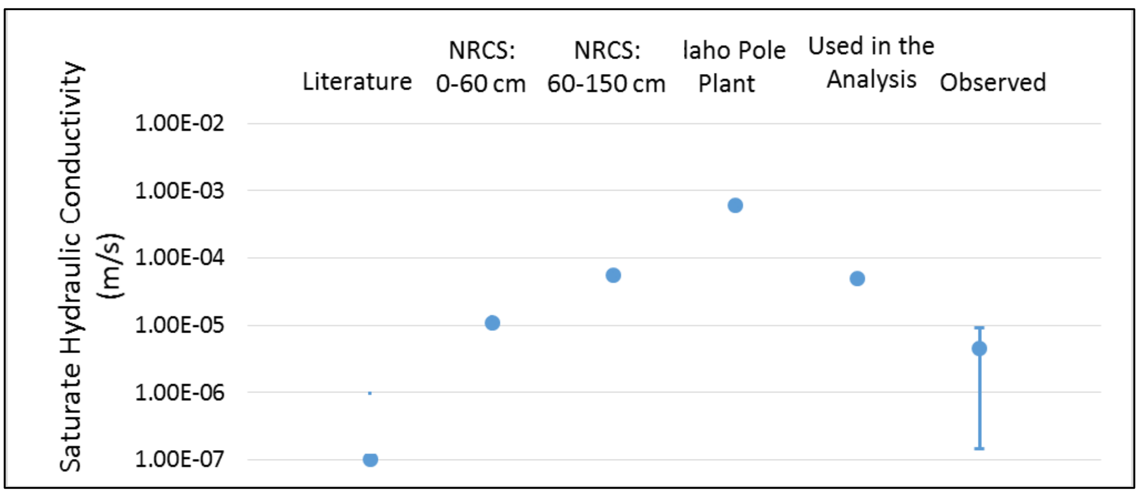

Figure 44. Plotted hydraulic conductivity values measured at Story Mill, and found in the literature. References by column heading: Literature (Domenico, 1990; Heath, 1983), NRCS (California Soil Resource Lab), Idaho Pole Plant (MultiTech Services, 1992).

Water Quality

Several important values for each parameter are summarized in Table 5, including site averages, maximums, and the location of persistently high values, relevant p-values and estimates.

Groundwater							
	TN	NO_3	TP	PO_4	SO_4	DO[1]	pH[2]
Number of samples (n)	126	387	126	387	387	331	342
Detection Limit	0.06	0.05	0.05	0.1	0.5	0	NA
% of samples below detection	2%	74%	94%	99%	0%	NA	NA
Mean (mg/L)	0.73	0.13	0.04	0.07	8.61	1.01	7.3
Standard deviation (mg/L)	0.62	0.31	0.03	0.01	6.11	1.39	0.25
Maximum (mg/L)	3.01	3.71	0.26	0.22	42.1	9	8.85
Location of highest value	Pond	TPL8	MSU2	MSU2	TPL4	TPL14	Pond
Location of the most instances of high values	MSU2	MSU2	MSU2	NA	MSU1	TPL14	Pond
Trend in time p-value	0.262	0.0421	0.557	0.822	0.307	<0.001	0.117
Estimate of annual change (mg/L)		0.10				-0.95	
Surface Water							
Bozeman Creek							
Number of samples (n)	26	61	26	61	61	52	52
% of samples below detection	0%	0%	100%	93%	0%	NA	NA
MT-DEQ Target	0.27		0.08				
Mean	0.91	0.48	0.04	0.08	2.26	9.10	7.68
Standard deviation	0.22	0.27	0.00	0.02	0.50	1.17	0.23
Maximum	1.33	1.15	0.04	0.19	3.56	11.50	8.44
East Gallatin River (before the confluence)							
Number of samples (n)	25	48	25	48	48	40	39
% of samples below detection	0%	54%	100%	92%	0%	NA	NA
MT-DEQ Target	0.30		0.03				
Mean	0.36	0.21	0.04	0.08	8.56	9.58	7.78
Standard deviation	0.17	0.16	0.00	0.03	2.42	1.21	0.26
Maximum	0.74	0.74	0.04	0.17	11.79	12.10	8.55
East Gallatin River (after the Confluence)							
Number of samples (n)	13	29	13	29	29	25	23
% of samples below detection	0%	14%	100%	97%	0%	NA	NA
MT-DEQ Target	0.29		0.05				
Mean	0.56	0.27	0.04	0.07	5.40	9.15	7.80
Standard deviation	0.23	0.21	0.00	0.01	1.14	1.21	0.27
Maximum	0.95	0.73	0.04	0.14	7.75	11.80	8.54

[1] the pond was not used in the DO analysis
[2] values reported as pH

Table 5. Summary of water quality parameter results. Each compound is reported in mg/L as its element, nitrogen, phosphorus, or sulfur.

The results of the water quality analyses are presented in a consistent manner for each of the seven parameters: total nitrogen, nitrate, total phosphorus, orthophosphate, sulfate, dissolved oxygen, and pH. Each compound is reported in mg/L as its element, nitrogen, phosphorus, or sulfur; the conversion is show below:

$$[Element] = [Compound] * \frac{M_{Element}}{M_{Compound}}$$

where [ ] denotes concentration, M = molecular weight, and x = mole fraction (moles of element in a mole of compound). Results are organized in a way that facilitates discussion based on both statistical and anecdotal observations. A similar graphic presentation of results is presented for each parameter: two series of panels that collectively illustrate temporal and spatial trends for the site as a whole, and allow focus on short-term or local phenomena that may have been influential in the analysis, or point to some physical occurrence of note. Figures 45 (A - E) and 46 (F - I) are for total nitrogen, but are referenced here to point out the intent of these figures which are consistently portrayed for each water quality parameter studied.

Figure 45 (A-E) is an example of the presentation of the temporal nature of a particular parameter from either the wells or the surface waters. All the observations of concentrations from groundwater wells for a particular compound are shown in panel A, with triangular symbols representing wells in the southern portion of the study area and circular symbols representing wells in the northern portion. The solid line represents the mean of all samples. Panel B shows the time sequence of concentration for only those wells that had a statistically significant relationship ($P<0.05$) between concentration and time (either decreasing or increasing with time). Panel C shows data from wells that were consistent with time, never having concentrations substantially different from the temporal mean. Panel D shows the concentrations from wells that had one or more occurrence of an observed concentration that deviated substantially from the mean – that is, wells that demonstrated more erratic concentrations than those of panel C. Finally, panel E shows the concentrations observed in the surface water samples.

Figure 46 (F - G) is an example of the way that spatial relationships are presented. Surface maps were constructed for each sample days showing interpolated (kriged) lines of constant concentration for all seven parameters. Because there were more than 40 sample days, this resulted in a total of approximately 250 surface maps. For brevity, however, only three surface maps are presented for each parameter. After discounting any surface map that was heavily influenced by data sparseness (dry wells, frozen wells, interference by construction activity, etc.), it was clear that for any compound there was no large-scale diversity in the maps. So, for each compound three maps were chosen for display to show surfaces typical over the duration of the study, and to demonstrate that even without having large scale diversity in the response surfaces, there were occasionally features in the surface maps of note. Panel I helps to visualize wells with occurrences of relatively high concentrations at one or more points in the time sequence. In this panel the frequency of relatively high concentrations is indicated by varying symbol size: extra big, big, medium, and small, representing more than 10, 5 to 10, 3 to 4, and less than 3 occurrences of notably high concentrations, respectively.

<u>Total Nitrogen</u>

Total nitrogen levels in the Story Mill groundwater averaged 0.73 mg/L, and by simple linear regression, concentrations did not exhibit a significant trend over the 12-month observation period (p=0.262). Of the 126 samples, the majority of the results fell under 1 mg/L, with only 30 values greater than this threshold, peaking at a maximum value of 3.01 mg/L in samples taken from the Pond on 9/25/2014, and from MSU2 on 7/23/2015.

Higher TN concentrations were most common in MSU2, with 5 occurrences above 1 mg/L. MSU2 showed a statistically significant increase in TN concentration over time. This trend is especially strong starting in January of 2015. TPL10, although it was only above 1 mg/L once, also showed a statistically significant increase in TN concentration, interestingly during the same period as MSU2. A handful of other wells also spiked above 1 mg/L TN, with most occurrences in the southern extent of the property. Higher concentrations of TN occurred in different wells throughout the year, with no obvious temporal trend. TN concentrations in several wells, both ones that spiked, and ones that remained below 1 mg/L TN, decreased in the fall, increased throughout the winter, and started decreasing again in the spring.

When wells were grouped by location (south or north), it was determined that the parallel lines model was the best fit. Total nitrogen concentrations were strongly dependent on location (p-value <0.001), but concentrations were not dependent on time, both as a whole (p-value = 0.526) and when grouped by location (p-value = 0.219). TN concentrations in the south were more erratic and were on average higher than TN concentrations in the north, by approximately 0.39 mg/L. By this model, if the location of the well is given (either being in the north or south), the TN concentrations can be estimated and at any time by the equation:

$$\mu\{[Nitrogen]|location\} = 0.73\ mg/L\ {}^{+}/{-}\ (0.195\ mg/L * location)$$

where (+) is used for locations in the south, and (–) is used for locations in the north.

Total nitrogen levels in the creeks ranged from non-detectable to 1.33 mg/L, and averaged 0.62 mg/L. Bozeman Creek, on average measured 0.55 mg/L higher in TN than the East Gallatin River before the confluence. After the confluence, the mixing of the more polluted Bozeman Creek is evident as the East Gallatin River became more elevated in TN. In each creek, samples taken from upstream and downstream of the Story Mill wetland were very similar, and there was no consistency in "influent" water being more or less polluted than "effluent" water. All four sampling locations climbed to their most elevated levels through November and December 2014, and gradually decrease again through the spring until May 2015. Bozeman Creek spiked again in April 2015, while TN in the East Gallatin River, both before and after the confluence, remained low. Interestingly, this is the opposite pattern as was observed for groundwater TN.

The surfaces shown display the more persistently elevated TN concentrations in the south. The pond always measured above 0.93 mg/L TN, and was particularly high on 9/25/2014, shown in Surface F. Typically, lower levels were observed in the meadows to the north, and in the meadow to the west of the pond, depicted in Surface G, on 12/14/2014. MSU2 spiked the most often, an example of which is shown in Surface H of

3/19/2015. This was also a date when several other wells throughout the wetland also measured above 1 mg/L.

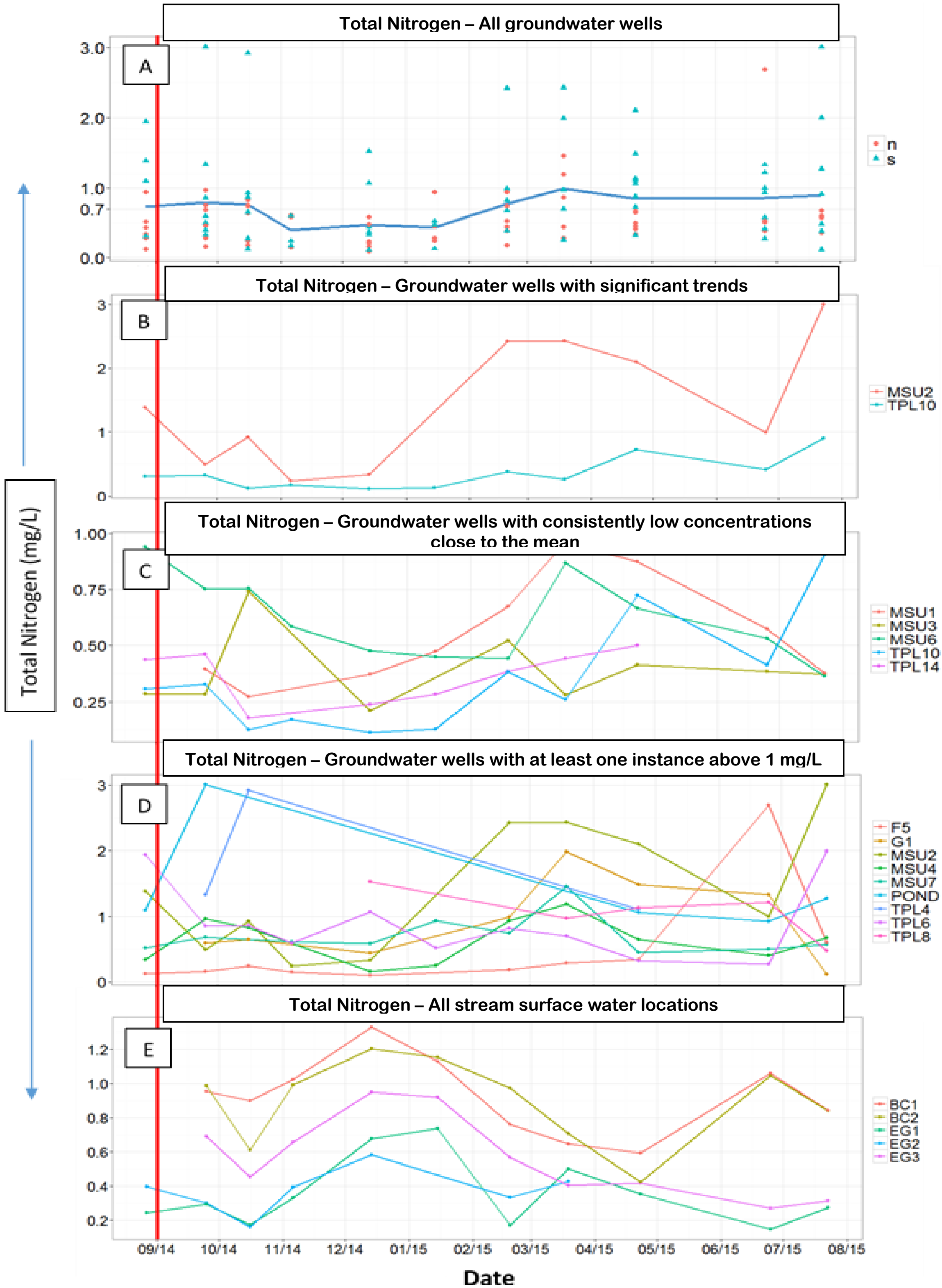

Figure 45. Total nitrogen time series graphs. All samples that measured below detection were plotted as 0.04 mg/L. Potential error = 0.05 mg/L. *Reported as nitrogen.

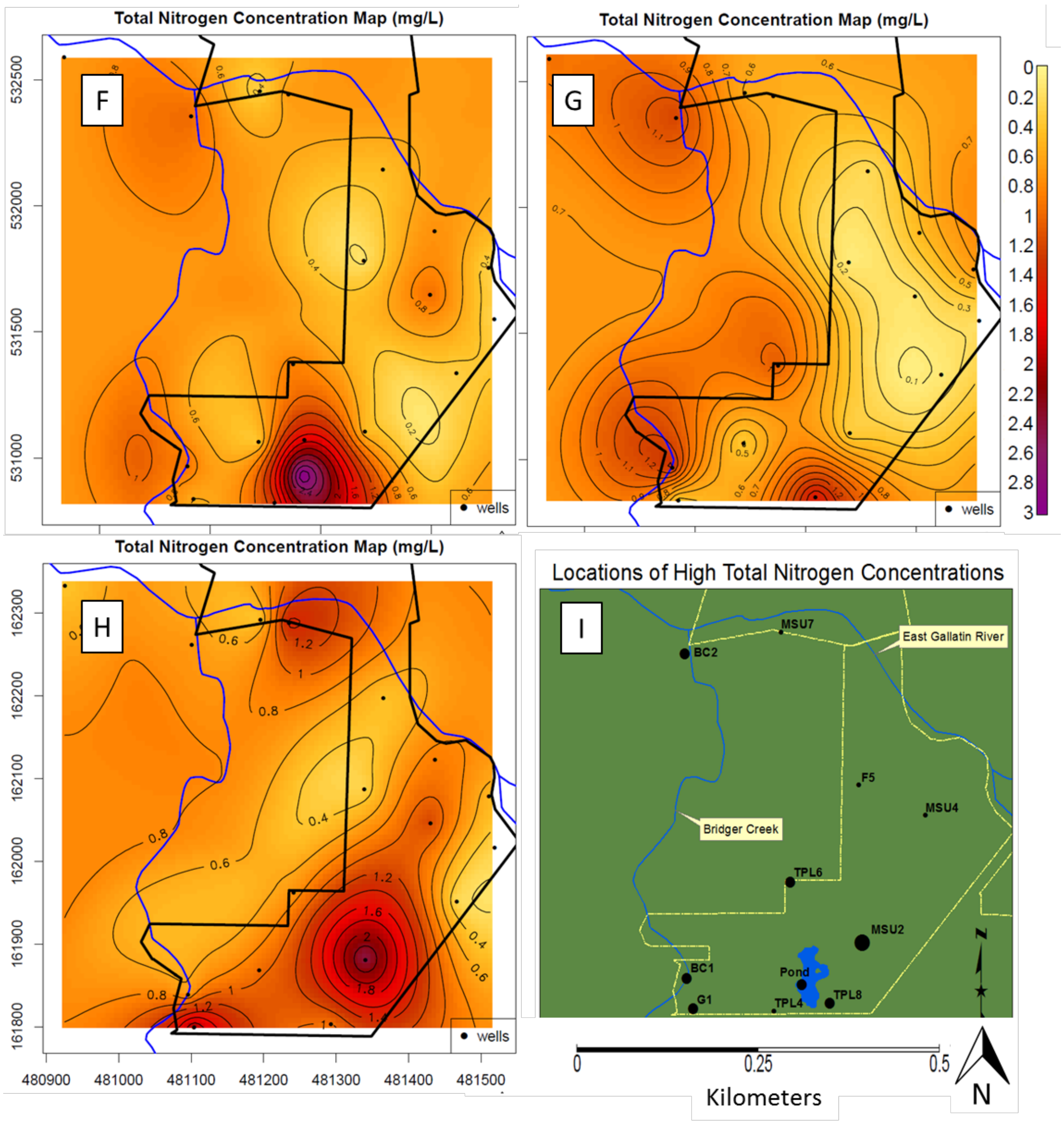

Figure 46. Total nitrogen surfaces and map of occurrence above 1 mg/L. All samples that measured below detection were plotted as 0.04 mg/L. Potential error = 0.05 mg/L. Clockwise from the top: Image F, 9/25/2014, Image G, 12/14/2014, Image H, 3/19/2015. *Reported as nitrogen.

<u>Nitrate</u>

There is moderate evidence to suggest that nitrate concentrations increased during the monitoring period by an average of 0.1 mg/L (p-value = 0.042). A total of 387 groundwater samples were measured for nitrate, with an average concentration of 0.13 mg/L, and with only 12 values above 1 mg/L, and 74% non-detects. The maximum value measured was 3.71 mg/L in TPL8; an anomaly for TPL8, which measured only slightly above detection in three other instances. The next highest concentration was measured in MSU2, at 1.84 mg/L. Only 5 wells, F5, G1, MSU2, and MSU7 held above detection for more than 5 measurements, or with any consistency.

Several localized patterns were observed in individual wells. Two wells, MSU2 and F5, exhibited individual trends, both of which statistically increased in nitrate concentrations with time. Concentrations began to rise in both wells in April 2015, where MSU2 continued to increase progressively until the end of sampling in August 2015, and F5 stepped back down. Other patterns such as the coincident oscillations observed for TN, were not as apparent for nitrate. G1, like F5, also followed a stepwise increase and subsequent decrease in nitrate concentrations, but began about two months earlier in February 2015. Towards the end of the monitoring period, a higher percentage of wells measured above detection nitrate levels. Spikes in nitrate concentrations were rarely coincident in time between several wells, and occurred throughout the year with no clear pattern.

It was also found, as with TN, that nitrate concentrations best fit a parallel lines model, with a moderately statistically significant difference between concentrations found in wells located in the north vs. the south (p-value = 0.05). All wells with at least one occurrence above 1 mg/L, were located in the south, and that on average northern wells were 0.13 mg/L lower in nitrate than southern wells. By this model, it was also found that concentrations were not dependent on time, both as a whole (p-value = 0.128) and when grouped by location (p-value = 0.773). Nitrate concentrations can be estimated by the equation:

$$\mu\{[Nitrate]|time, location\} = 0.13\ mg/L\ {}^{+}/_{-}\ (0.065\ mg/L * location)$$

where (+) is used for locations in the south, and (–) is used for locations in the north.

Nitrate levels in the creeks had a similar range and followed a similar pattern as TN levels. Bozeman Creek was persistently higher in nitrate than the East Gallatin River, and showed no increase or decrease in concentration along the length of the property. The East Gallatin River also showed no change in concentration after it flowed past Story Mill, but notably increased after the confluence of the more polluted Bozeman Creek. Throughout the sampling period, nitrate concentrations first decreased in the fall, then increased at the inception of winter and remained elevated throughout February 2015, and gradually decreased into the spring. For the remainder of the sampling period EG1 measured below detection, whereas Bozeman Creek spiked again in July 2015.

The surfaces shown display the persistently low nitrate concentrations throughout most of the Story Mill site. Surfaces F of 12/14/2014, and G of 4/9/2015, show Bozeman Creek's tendency to be higher in nitrates than the groundwater, and Bozeman Creek's influence on the East Gallatin River after the confluence. Surface F is from a day (12/14/2014) when nitrate levels in the creeks were highest, and Surface H is from a day (5/28/2015) when levels in the creeks were lowest. The latter is from a day towards the end of sampling where several wells that were typically non-detects within the wetland registered measurable concentrations, and MSU2 and F5 and G1 where all above 0.25

mg/L. Although several wells in the south spiked, they were rarely coincident, which created the occasional bulls eye of high concentration in the south.

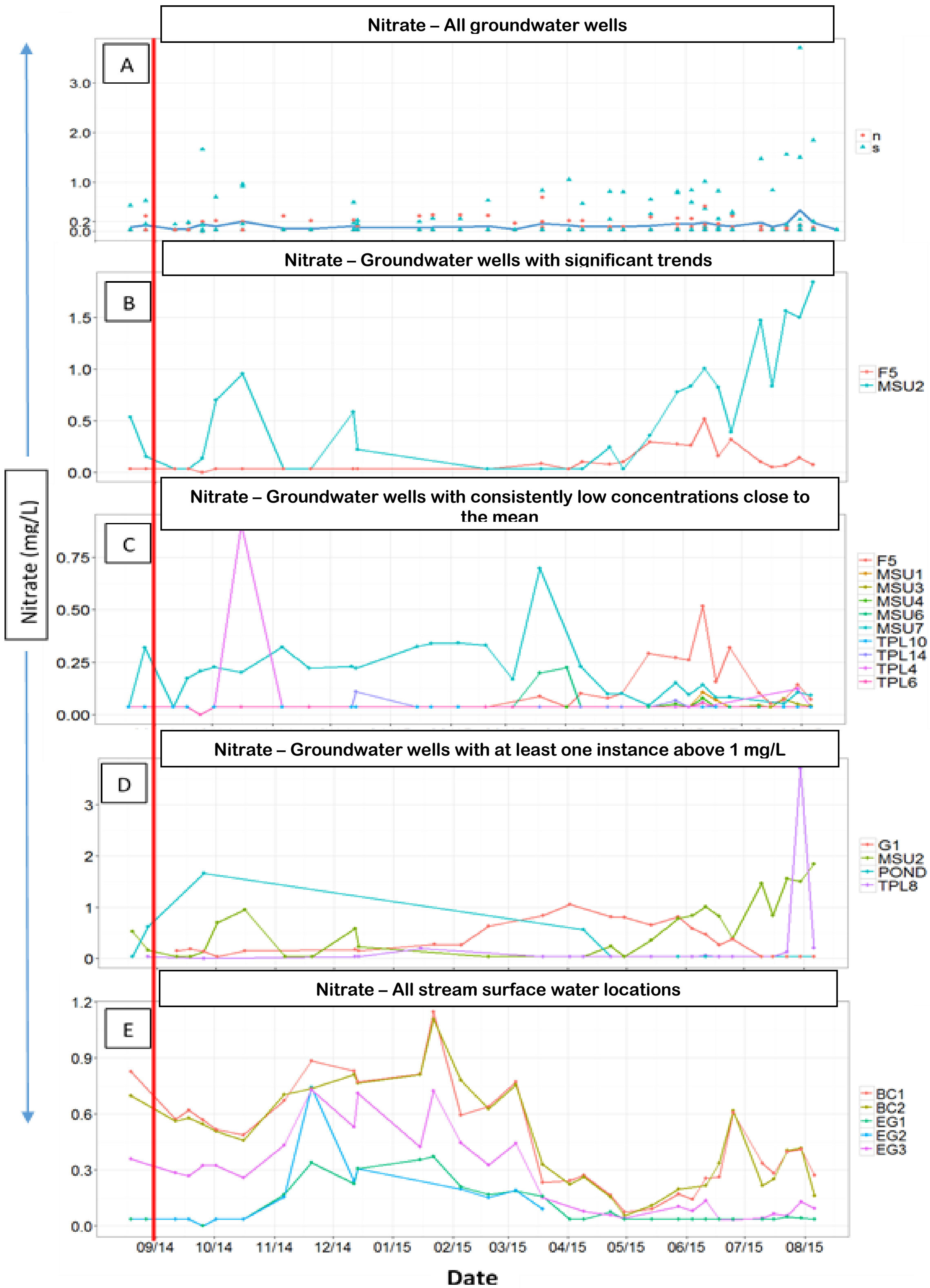

Figure 47. Nitrate time series graphs. All samples that measured below detection were plotted as 0.04 mg/L. Potential error = 0.02 mg/L. *Reported as nitrogen.

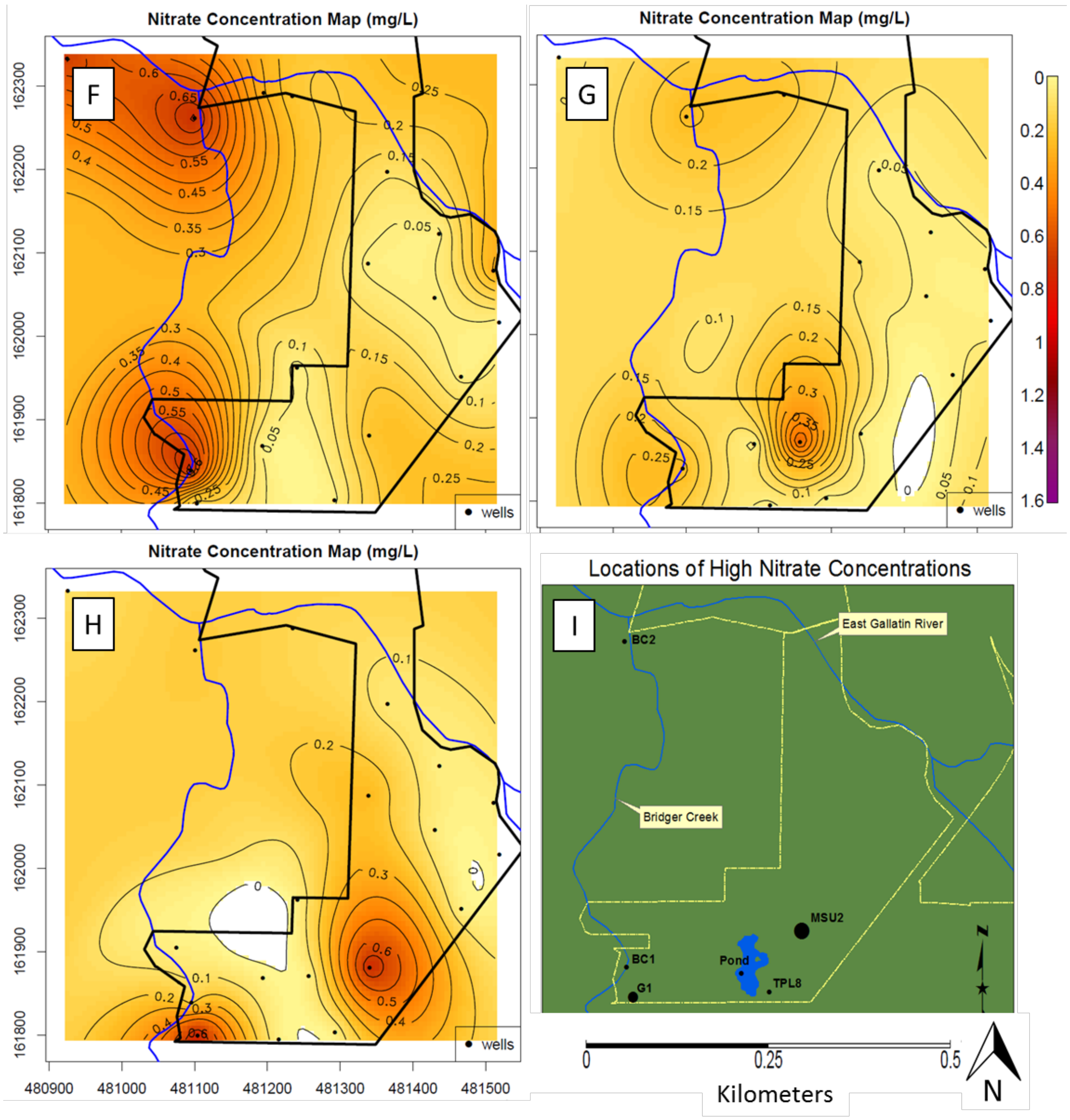

Figure 48. Nitrate surfaces and map of occurrence above 1 mg/L. All samples that measured below detection were plotted as 0.04 mg/L. Potential error = 0.02 mg/L. Clockwise from the top: Image F, 12/14/2014, Image G, 4/9/2015, Image H, 5/28/2015. *Reported as nitrogen.

Total Phosphorus

Total phosphorus was measured in 126 groundwater samples, only seven of which measured above detection, occurring in six different wells. MSU2 had the highest value of 0.26 mg/L, and was the only well with more than one value above detection. When data from the site as a whole were assessed, there was no significant trend over time (p-value=0.557). Almost all spikes that occurred were between mid-January and mid-March 2015, except for F5, which spiked in June 2015. Other than these occasional spikes, there was no statistically significant trends or variability in the data. Predictably, due to the scarcity of measurable data, when parallel and separate lines models were fitted, there was no difference between TP concentrations in wells located in the south vs. the north (p-value = 0.317), and concentrations did not change significantly with time (p-value = 0.645). No detectable levels of TP were measured in the creeks.

Samples with measureable TP were collected on only four dates, equally limiting the number of surfaces created. The surfaces shown, 1/15/2015 (F), 2/19/2015 (G), and 3/19/2015 (H), display these temporary, localized hot-spots, and the tendency for the surface water and groundwater to have very low TP concentrations. Instances of measurable TP occurred in locations across the site, and were not centralized in one area.

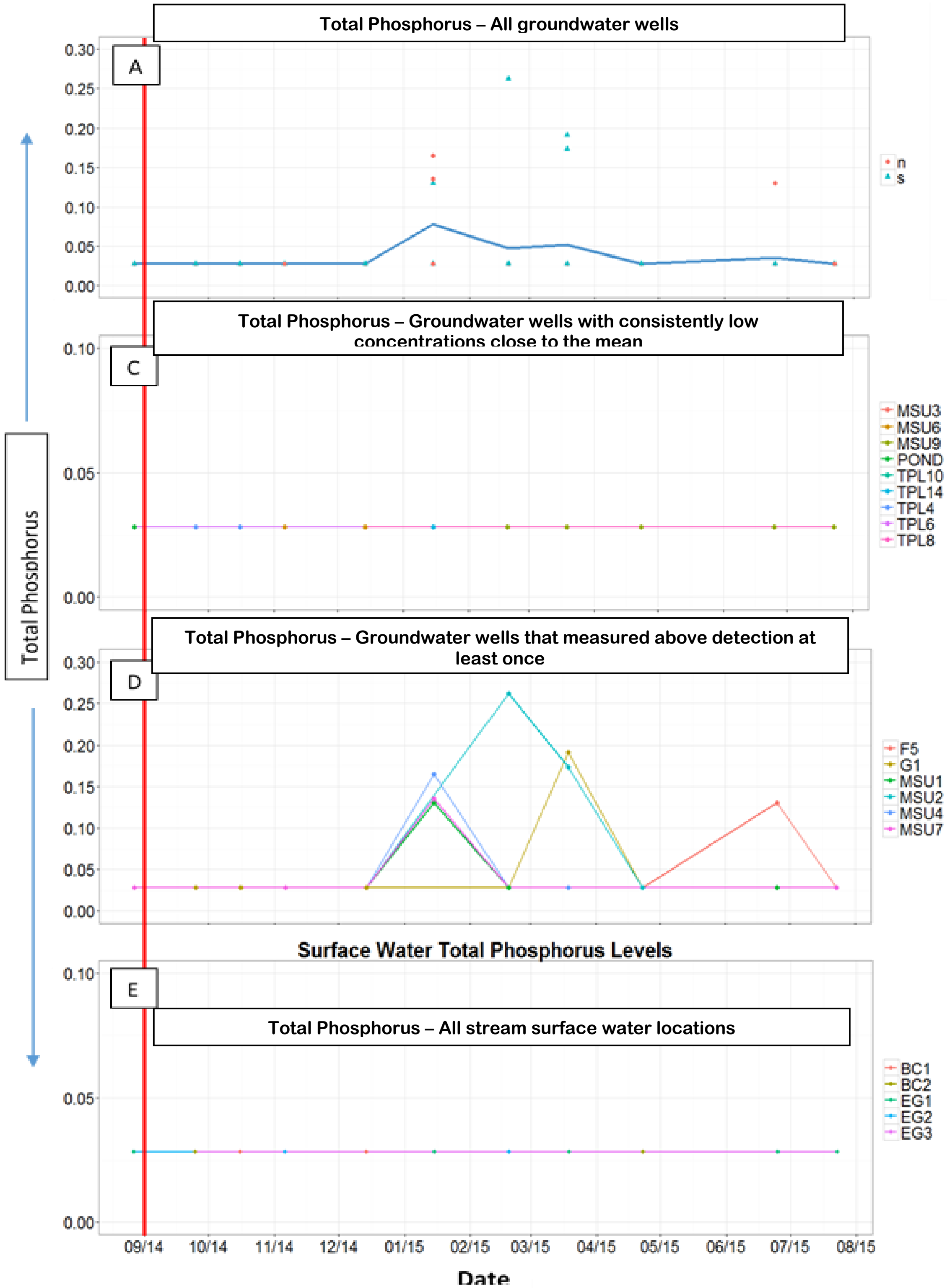

Figure 49. Total phosphorus time series graphs. All samples that measured below detection were plotted as 0.03 mg/L. Potential error = 0.04 mg/L. No wells exhibited significant trends, so this panel was left out. *Reported as phosphorus.

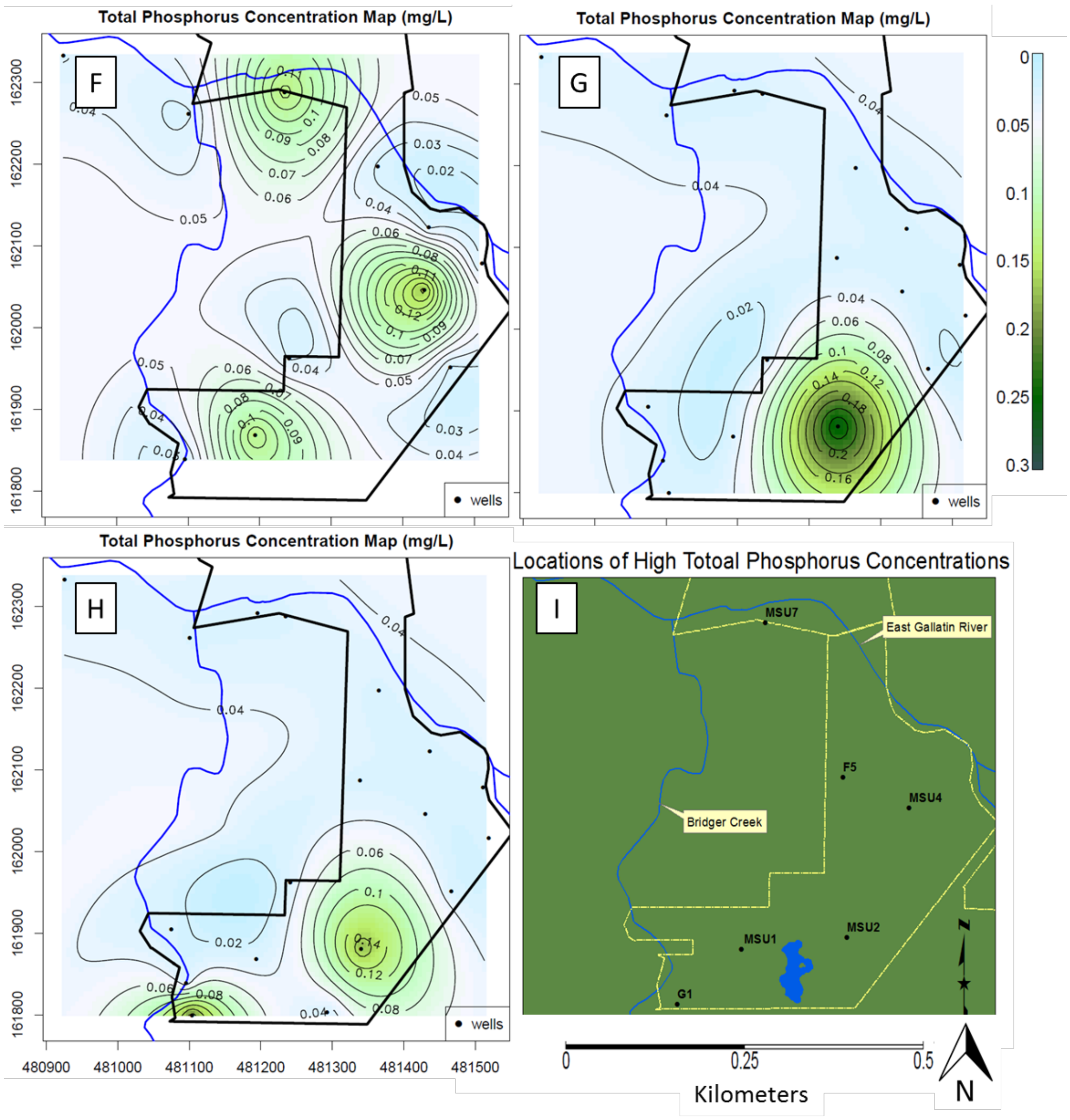

Figure 50. Total Phosphorus surfaces and map of occurrence above detection. All samples that measured below detection were plotted as 0.03 mg/L. Potential error = 0.04 mg/L. Clockwise from the top: Image F, 1/15/2015, Image G, 2/19/2015, Image H, 3/19/2015. *Reported as phosphorus.

Orthophosphate

Orthophosphate was analyzed in 387 groundwater samples, and only measured above the detection limit twice. The first instance was in MSU9 on 3/5/2015 (G), measuring 0.22 mg/L, and then in MSU2 on 4/2/2015 (H), measuring 0.17 mg/L. Similarly as with TP, with such a consistent set of data, fitting a linear model is not relevant: concentrations were universally low across the site for the duration of the monitoring period. Surface water samples measured above detection on only two occasions, on 2/19/2015 (F) and 3/5/2015 (G). Although TP was not measured from samples collected on these specific dates, orthophosphate registered above detection during the same period in the winter when TP spiked in several wells.

Orthophosphate in the creeks measured above detection collectively six times, and at least once in each of the five locations. All detectable levels were measured between February and April 2015, during the same time period as the only two detectable orthophosphate levels in the groundwater were measured. All surface water locations spiked to similar concentrations, between 0.14 and 0.19 mg/L.

Three surfaces were created for the three dates with measurable orthophosphate values, again illustrating the pervasively low phosphorus associated with the water at Story Mill.

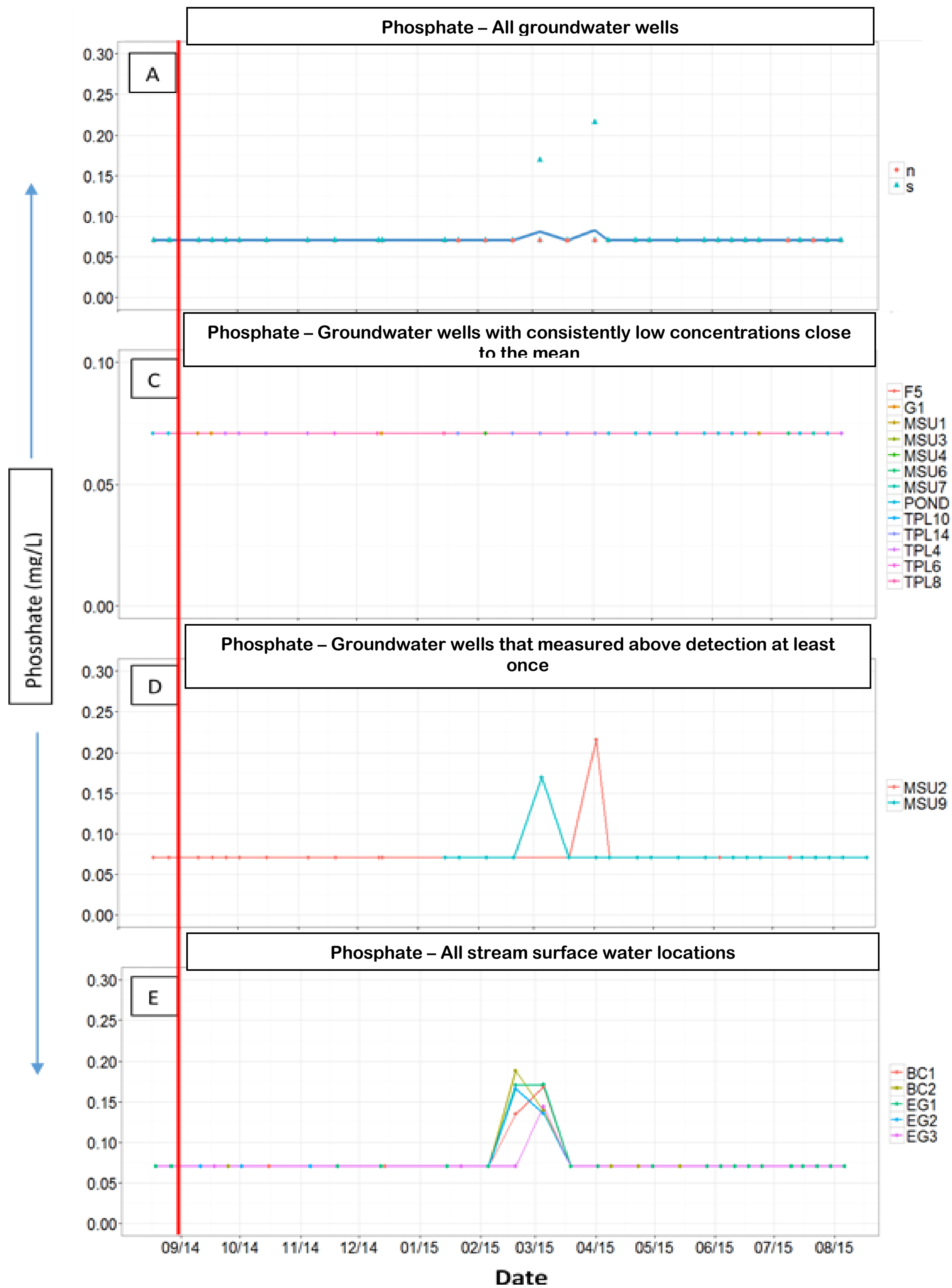

Figure 51. Orthophosphate time series graphs. All samples that measured below detection were plotted as 0.07 mg/L. Potential error = 0.05 mg/L. No wells exhibited significant trends, so this panel was left out. *Reported as phosphorus.

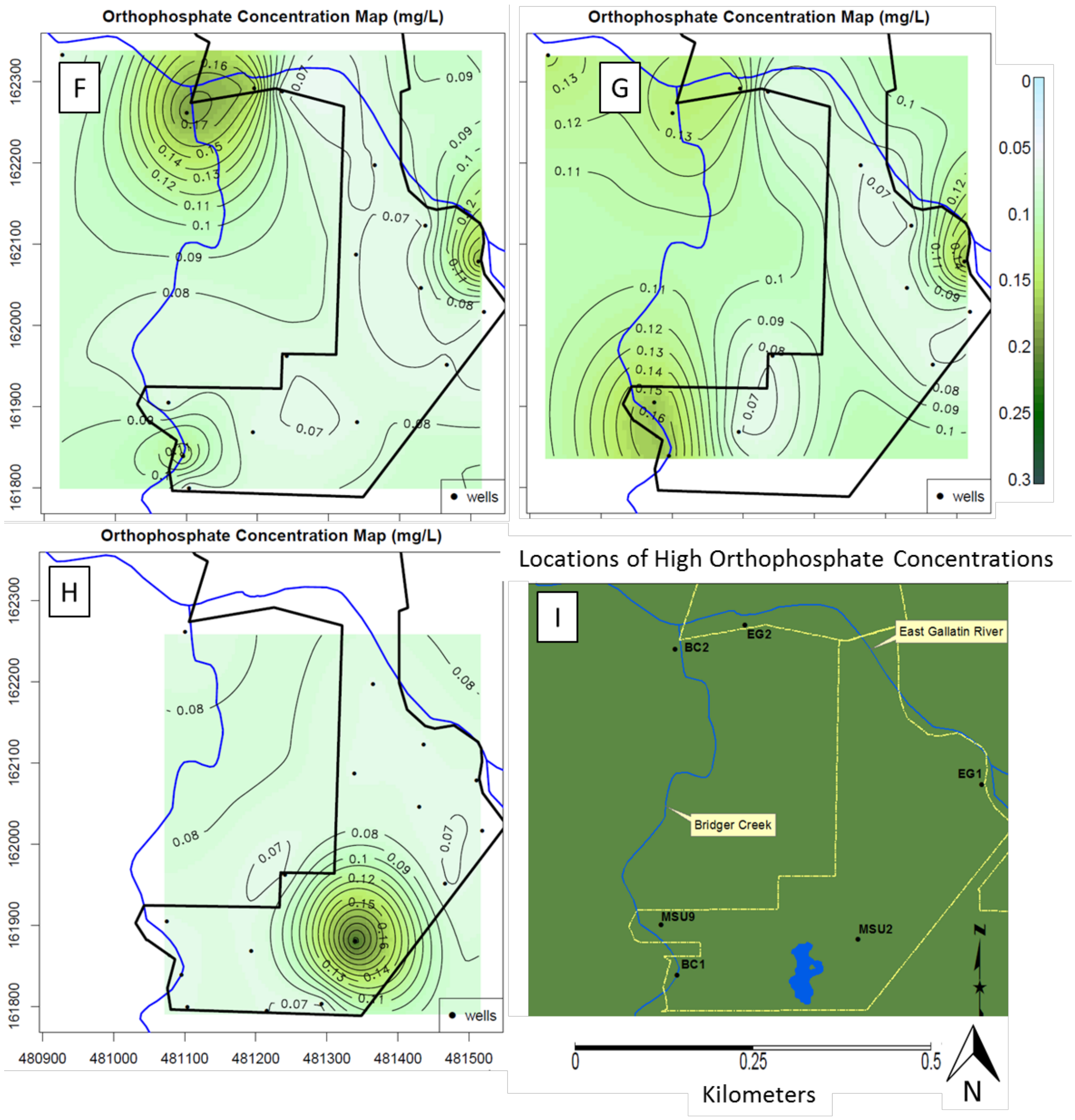

Figure 52. Orthophosphate surfaces and map of occurrence above detection. All samples that measured below detection were plotted as 0.07 mg/L. Potential error = 0.05 mg/L. Clockwise from the top: Image F, 2/19/15, Image G, 3/5/15, Image H, 4/2/2015. *Reported as phosphorus.

Sulfate

Sulfate was measurable in all 387 collected groundwater samples, and ranged from 0.57 mg/L to 42.1 mg/L, with an average of 8.61 mg/L. Overall, the groundwater at

Story Mill did not exhibit a statistically significant trend in sulfate concentration over time (p-value = 0.307). Individually, only MSU4 showed a significant trend: after an initial spike sulfate levels progressively decreased from 17.76 mg/L to 3.37 mg/L, and leveled off around 8 mg/L in mid-May for the duration of the monitoring. Four wells, MSU1, MSU2, TPL4, and TPL8, measured sulfate levels greater than 20 mg/L, all of which tended to have higher than average values with some regularity, instead of seemingly random spikes. TPL6, on the other hand, stands out as having had the lowest and least variable sulfate levels, and averaged 3 mg/L. Overall, wells in the south were more erratic, ranging the full span of sulfate values measured, whereas the wells in the north seemed to be more stable and clustered more closely around the average.

Sulfate levels in the creeks were relatively low compared to those found in the groundwater, never occurring above 12 mg/L. The East Gallatin before the confluence of Bozeman Creek had the highest concentrations. Bozeman Creek had the lowest concentrations, which likely diluted the concentrations in the East Gallatin, where after the confluence, EG3 measured on average 5.4 mg/L throughout the year. Bozeman Creek, both in BC1 and BC2, remained nearly constant around 2 mg/L for the duration of the project. EG1 and EG2 measured around 10 mg/L until April 2015, when it decreased and fluctuated between 8 and 2 mg/L. Sulfate concentrations in EG3, for the most part, mimicked those measured upstream in EG1 and EG2, but without the significant decrease.

The surfaces illustrate the variability that was observed in sulfate concentrations at the site. Surface F is of 9/25/2014, when the site as a whole was close to the average, with relatively elevated levels in TPL4 and MSU4. Surface G is representative of a time when sulfate levels were high during the spring of 2015, and was generated from data collected on 4/9/2015. Surface H shows the opposite, when levels were relatively low across the site on 6/4/2015.

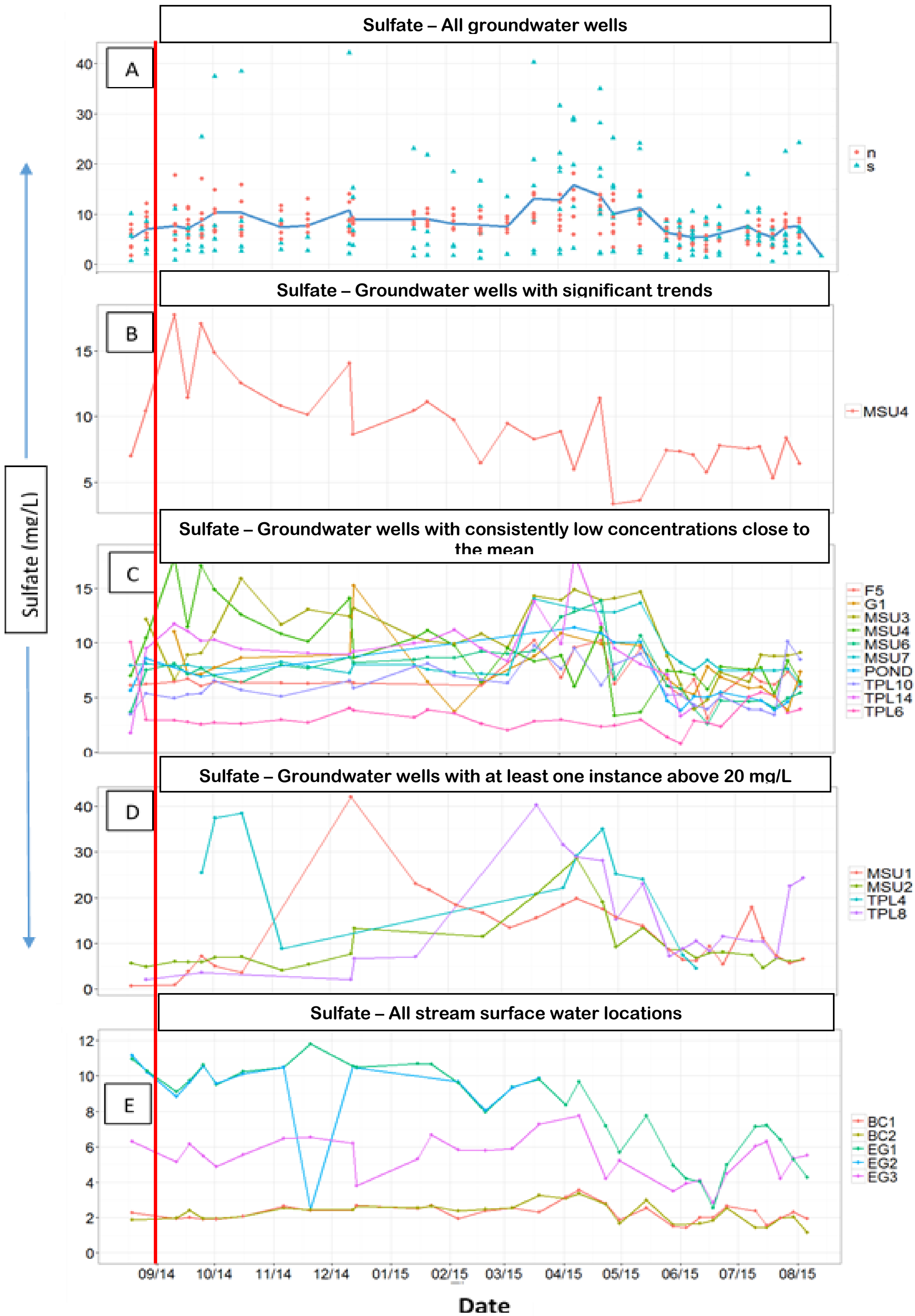

Figure 53. Sulfate time series graphs. All samples that measured below detection were plotted as 0.35 mg/L. Potential error = 1.48 mg/L. *Reported as sulfur.

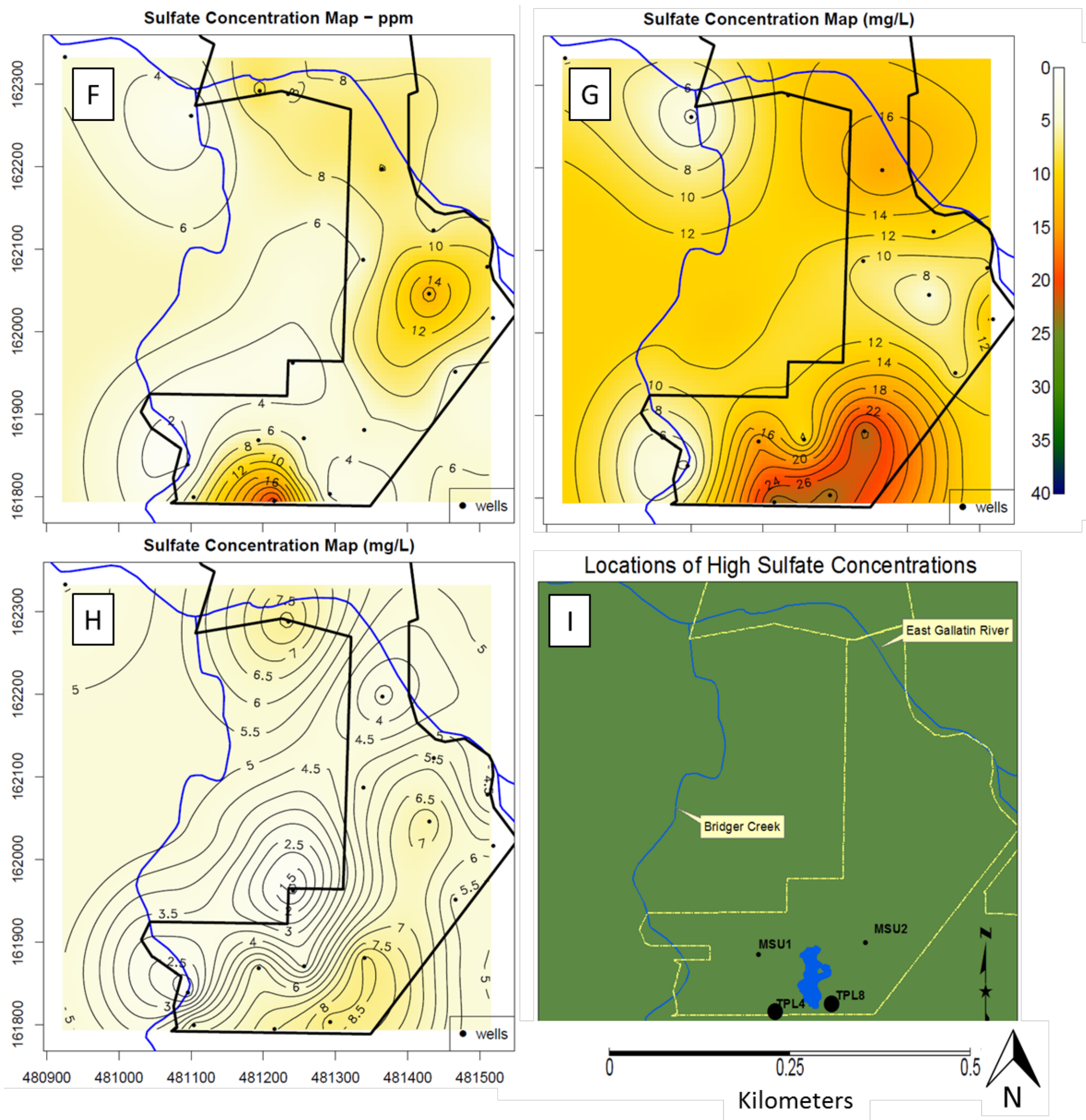

Figure 54. Sulfate surfaces and map of occurrence above 20 mg/L. All samples that measured below detection were plotted as 0.35 mg/L. Potential error = 1.48 mg/L. Clockwise from the top: Image F, 9/25/2014, Image G, 4/9/2015, Image H, 6/4/2015. *Reported as sulfur.

<u>Dissolved Oxygen</u>

Dissolved oxygen (DO) levels were overall persistently low in the groundwater environment, and showed a decreasing trend over time (p-value = <0.001). Average DO was 1.01 mg/L, with a wide range of values: almost all values remained below 2 mg/L, while TPL14 was the most notable exception, which persistently rose to levels between 5 mg/L and 9 mg/L. The overall decreasing trend is due to a group of six wells, MSU1, MSU6, TPL10, F5, and TPL14, that had higher DO early in the monitoring period during 2014, creating a cluster of data points ranging from 2.5-4.5 mg/L. Four of these wells quickly dropped to, and for the most part, remained below 2 mg/L after mid October 2014. TPL 14 on the other hand, increased again to sustain high DO levels throughout the winter, and then decreased steadily after April 2015, until it ran dry in June. There was no consistent correlation between DO and depth of water in the wells, or depth to water below the ground surface. The location of these wells is spread out over the site, and not necessarily correlated with site characteristics, except for perhaps TPL14, which may be influenced by some hyporheic flow from the East Gallatin River.

Dissolved oxygen in the creeks followed a predictable pattern, increasing in the fall as temperatures became colder, and decreasing again in the spring with the onset of warmer summer months. Levels as high as 12.1 mg/L were observed, likely due to aeration from shallow flow mixing over a rocky river bed. Dissolved oxygen concentrations in all surface water locations followed similar trends, where on any given day concentrations rarely varied more than 1 mg/L, and no location was consistently higher or lower than any other.

The surfaces shown clearly illustrate the dramatically higher DO in the creeks and pond compared to the near zero levels found in the groundwater. Surface F was created with data from 9/25/2014, during the time when several wells measured uncharacteristically high DO levels. Surface G shows a time when DO was elevated in TPL14, on 4/30/2015, and was almost as oxygenated as the surface water. Surface H depicts DO levels on 6/11/2015, when all wells measured below 1 mg/L. It can be seen that instances of high DO largely occurred on the periphery of the property, with the exception of the pond, and that the central areas of the wetland remained persistently low.

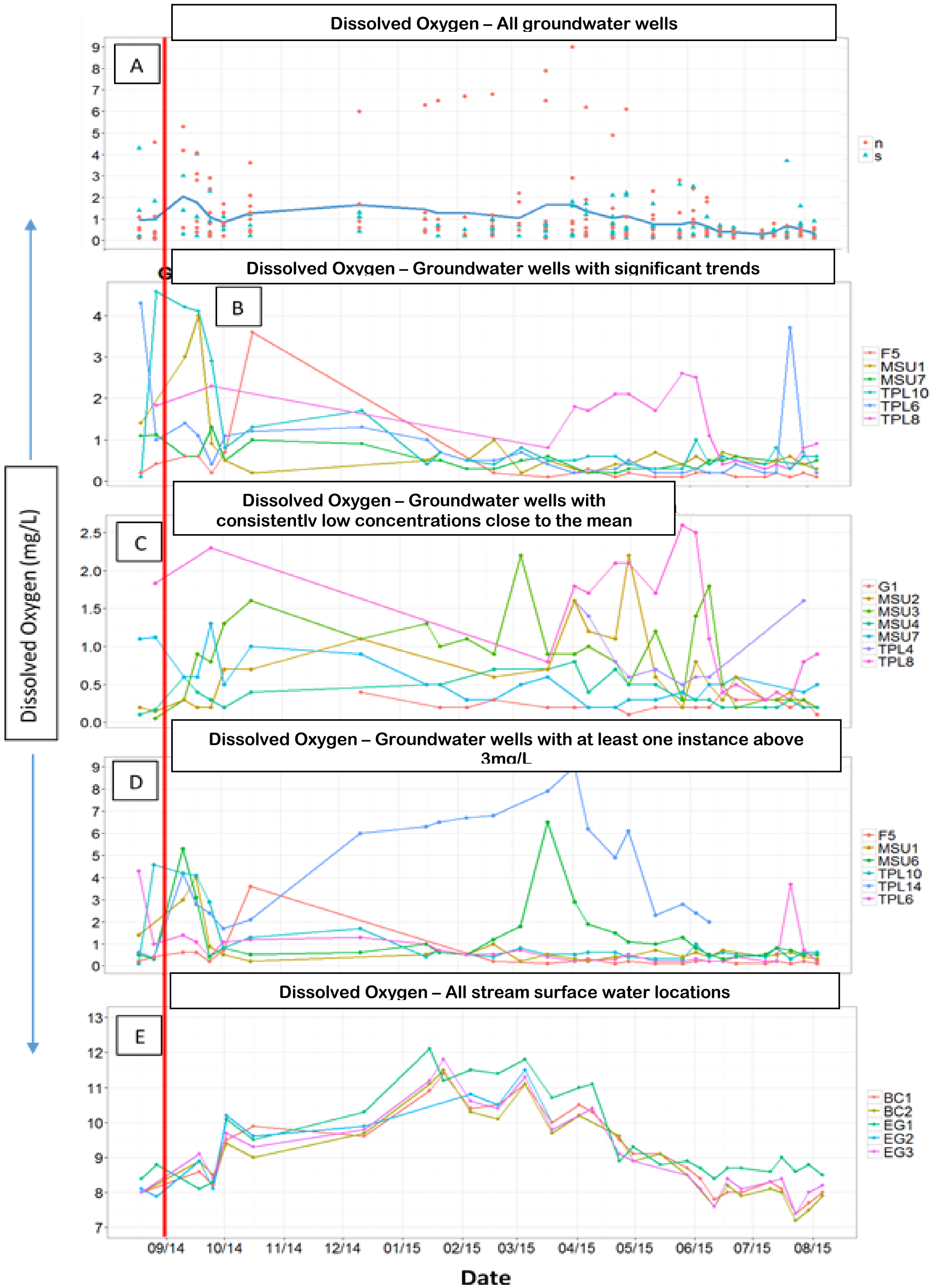

Figure 55. Dissolved oxygen time series graphs. Potential error = 0.2 mg/L.

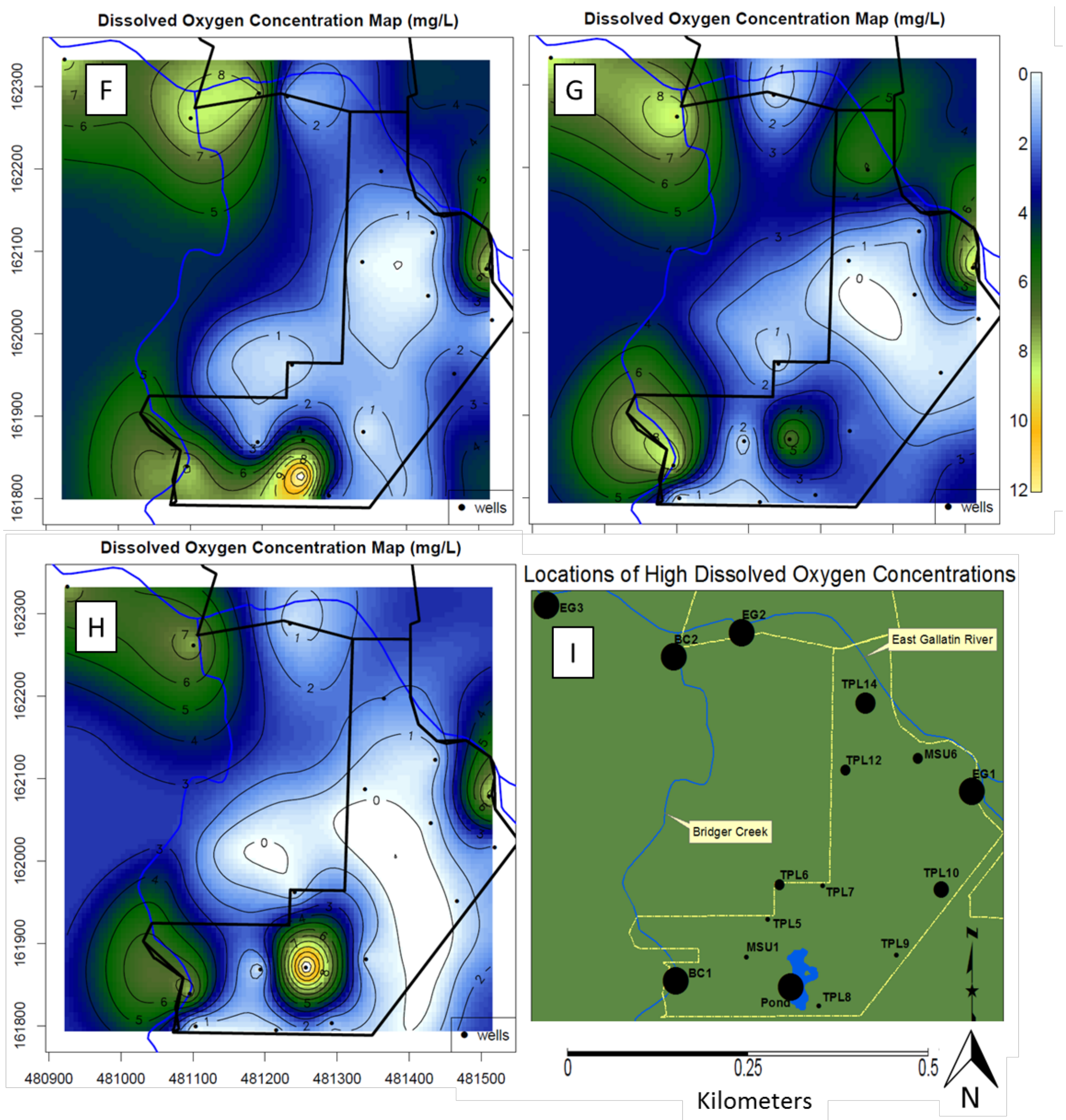

Figure 56. Dissolved oxygen surfaces and map of occurrence above 3 mg/L. Potential error = 0.2 mg/L. Clockwise from the top: Image F, 9/25/2014, Image G, 4/30/2015, Image H, 6/11/2015.

pH

Groundwater pH remained almost entirely just basic of neutral, at an average of 7.3. The Pond was the only location that measured above 8. pH in all groundwater wells, except G1 and the Pond, tracked in a similar pattern, varying as a whole between 6.76 and 8.85, and rarely differed by more than 1 for a given day. pH rose quickly in August and September 2014, dropping suddenly on October 2nd, after which it gradually increased. pH spiked again in February and March, only to quickly return down to about average, and then slowly increased by about 0.1 over the next five months for the remainder of the study. No significant trends in pH with time were observed for the site as a whole, or in any wells individually.

Surface water pH tracked almost exactly with the groundwater, but was consistently more basic. The pH of the groundwater typically remained between 7 and 7.5, whereas the surface water typically remained between 7.5 and 8, approximately averaging a 0.47 difference on any given day. No locations stood out as having consistently higher or lower pH; for the duration of the study, measurements for each sampling event were similar across all sampling locations.

The surfaces shown depict one of the most basic sampling dates on 9/25/2014 (image F), the most acidic sampling date on 10/2/2014 (image G), and a more typical surface representative of conditions near the average pH on 4/23/2015. From these images, it can be seen that the wetland environment is typically neutral, and the creeks and pond are slightly more basic. Spikes in pH tended to occur simultaneously across the site, with only three groundwater locations that were consistently more basic: G1, MSU9 and the Pond.

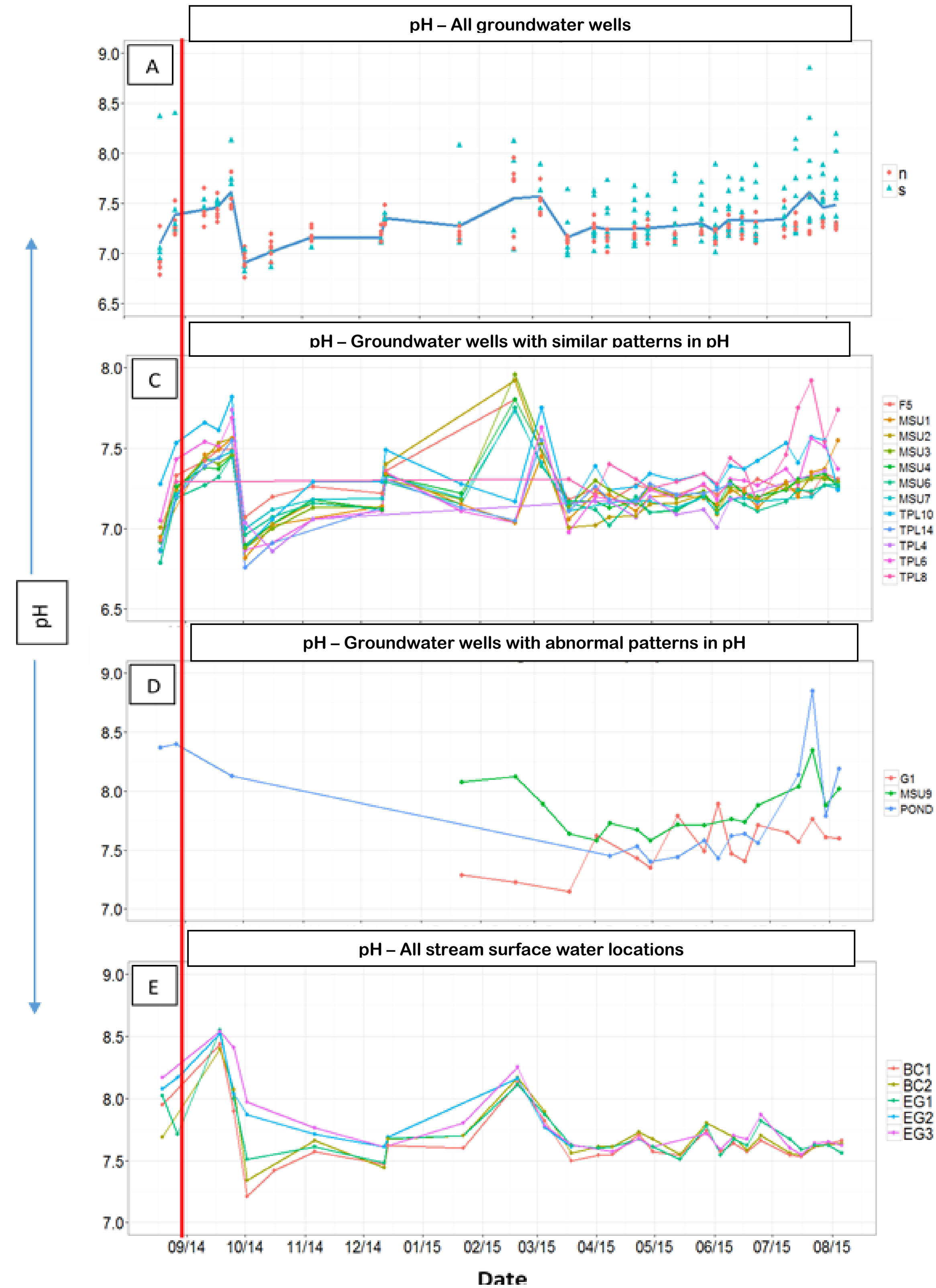

Figure 57. pH time series graphs.

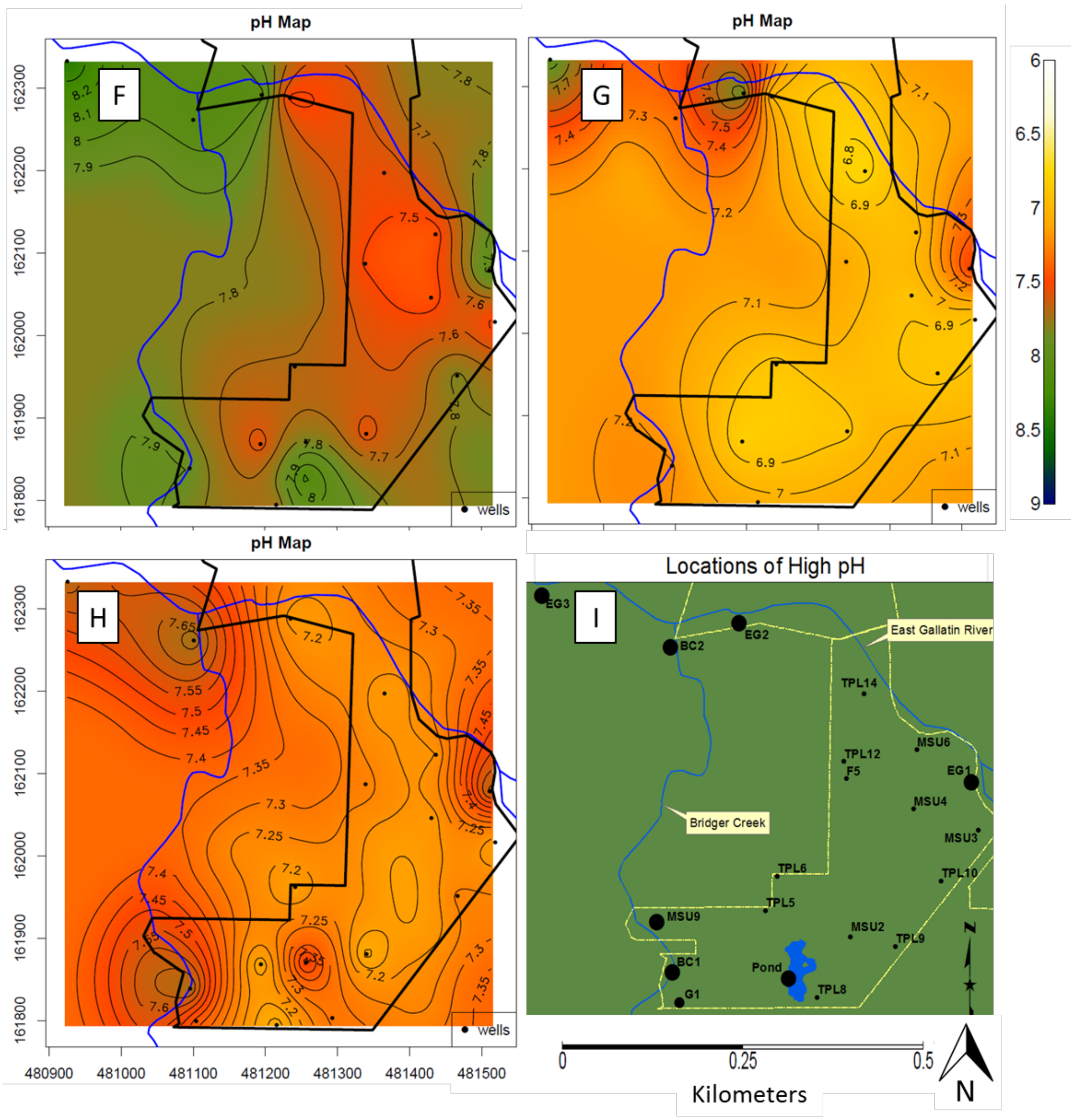

Figure 58. pH surfaces and map of occurrence above 7.75. Clockwise from the top: Image F, 9/25/2014, Image G, 10/2/2014, Image H, 4/23/2015.

DISCUSSION

This discussion makes inferences using the results from the measured data. It is important to recognize that this 20-hectare site is part of a much larger 33,000 ha watershed, with innumerable activities carrying on upstream that could potentially affect the hydrology and water quality in the streams and groundwater at Story Mill.

Results are used to discuss potential changes in groundwater/surface water hydrologic connectivity, wetland volume and area, and groundwater and surface water quality. Inferences about hydrologic connectivity are made using groundwater surfaces, continuous stage graphs, and groundwater velocity. Possible changes in wetland volume and area are discussed based on trends in depth to groundwater and groundwater elevation in individual wells, as well as site wide averages of wetland area and storage. The distinction between the effects on wetland area caused by lowering the ground surface and filling the ditch is assessed by comparing trends in depth to groundwater and groundwater elevation. Inferences about restoration impacts on water quality in the groundwater were made based on temporal trends in each water quality parameter. Trends in individual wells, and the site as whole, are both discussed. Possible water treatment within the wetland was considered with the results from the spatial analysis, from both the surfaces and the parallel lines models.

Recall that the desired response to the restoration is an increased hydrologic interaction between Bozeman Creek and the East Gallatin River with their respective floodplains, and ultimately with the wetland. The restoration also intended to increase wetland area, with the anticipation that dissolved oxygen levels would decrease as a response, and ultimately result in improved treatment and water quality.

Surface and Groundwater Interaction

Two restoration activities in the Southern parcel were intended to change the characteristics of the interaction between the creeks and the wetland: excavating the floodplain and filling the drainage ditch. One year after construction, conclusions on the connectivity between surface and groundwater, both in flow direction and magnitude, are based on results from groundwater surfaces and data logger hydrographs. The surface contours and HOBO data suggest that there is very little surface water from either East Gallatin River or Bozeman Creek flowing into the wetland. This limited connection means that any excessive nutrients in the creeks cannot be treated by the restored wetland. The greatest potential for interaction is during flood events when the Bozeman Creek Backwater Slough is activated. The designed 2-year flood frequency is not likely to make a water quality impact, because the percentage of stream flow (with quite low average TN levels of 0.9 mg/L) captured in the Slough during these flood events is so minor compared with total flow rates. Considering the hydraulic gradients, it appears that any water entering the Slough is only potentially treated by this area, and not the wetland as a whole.

Little is known about the detailed hydraulics in the Slough, but there are several possible concerns. As floodwater fills and leaves the Slough, are the velocities slow enough to deposit sediment, or could they be fast enough to reconstitute particle bound nitrogen and phosphorus? For any biotic removal of nutrients, the HRT must be long enough for microbial reduction or plant uptake to take place. Another concern may be that restoration activities depleted this area of its carbon stores, and that denitrification rates may be limited by available carbon. As the wetland in the slough matures, accumulates carbon, and its plant community becomes more verdant, these concerns may diminish. To better understand the effects of the slough on water quality, it is recommended that an intensive study during storm events, specifically focused on the Slough, be conducted. Kronvang et al. (2007) conducted a similar study measuring water exchange, flow patterns and deposition of sediment and phosphorus, which could serve as a model. At Story Mill the specific parameters of interest are water velocities, HRT, flow direction further into wetland (potentially driven by a mounding gradient), and sediment deposition and composition (carbon, nitrogen, phosphorus).

Since it is appears that the wetland is supplied by up-gradient groundwater, and not with water from the creeks, the magnitude of interaction is defined by the rate at which groundwater leaves the wetland into the creeks. The flow rates in the East Gallatin River dwarf the amount of groundwater conveyed by the wetland, considering the slow velocity of the groundwater. Since aquifer depth is unknown, a groundwater flow rate can only be estimated as a function of seepage velocity and an assumed effective crossectional area, using the equation:

$$Q = v_s * A * n$$

where v_s= seepage velocity (0.11 m/d), and *A* = total crossectional area from east to west (400 m), and *n* = porosity (0.4). In an extreme case, even if an aquifer depth of 20 m was assumed, the flowrate from the wetland would only amount to 350 m^3/d, 0.16% of the average flow rate in the East Gallatin River, at 221,000 m^3/d. At this ratio, conservatively assuming the highest groundwater contamination measured during this research of 3.7 mg/L TN, and that prior to restoration no treatment occurred, and now 100% treatment occurs, river water quality would only change by 0.006 mg/L. Regardless of pollution levels and aquifer depth, the water leaving the wetland would become so diluted by the flow in the creeks, it would not have an impact on overall water quality, and any change in treatment would be imperceptible.

Before construction, a portion of this up-gradient groundwater was intercepted and conveyed more quickly into the East Gallatin. By filling the ditch, this interaction changed, and the HRT of the flow in the ditch increased. By converting this flow to groundwater, both the flow path length was increased, and velocity was decreased, having compounding effects on increasing HRT. Based on a limited observation, if the flow in the ditch is assumed to be 12 m^3/d (8 L/min), it previously contributed an insignificant flowrate of water to the East Gallatin. Again, when the magnitude of interaction is so small, a potential change in treatment is negligible.

Volume/Area Discussion

All three major restoration efforts in the Southern Parcel were intended to increase wetland volume (groundwater storage at the site) and wetland area (defined here as the portion of the site where depth to water is less than 0.3 m) - filling the ditch, excavating areas of the floodplain, and recontouring various aspects of the site.

Estimated changes in wetland volume and/or wetland area can imply changes in water treatment potential, and are based on observed changes in the groundwater elevations and depths to groundwater. Although distinct localized changes were seen in the groundwater surface contour maps directly before and after filling the drainage ditch, significant increases in groundwater elevations trends with time were only observed in three wells, and total (spatially integrated) groundwater storage across the site decreased over the study duration. This suggests that filling the ditch had limited impact on changes in the shallow groundwater surface over the site as a whole. Wetland area on the other hand increased over the study duration, benefiting from the combined effects of lowering the ground surface and filling the ditch. This phenomena was evident locally in statistically significant trends of decreased depth to groundwater in three wells, though 26 other wells showed no similar trend. An increase in wetland area was also observable in the depth to groundwater surfaces. The increase in wetland area despite a decrease in groundwater storage, indicates that lowering the ground surface was the more instrumental restoration activity at Story Mill.

One way to cause an increase in the long-term steady water surface elevation (groundwater stored) would be a combination of increased subsurface flow into the site or decreased subsurface flow out of the site. This would result in a new equilibrium groundwater gradient that would, over the site as a whole, have a greater integrated groundwater volume. In the case of filling the ditch, it does not appear either that the inflow or outflow were measurably altered, and the effect of filling the ditch likely had minimal effects on the overall groundwater gradient. This assertion hinges on the limited before-restoration water level data. It is unclear whether, before restoration, the ditch was gaining or losing to the surrounding groundwater. The few observations from before filling the ditch, and the unknown role of the head gate in regulating water levels in the pond leads to the notion that at times the ditch may have actually served as an infiltration gallery, receiving effluent from the pond, mounding groundwater in the ditch, and creating a gradient that drove water into the wetland.

If it is assumed that the ditch was persistently acting to dewater the wetland, a radial sink can be used to approximate the extent of the before-restoration-drawdown, and thus the lateral extent over where increased water elevations could be expected as a result of filling the ditch. Parabolic profiles are used to estimate drawdown in most groundwater scenarios, where the impact of a dewatering feature becomes negligible as the distance away from the feature increases (R. A. Cooke et al., 2001). For this model, it was assumed that the ditch behaved as a groundwater well would in an unconfined aquifer, drawing groundwater in a typical cone of depression. Darcy's law could then be used to calculate the height of water at various distances away from the ditch extending to a perpendicular distance at which the drawdown fell within the error for water elevation measurements in this study (3 cm). The equation for the height of water above a confining layer at any given radius away from the ditch is:

$$h_R = \sqrt{h_o + \frac{Q_{Ditch} * \ln\frac{R}{r}}{2\pi * K_{sat}}}$$

where h_o= height of water in the ditch above a confining layer, under steady state drawdown conditions, assumed to be 5 m, Q_{Ditch}= flow rate conveyed by the ditch assumed to be 8 L/min, R = distance away from the ditch where desired height is calculated, r = radius of the pumping well assumed to be 0.3 m, and K_{sat}= hydraulic conductivity of the soil. A 5 m deep aquifer was assumed (for groundwater flow rate calculations, 20m was assumed) to conservatively capture the greatest feasible drawdown effect, because drawdown increases with decreased aquifer depth. The resulting drawdown profile is shown in Figure 59.

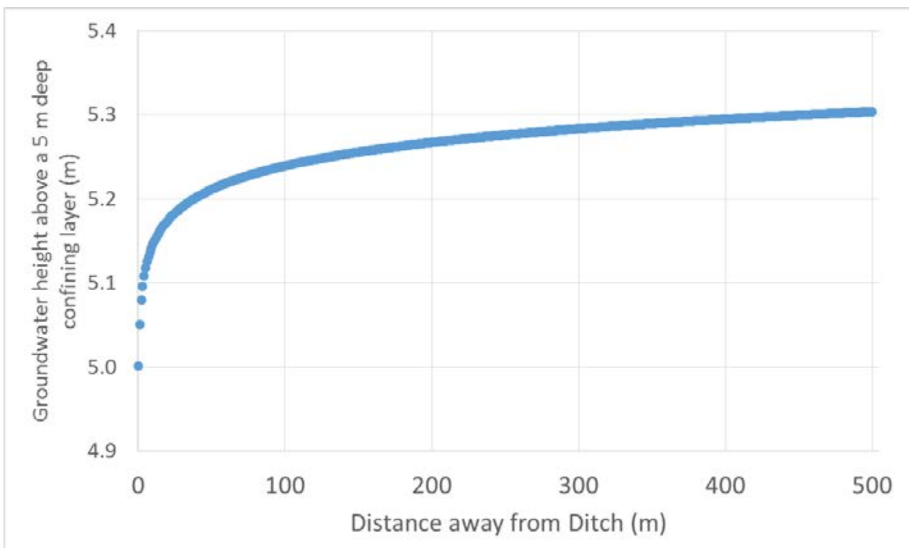

Figure 59. Drawdown profile as predicted by Darcy's Law, approximately representing the ditch's effect on groundwater levels, assuming it was acting as a drainage mechanism.

The change in groundwater height after filling the ditch at a distance R away from the ditch was calculated using the equation:

$$d_R = h_{500} - h_R$$

where h_{500}= the unaffected height of water above a confining layer assumed to occur at a distance of 500 m away. With these assumptions, and for the sake of establishing a conservative influence distance for filling the ditch, the maximum distance away from the ditch where water levels may measurably increase is 232 m. Table 6 reports predicted increases in groundwater elevation at various distances away from the ditch. The values in Table 6 and the drawdown profile show that the effect of filling the ditch quickly diminishes with increased distance, and that the most pronounced effect is closest to the ditch. For reference, TPL6 and TPL7 are 33 and 26 m away from the ditch, respectively. By this same philosophy, the steeper hydraulic gradient around the ditch will cause changes in this area to happen relatively quickly, but changes farther away could potentially take years to realize. Further slowing flow out of the site will have limited additional effects. Increasing flow into Story Mill is the most powerful tool for increasing wetland volume and area.

R (m)	d_R (cm)
232	3
142	5
41	10
12	15

Table 6. Darcy's Law was used to predict the parabolic drawdown curve and estimate groundwater elevation increases as a result of filling the ditch. Groundwater elevation increase, d_R, is reported at various distances away from the ditch, R.

Treatment/Nutrient Removal

Changes in nutrient removal and water quality at Story Mill were expected to be brought about by manipulations of the hydrology at the site. From analyzing the hydrologic data, through the combined efforts of lowering the ground surface and filling the ditch, there is evidence to suggest that the wetland area has increased at the site. HRT has increased for a small portion of flow through the wetland that had been channelized in the ditch. Both of these factors have the potential to improve the water quality treatment capacity of the site. The hydrologic connectivity between the creeks and the groundwater in the wetland is likely limited to the Backwater Slough on a two-year flood basis, and the percentage of flow contributed to the creeks from the wetland was calculated to be imperceptibly small. This limited interaction implies that a measurable reduction in nutrient concentrations in the East Gallatin River due to the three restoration components is not feasible. The following is a discussion of the nutrient concentration observations to assess if any restoration-associated changes in hydrology impacted water quality in the groundwater and/or surface water associated with Story Mill.

The ranges of TN, nitrate, TP, and orthophosphate data observed at Story Mill are all relatively small, where intrinsic site variability, measurements and laboratory error, and method detection limits can limit the definition of spatial or temporal trends and patterns. Longer data sets will enhance any conclusions, clarifying persistent trends, seasonal oscillations, and natural variability. The following discussion is speculative, but the validity and potential causes of patterns and trends are yet to be determined until more years of data can be included in future analyses.

Nitrogen

For the site as a whole, no temporal trends were observed for TN concentrations, and nitrate concentrations increased only very slightly. The combined effects of restoration activities in the watershed and at the site did not cause either an improvement or a degradation of the groundwater quality. This is despite an increase in wetland area and a decrease in average dissolved oxygen levels. A closer look at individual wells showed that two locations, MSU2 and TPL10, increased in TN, and two locations, MSU2 and F5, increased in nitrate. These trends could be due to worsening treatment capacity or increased loading, but without longer data sets to confirm the persistence of these trends, they may also simply be noise within the system. None of these wells are coincident with areas that suggest lowered treatment capacity due to changes in hydrology or dissolved oxygen concentrations (Figure 60), and other areas that may have improved in treatment capacity did not improve in water quality. In fact, there is little in the way of consistency: a clear connection between decreased depth to groundwater, decreased dissolved oxygen, and improved water quality, as would be expected, is not apparent. F5 for example, decreased in depth to groundwater and dissolved oxygen levels, implying improved treatment capacity, but increased in nitrate levels. MSU2 and TPL10 also increased in nitrate levels, but neither well showed changes in hydrology, and dissolved oxygen levels decreased in TPL10.

For wells that exhibited potentially improved treatment capabilities, but did not exhibit improved water quality, there are several possible theories. It could be that regardless of a reduction in dissolved oxygen, DO concentrations were still high enough to inhibit denitrification. Denitrification in soil has been found to proceed when dissolved oxygen concentrations are as high as 4 mg/L. It is thought that this is due to microenvironments within the soil pores of anaerobic conditions that can host denitrifying bacteria (Kronvang et al., 2007). All six wells that decreased in dissolved oxygen levels dropped to almost always measure below 2 mg/L, well within the range that could potentially support denitrification. A more likely scenario is that nitrate concentrations in these wells were actually the limiting factor for improved removal. In these wells nitrate concentrations rarely rose above detection, except for in F5 and TPL10, and even in these wells, levels typically stayed below 0.3 mg/L.

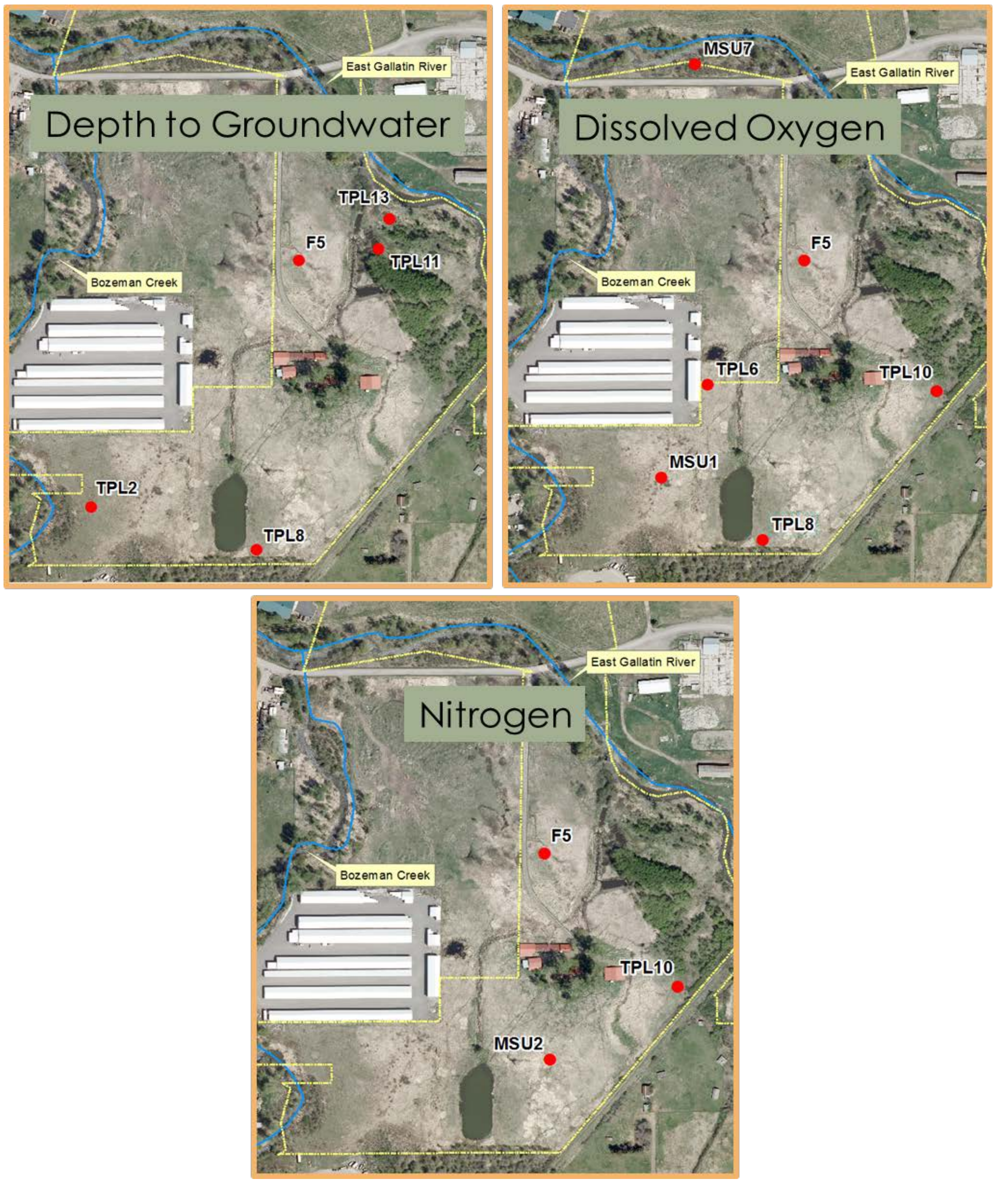

Figure 60. Each map indicates the locations of wells that exhibited statistically significant ($p<0.05$) temporal trends (both increasing and decreasing) in the specified parameter. The nitrogen map includes trends observed in TN and nitrate. Notice how few wells showed trends in any parameter, and that there is very little commonality between maps.

Other factors may have affected treatment capacity such as pH and the removal vegetation. F5 is located in an area where construction activities left the soil bare, which may have temporarily curtailed nutrient removal by plant up-take. This theory is not corroborated by trends in any other wells that were in similarly disturbed areas. Nitrogen removal is also affected by pH, where denitrification occurs optimally in a range from pH 6 to 8.5 (Wijler et al., 1954). In exceedingly acidic or basic conditions, the soil environment cannot support the plants and microbes that transform nitrogen. The pH observed at Story Mill was only slightly basic, and stayed comfortably within an acceptable range for nitrogen removal to take place.

There are several possibilities that may have caused increased nitrogen loading. Construction activities around F5 disturbed the soil, possibly exposing buried organic nitrogen to the atmosphere, which may have increased mineralization and nitrification rates. Again, similar responses were not observed in other wells affected by construction. Of the three wells that increased in nitrogen, only hydrologic changes were seen in F5, but it may be that increased intermittent wetting in all three wells may have been just enough to encourage a more robust microbial community, but not enough to select for denitrifiers, thus increasing mineralization and nitrification rates, but not nitrogen removal. The re-contouring of the pond may also have contributed to increased loading in MSU2. The banks of the pond were lowered, and overflow was encouraged to spill into the meadow where MSU2 is located. Although the data from the pond is sparse, all measurements there were above 1 mg/L TN, and could have potentially contributed to the increased TN and nitrate observations in MSU2.

Another pattern observed in TN concentrations for several wells was an increasing trend throughout the winter, and a decreasing trend starting in the spring and persisting through the summer. Nitrate levels did not follow as clear a pattern, but rather seemed to spike more randomly. Again, this could be due to changes in loading or removal, and several possible theories exist for this phenomenon. It may be that during the growing season and periods of warmer weather, plants and microbes more efficiently removed nitrogen. After the vegetation senesced in the fall and microbes began to slow down with the onset of winter, nitrogen concentrations increased as the removal capacity decreased. It may also be due to variations in deposition, such as precipitation water quality (Montana precipitation average for 2014 = 0.23 mg N/L total nitrogen), and fecal and plant decomposition cycles.

Spatially, for both TN and nitrate, wells in the south were typically more erratic and likely to spike with higher concentrations than wells in the north. Despite the propensity for wells in the south to spike to higher levels, this did not seem to cause the wells in the north to reciprocate, but rather they remained consistently low. Due to the slow nature of the groundwater flow, this spatial phenomenon may be due to several factors. It may be evidence of treatment along the flow path of groundwater through the wetland, but it also could be that through historical land use, the deposition of nitrogen was more intensive in the southern property. For example, this area may have been used as pasture for cattle, or was fertilized for crop production, more so than the northern meadows. The inclination of waterfowl to gather on the pond may also contribute to elevated TN and nitrate levels in the groundwater in this area.

If it is assumed that the spatial difference in nitrogen concentrations was due to treatment, a speculative estimate of nitrogen loading and percent reduction can be calculated, using the equations:

$$\dot{m} = [\mathrm{NO_3}] * Q$$

$$[\mathrm{NO3}]_{LR} = \frac{\dot{m}}{A}$$

$$\%\, reduction = \frac{[NO_3]_S - [NO_3]_N}{[NO_3]_S}$$

where $\dot{m}$ = mass flow rate, $[NO_3]$ = nitrate concentration, and subscripts *S* and *N* indicate concentrations measured in either in the south or the north respectively, Q = groundwater flow rate (350 m^3/d assuming aquifer depth of 20 m), $[NO_3]_{LR}$ = nitrate loading rate, A = area of the Southern Parcel (104,500 m^2). Maximum loading and treatment can be found by assuming influent groundwater nitrate levels are equal to the maximum levels measured in the south during the study (3.7 mg/L), and effluent groundwater is equal to the average levels measured in the north (0.065 mg/L). By these calculations, the nutrient loading rate is 4.52 gN/m^2/yr, with a 98% reduction. This nutrient loading rate is on the extreme low end of the spectrum observed in other natural wetlands, and fits well with the model created by W. J. Mitsch, Day, et al. (2005), where wetlands receiving such low influent regularly removed upwards of 90% of influent nitrate. The potential removal rate observed at Story Mill equates to approximately 470 kg N prevented from entering the East Gallatin River annually.

Nitrate and TN concentrations in the creeks oscillated throughout the monitoring period, but showed no distinct difference in water quality upstream vs downstream of

Story Mill. Samples taken from BC1 and BC2 increased and decreased in unison, as did samples taken from EG1 and EG2. Concentrations measured in EG3 could generally be explained by the mixing of the two creeks. These patterns suggest that the water quality in the creeks is unaffected by the restoration, and that any trends are at a watershed scale.

The surface water results also suggest that for the greatest impact on water quality, treating water in Bozeman Creek should be the top priority. Hypothetically speaking, if the entire flow of Bozeman Creek were to be re-routed into the wetland, an annual nutrient loading could be calculated by finding the average mass flow rate and dividing it by the area of the Southern Parcel, using the same equations presented for groundwater loading. Similarly the hydraulic loading rate could be calculated:

$$H_{LR} = \frac{Q}{A}$$

where H_{LR} = average annual hydraulic loading rate, Q = average flow rate in Bozeman Creek (73,400 m^3/d), and A = area of the Southern Parcel (104,500 m^2). By these calculations, the nitrate loading rate would be 123 $gN/m^2/yr$, and H_{LR}= 256 m/yr, or 70 cm/day. This nutrient loading rate is considered to be on the high end of the spectrum observed in other natural wetlands. According to the model created by W. J. Mitsch, Day, et al. (2005), as much as 40% could potentially be removed. The increased hydraulic loading is extremely high for a natural wetland system, where typical H_{LR} is more in the range of 1 to 10 cm/d (Kadlec et al., 2009; Kovacic et al., 2006; Spieles et al., 1998; Van et al., 2015), but H_{LR} have been reported as high as 27 cm/d (Avila et al., 2016) and 70-180 cm/d (Braskerud, 2002). Increasing H_{LR} to this degree would have several implications regarding flow regimes, and if this route were considered, the hydraulics of the site would need to be investigated thoroughly.

Phosphorus

On the whole, phosphorus concentrations associated with surface and groundwater at Story Mill were minimal, with only a handful of samples that registered detectable levels of TP or orthophosphate. All measurable TP and orthophosphate values occurred during the winter, with only one exception. No site-wide or individual trends were observed, and there was no correlation based on whether wells were located in the south or north; simply, phosphorus concentrations in all the groundwater and surface water throughout the site were persistently low.

There are several potential explanations for the typically low phosphorus concentrations and anomalous spike. As was discussed with nitrogen, there are two major ways to affect phosphorus concentrations in the groundwater: deposition and removal. It cannot be the case that there is simply no phosphorus at Story Mill, because its cycling is integral to life and senescence, therefore treatment of phosphorus must have matched deposition, except during a few months in the winter. It is evident from the data that phosphorus is equally low in any influent surface and groundwater, and therefore sources must be from rain, urine, and decomposing plants and fecal matter. These combined sources may have been just enough to supply the vegetation with phosphorus to thrive, but when the plants were dormant and no longer taking up nutrients, a spike occurred. It could also be that removal rates are relatively constant, but that deposition increased during this time.

Some removal must be due to plant up-take, but it is unclear the role of sorption and precipitation, or what the site's potential may be for removing phosphorus. pH and DO play important roles in the P cycle. At neutral pH, phosphorus is the most soluble and plant available – which is the case at Story Mill – and when pH is more basic, phosphorus will precipitate out of solution with available calcium ions. At very low dissolved oxygen levels, iron oxides are reduced, subsequently releasing any sorbed phosphorus into solution. Then, at this point, iron and aluminum ions are available to react with phosphorus to form precipitate, but this reaction is much slower than sorption kinetics. The underlying parent material at Story Mill is composed of cobbles derived from a broad range of lithologies, many of which are comprised of minerals containing iron, aluminum and calcium (Bourne et al., 1962; Hackett et al., 1960). Therefore, it is possible that the soil at Story Mill and sediment carried by the creeks, likely contain the elements ideal for phosphorus sorption and precipitation. There is also evidence of iron in the soils at the site from the redoximorphic features found by Stephanie Ewing's class (Ewing et al., 2013-2015). At the pH and dissolved oxygen ranges found in the groundwater, it is likely that sorption does not play a significant role in phosphorus removal, but that precipitation might.

SUGGESTION FOR FURTHER RESEARCH

Future studies of the impact of the Story Mill site on local water quality could be taken in several different directions. Hydrologic monitoring and modeling to-date indicate only a slight interaction between the shallow groundwater of the property and the surface streams that border the property. At this point, it appears that the site does not exhibit substantial water treatment capacity largely due to limited loading of treatable compounds and limited capability to transport either treatable or treated water. The data set collected so far at the Story Mill site only spans a few years, so of course the most pressing recommendation for further work is to continue to monitor hydrology and water quality parameters over time. There may be spatial or temporal phenomena of note that occur as plant communities adapt to the new land sculpture and altered land use. Annual scale changes in weather may have unpredicted effects – a few consecutive dry or wet years, for example. Natural changes in the morphology and vegetation in the nearby stream reaches could alter the water surface profiles there relative to the groundwater surface within the property.

Because the site is a public destination for learning and teaching, it is likely that our understanding of the soils and plants at the site will continue to increase with time. Perhaps in the future, hydraulic conditions will change enough as to perpetuate clearer associated changes in dissolved oxygen and nutrient concentrations, but it is likely that these relationships will always be hard to decipher because dissolved oxygen and nutrient levels are already so low. The effects of hydrologic changes can also be observed through other means: the formation of hydric soils and hydrophytic plant assemblages can confirm more wetland like conditions and imply improved treatment capacity. Focusing research efforts intensively on the Backwater Slough, especially during activation would also give a more complete idea of wetland treatment capacity. Research directed at the Slough would benefit from both hydraulic modeling and field data collection. There are several questions of interest: What are the surface water velocities? What is the hydraulic residence time? Are the flow regimes such that surface water scours or deposits sediment? What are deposited sediments comprised of, particularly carbon, nitrogen, phosphorus, iron, aluminum, and calcium? What are the soil formation and mineralization rates, including possible nitrogen and carbon accumulation rates?

Speculative nitrogen loading and removal rates were estimated based on a simple parallel lines model which showed a significant difference between concentrations of nitrogen measured from wells located in the south vs the north. To confirm treatment, a more detailed nitrogen balance should be carried out. Robust observations of water quality parameters would be of even more value if coupled with a fully developed groundwater model of the site.

There are many ways to think about treatment capacity and nutrient removal pathways at Story Mill, but it ultimately begs the question: Why look further? Regardless of how well water quality treatment capacity is understood at the Story Mill wetland, the implications for water quality enhancements in the East Gallatin River remain the same. Nutrient loading levels are low and the groundwater hydrology of the site is slow. The Story Mill site may be still an ideal case study to contribute to the working knowledge of wetland processes, but in this case relatively modest water quality treatment outcomes should not distract from the other important educational opportunities the site provides. The Story Mill site will serve as a teaching and learning venue for students and community members for years to come, promoting an understanding of the importance of natural systems and the benefits of the interactions between plants, soil, water, animals, and humans.

CONCLUSION

Restoration efforts in the Southern Parcel of the Story Mill site were, at least partially, intended to effect changes in groundwater and surface water hydrology to enhance water treatment opportunity and improve water quality. The development of the Backwater Slough was intended to improve the connection between Bozeman Creek and groundwater in the floodplain, ultimately promoting enhanced and enlarged wetlands at the site. Filling the ditch was intended to elevate the water table promoting a shallower vadose zone with more shallow groundwater availability to wetland plant species. In some areas the ground surface was lowered with the same goal of increasing the land area having shallow wetland-like groundwater availability. If the total land area having wetland-like conditions increased as a result of restoration activity, the hope was that dissolved oxygen levels would subsequently decrease, organic matter accumulation would be enhanced, and denitrification rates would increase in turn. Having a larger portion of the site exhibit wetland-like conditions might also have increased nutrient removal by plant uptake as a result of improved plant root access to potentially polluted groundwater.

Models of the groundwater surface for the site that reflect the first year after restoration activity indicate that the predominant hydraulic gradient drives water out of the wetland. Water generally enters the site as precipitation, surface run-on or groundwater flow along the southern property boundary, and moves generally northward towards and into the two streams. The interaction between groundwater and surface water is defined by slow moving groundwater seeping into the creeks. Localized interaction attributable to the development of the Backwater Slough was not evident over the duration of the project. Without increased interaction between the streams and the groundwater, nutrient loading remained the same before and after the restoration, driven by precipitation, influent groundwater, and onsite nutrient cycling.

Water quality analyses revealed that average nutrient levels in the groundwater for the duration of the study were low, with 74% of samples measuring non-detectable levels of nitrate, and almost 100% measuring non-detectable levels of TP and phosphate. Total nitrogen concentrations were consistently higher, but on average were less than 1 mg/L, still a remarkably small concentration. Spatially, there was evidence of higher nitrogen concentration in the groundwater entering from the south, and lower concentrations exiting to the north, but over the duration of the study there was no statistical evidence that that pattern changed over time. With groundwater conservatively estimated to travel through the wetland at 0.11 m/d, and relatively pristine influent groundwater quality, loading into the Story Mill system is fairly low compared with other natural wetland systems designed to improve water quality.

Changes in hydrology that were intended to increase the total on-site area exhibiting wetland characteristics were minor, with significant temporal trends in only a few isolated wells; in other words, hydrologic changes were not ubiquitous or particularly convincing. Only two of the wells that indicated elevated groundwater levels also exhibited significantly decreased dissolved oxygen concentration. In a few of other wells, dissolved oxygen decreased with no corresponding change in groundwater elevation. Nitrogen levels did not decrease in any wells, regardless of hydrology or dissolved oxygen trends, and actually increased in two wells that showed a decrease in dissolved oxygen - the opposite of what would be expected. After this short-duration study and in light of the low concentrations of dissolved oxygen and nutrients observed at the site, there are no apparent ties between restoration efforts, creek-wetland hydrologic connectivity, and nutrient treatment in the groundwater. With time, and perhaps changes in upstream hydrology or land use, the water quality treatment potential of the site may yet be realized. It is anticipated that the effects of filling the ditch, while likely on a spatial scale very local to the fill, will take more time to realize. As the ecosystem finds its new normal in the years to come, potentially hosting different plant assemblages and fostering slow changes in soil chemical and microbial characteristics, there may yet be enhanced water quality treatment opportunities. Even if treatment processes improved radically with time, however, the miniscule volumetric flow contribution from the on-site wetlands relative to the flow rates in the creeks will most likely render the effects non-detectable when measured as stream-water quality. When the potential mass of nutrients prevented from entering the East Gallatin River is considered (470 kg N/yr), these metrics make the importance of wetlands on the landscape more obvious. Although the preservation and restoration of Story Mill alone may be making an imperceptible difference in nutrient concentrations in the East Gallatin River, when wetlands like it are considered as a cumulative whole within a watershed, their incremental effects provide essential resilience.

The pollution levels in Bozeman Creek were consistently higher than in the East Gallatin River and the site's shallow groundwater. Story Mill's impact on water quality in the East Gallatin River could be significantly increased if more of Bozeman Creek were to flow through the wetland. This would increase nutrient and hydraulic loading such that the flow rate of water potentially treated by the wetland could have measurable impacts on the water quality in the East Gallatin River. Improved measureable impacts could make Story Mill a more viable site for possible nutrient trading under Circular-DEQ 13.

Although groundwater/surface water interaction is limited, and water quality was found to be relatively pristine, wetland function encompasses a litany of other definitions. Lowering the ground surface and elevating the groundwater table significantly increased wetland area, potentially satisfying several other goals of the restoration, such as improving native plant and habitat diversity. This property is preserved as an open space and will serve as a gathering place for people and animals alike. Providing exposure to wetlands and fostering a paradigm shift where everyone understands the vitality of wetland ecosystems is a crucial piece of protecting water resources in the future.

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APPENDICES

APPENDIX A

GROUNDWATER WELL AND GUAGING STATION OWNERS, INSTALLATION SCHEDULE, AND LOCATIONS

Well ID	Well Owner	Survey ID	Action	Date	Northing (ft)	Easting (ft)	Elevation (ft)
F1	Hyalite	F139173	Installed	4/1/2007	530875.902	1579079.25	4734.5302
F1	Hyalite		Removed	10/2/2014			
F2	Hyalite	F239173	Installed	4/1/2007	530937.704	1579287.51	4732.932
F3	Hyalite	F339173	Installed	4/1/2007	531309.033	1578912.36	4728.721
F4	Hyalite	F439173	Installed	4/1/2007	531422.678	1579455.21	4728.114
F5	Hyalite	F539173	Installed	4/1/2007	531781.307	1579195.37	4722.77
F6	Hyalite	F639173	Installed	4/1/2007	532071.648	1579186.56	4721.39
F7	Hyalite	F739173	Installed	4/1/2007	532197.333	1579187.6	4721.6509
G1	Hyalite	G139173	Installed	4/1/2007	530837.697	1578423.87	4737.003
G3	Hyalite	G339173	Installed	4/1/2007	531190.989	1578491.89	4731.4379
G3	Hyalite		Removed	9/5/2014			
Pond	MSU	Pond41835	Installed	7/15/2014	530929.84	1578919.85	4729.5716
Pond	MSU		Removed	10/2/2014			
Pond	MSU	Pond41971	Installed	11/28/2014	530929.909	1578919.79	4729.6023
Pond	MSU		Removed	2/10/2015			
Pond	MSU	Pond42102	Installed	4/8/2015	531072.456	1578925.39	4732.939
BC1	MSU	BC141835	Installed	7/15/2014	530967.573	1578396.75	4731.613
BC2	MSU	BC241835	Installed	7/15/2014	532355.788	1578414.37	4713.954
BC2	MSU		Removed	6/2/2015			
BC2	MSU	BC242164	Installed	6/9/2015	532355.526	1578414.08	4714.01
EG1	MSU	EG141835	Installed	7/15/2014	531752.946	1579759.35	4722.306
EG2	MSU	EG241835	Installed	7/15/2014	532455.155	1578724.34	4714.4363
EG3	MSU	EG341835	Installed	7/15/2014	532588.679	1577839.25	4709.902
MSU1	MSU	MSU141849	Installed	7/29/2014	531063.81	1578718.65	4732.049
MSU2	MSU	MSU241849	Installed	7/29/2014	531104.365	1579199.7	4731.11
MSU3	MSU	MSU341849	Installed	7/29/2014	531550.363	1579785.12	4724.993
MSU4	MSU	MSU441849	Installed	7/29/2014	531646.365	1579494.31	4724.251
MSU5	MSU	MSU541849	Installed	7/29/2014	531752.099	1579288.21	4723.467
MSU6	MSU	MSU641849	Installed	7/29/2014	531899.063	1579514.49	4720.819
MSU7	MSU	MSU741849	Installed	7/29/2014	532442.105	1578853.98	4716.653
MSU8	MSU	MSU841902	Installed	9/15/2014	531187.893	1578320	4730.605
MSU9	MSU	MSU941971	Installed	11/28/2014	531183.823	1578329.39	4730.663
MSU10	MSU	VISU1042024	Installed	1/20/2015	531789.326	1579445.19	4723.547
Ditch1	MSU	Ditch141852	Installed	8/1/2014	531171.449	1578974.26	4727.1254
Ditch1	MSU		Removed	10/14/2015			
Ditch2	MSU	Ditch241852	Installed	8/1/2014	531369.645	1578982.81	4726.3877
Ditch3	MSU	Ditch341852	Installed	8/1/2014	531535.741	1579226.11	4724.6247
Ditch4	MSU	Ditch441852	Installed	8/1/2014	531859.024	1579370.62	4721.5051
Ditch4	MSU		Removed	10/14/2014			
TPL1	TPL	TPL141400	Installed	5/6/2013	531133.456	1578368.61	4730.7977
TPL1	TPL		Removed	9/5/2014			
TPL2	TPL	TPL241400	Installed	5/6/2013	530854.558	1578508	4735.7881

Well ID	Well Owner	Survey ID	Action	Date	Northing (ft)	Easting (ft)	Elevation (ft)
TPL2	TPL		Removed	9/5/2014			
TPL2	TPL	TPL241971	Installed	11/25/2014	530854.558	1578508	4735.7881
TPL3	TPL	TPL341400	Installed	5/6/2013	530855.601	1578511.06	4735.619
TPL4	TPL	TPL441400	Installed	5/6/2013	530822.619	1578790.8	4733.473
TPL5	TPL	TPL541400	Installed	5/6/2013	531225.44	1578819.07	4730.299
TPL6	TPL	TPL641400	Installed	5/6/2013	531371.471	1578874.99	4728.917
TPL7	TPL	TPL741400	Installed	5/6/2013	531363.408	1579072.53	4729.222
TPL8	TPL	TPL841400	Installed	5/6/2013	530849.603	1579043.82	4734.1853
TPL8	TPL		Removed	10/2/2014			
TPL8	TPL	TPL841968	Installed	11/25/2014	530849.503	1579043.82	4732.846
TPL9	TPL	TPL941400	Installed	5/6/2013	531061.076	1579402.35	4732.82
TPL10	TPL	TPL1041400	Installed	5/6/2013	531336.176	1579613.42	4728.867
TPL11	TPL	TPL1141400	Installed	5/6/2013	531813.277	1579447.04	4723.307
TPL12	TPL	TPL1241400	Installed	5/6/2013	531855.388	1579182.85	4722.921
TPL13	TPL	TPL1341400	Installed	5/6/2013	531910.022	1579484.84	4721.157
TPL14	TPL	TPL1441400	Installed	5/6/2013	532143.025	1579281.42	4719.489
TPL15	TPL	TPL1541400	Installed	5/6/2013	532124.013	1579166.53	4721.372

APPENDIX B

WATER LEVEL FIELD SHEET

Groundwater level field sheet						
Date:		Scientist:		Weather:		
Well ID	Time of arival	Stick up	Depth to GW	Notes		
TPL 1						
TPL 2						
TPL 3						
TPL 4						
TPL 5						
TPL 6						
TPL 7						
TPL 8						
TPL 9						
TPL 10						
TPL 11						
TPL 12						
TPL 13						
TPL 14						
TPL 15						
MSU 1						
MSU 2						
MSU 3						
MSU 4						
MSU 5						
MSU 6						
MSU 7						
MSU 8						
MSU 9						
MSU 10						
F1						
F2						
F3						
F4						
F5						
F6						
G1						
G3						

Surface water level field sheet						
Date:		Scientist:		Weather:		

Well ID	Time of arival	Staff gauge	Notes
BC1			
BC2			
EG1			
EG2			
EG3			
Pond			
Ditch1			
Ditch2			
Ditch3			
Ditch4			

APPENDIX C

WATER CHEMISTRY FIELD SHEET

Sampling Field Sheet

Date: Weather:

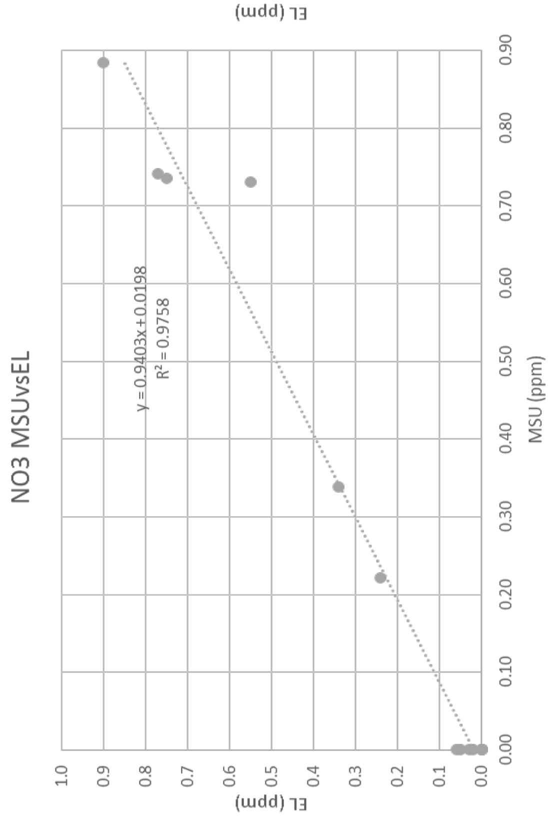
Time pH is measured:

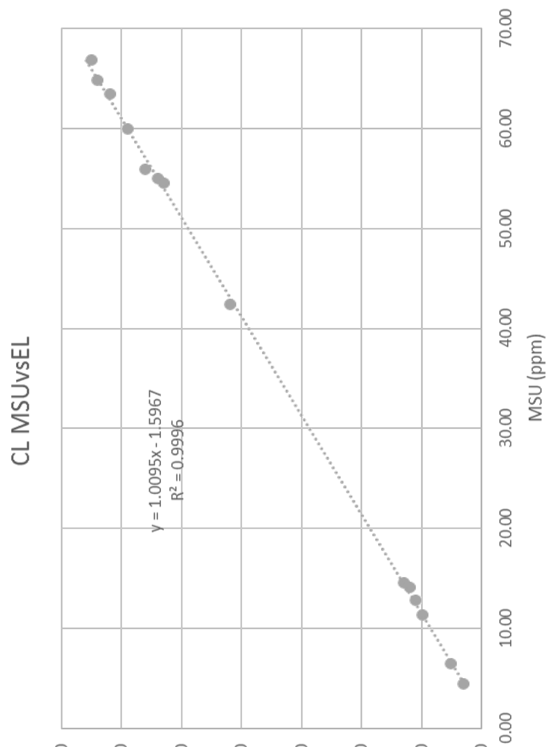
Time samples are stored in MSU Cold Lab: Scientist:

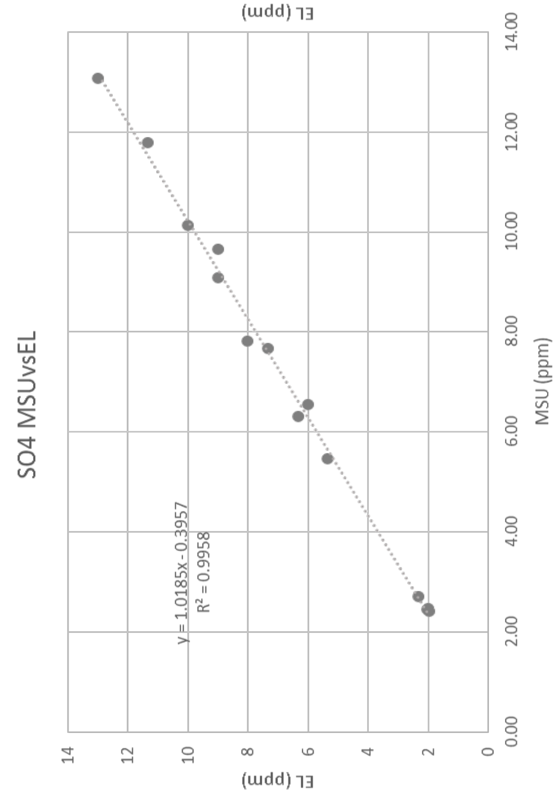
Well Name	Test Tube ID	Time Collected	Dissolved Oxygen	pH	Notes
TPL6	WellName_MMDDYY_A				
	WellName_MMDDYY_B				
TPL8					
TPL10					
TPL14					
MSU1					
MSU2					
MSU3					
MSU4					
MSU6					
MSU7					
MSU9					
G3					
F5					
BC1					
BC2					
EG1					
EG2					
EG3					
Pond					

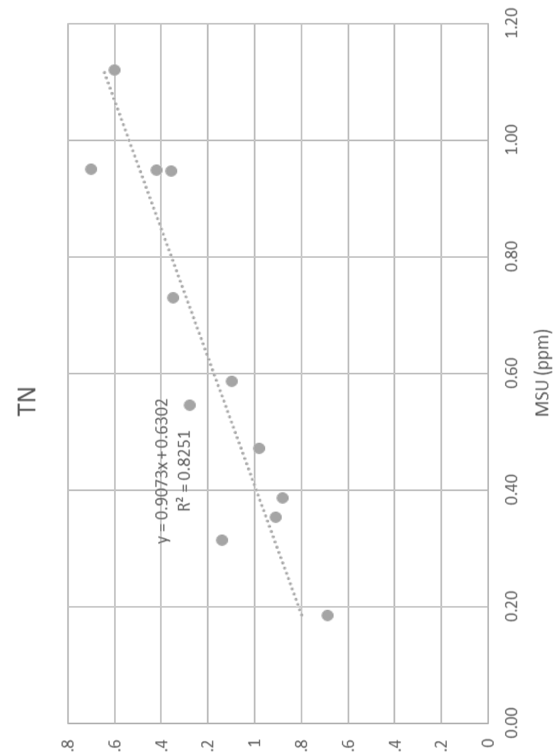
APPENDIX D

SPLIT SAMPLES COMPARISION


NO3 MSUvsEL
y = 0.9403x + 0.0198
R² = 0.9758
EL (ppm)
MSU (ppm)


CL MSUvsEL
y = 1.0095x - 1.5967
R² = 0.9996
EL (ppm)
MSU (ppm)


SO4 MSUvsEL
y = 1.0185x - 0.3957
R² = 0.9958
EL (ppm)
MSU (ppm)


TN
y = 0.9073x + 0.6302
R² = 0.8251
EL (ppm)
MSU (ppm)

APPENDIX E

HYDRAULIC CONDUCTIVITY CALCULATIONS

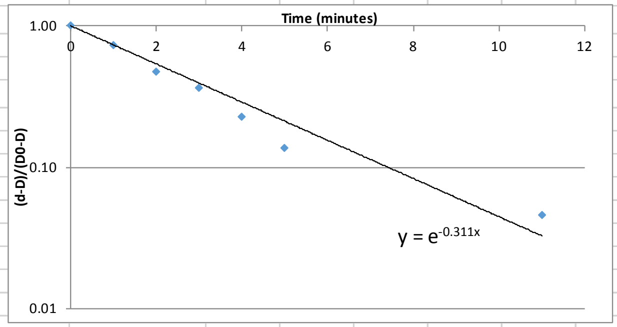
TPL 3

Depth to GW after pumping =	D_0=	191	cm
Depth to GW before pumping =	D=	169	cm

Time	$(d-D)/(D_0-D)$	Depth to GW (d)
0	1.00	191
1	0.73	185
2	0.48	179.5
3	0.36	177
4	0.23	174
5	0.14	172
11	0.05	170

To =	ln(0.37)/(-0.311)*60	191.82	sec

* To = time when $(d-D)/(D_0-D) = 0.37$

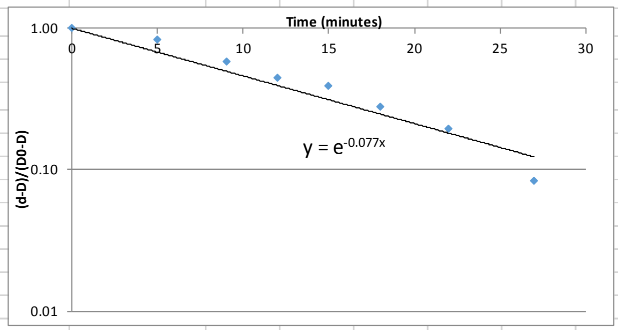


TPL 4

Depth to GW after pumping =	D_0=	103	cm
Depth to GW before pumping =	D=	85	cm

Time	$(d-D)/(D_0-D)$	Depth to GW (d)
0	1.00	103
5	0.83	100
9	0.58	95.5
12	0.44	93
15	0.39	92
18	0.28	90
22	0.19	88.5
27	0.083	86.5

To =	ln(0.37)/(-0.077)*60	774.74	sec

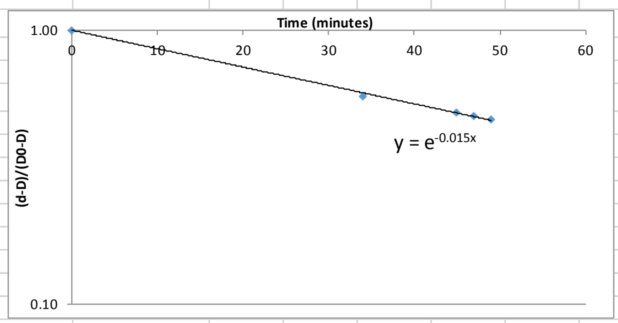


TPL 5

Depth to GW after pumping =	D_0=	146	cm
Depth to GW before pumping =	D=	105	cm

Time	$(d-D)/(D_0-D)$	Depth to GW (d)
0	1.00	146
34	0.57	128.5
45	0.50	125.5
47	0.49	125
49	0.48	124.5

To =	ln(0.37)/(-0.015)*60	3977.01	sec

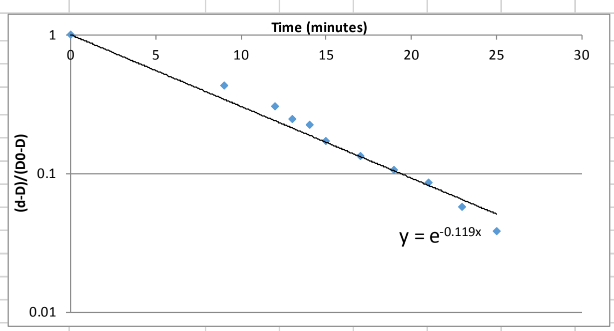


TPL 6

Depth to GW after pumping =	D_0=	200.5	cm
Depth to GW before pumping =	D=	95	cm

Time	$(d-D)/(D_0-D)$	d
0	1	200.5
9	0.426540284	140
12	0.303317536	127
13	0.246445498	121
14	0.222748815	118.5
15	0.170616114	113
17	0.132701422	109
19	0.104265403	106
21	0.085308057	104
23	0.056872038	101
25	0.037914692	99

To =	ln(0.37)/(-0.119)*60	501.30	sec

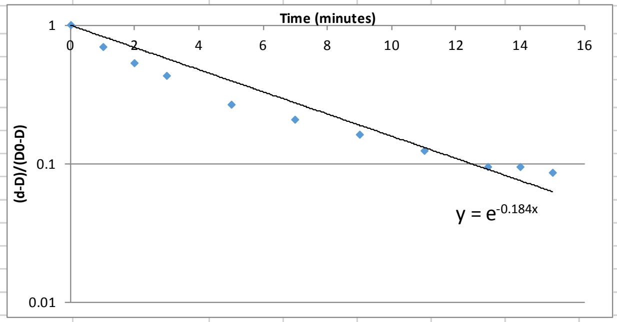


TPL 7

Depth to GW after pumping =	D_0=	155	cm
Depth to GW before pumping =	D=	102	cm

Time	$(d-D)/(D_0-D)$	d
0	1	155
1	0.698113208	139
2	0.528301887	130
3	0.424528302	124.5
5	0.264150943	116
7	0.20754717	113
9	0.160377358	110.5
11	0.122641509	108.5
13	0.094339623	107
14	0.094339623	107
15	0.08490566	106.5

To =	ln(0.37)/(-0.184)*60	324.21	sec

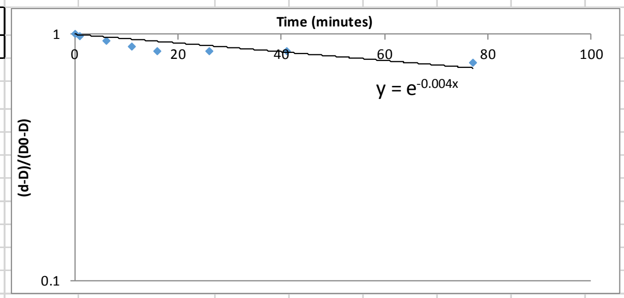


TPL 8

Depth to GW after pumping =	D_0=	167	cm
Depth to GW before pumping =	D=	120	cm

Time	$(d-D)/(D_0-D)$	d
0	1	167
1	0.978723404	166
6	0.936170213	164
11	0.893617021	162
16	0.85106383	160
26	0.85106383	160
41	0.85106383	160
77	0.765957447	156

To =	ln(0.37)/(-0.004)*60	14913.78	sec

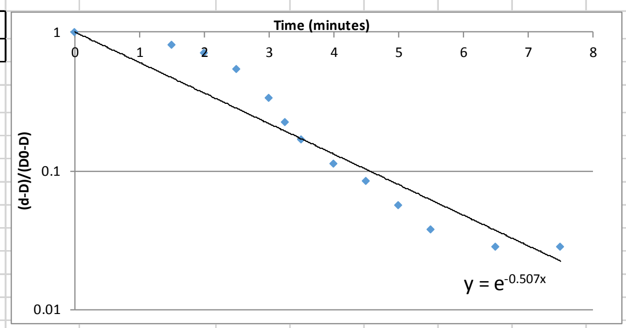


TPL 9

Depth to GW after pumping =	D_0=	185	cm
Depth to GW before pumping =	D=	132	cm

Time	$(d-D)/(D_0-D)$	d
0	1	185
1.5	0.811320755	175
2	0.716981132	170
2.5	0.547169811	161
3	0.339622642	150
3.25	0.226415094	144
3.5	0.169811321	141
4	0.113207547	138
4.5	0.08490566	136.5
5	0.056603774	135
5.5	0.037735849	134
6.5	0.028301887	133.5
7.5	0.028301887	133.5

To =	ln(0.37)/(-0.507)*60	117.66	sec

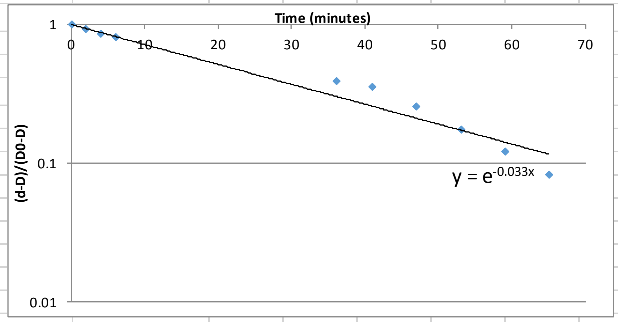


TPL 10

Depth to GW after pumping =	D_0=	169	cm
Depth to GW before pumping =	D=	103	cm

Time	$(d-D)/(D_0-D)$	d
0	1	169
2	0.924242424	164
4	0.863636364	160
6	0.818181818	157
36	0.393939394	129
41	0.356060606	126.5
47	0.257575758	120
53	0.174242424	114.5
59	0.121212121	111
65	0.083333333	108.5

To =	ln(0.37)/(-0.033)*60	1807.73	sec

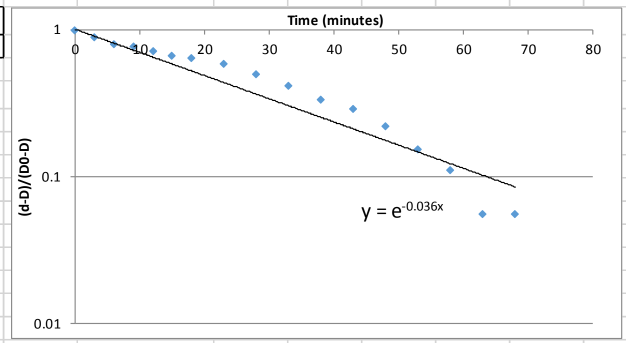


TPL 11

Depth to GW after pumping =	D_0=	155	cm
Depth to GW before pumping =	D=	119	cm

Time	$(d-D)/(D_0-D)$	d
0	1	155
3	0.888888889	151
6	0.805555556	148
9	0.763888889	146.5
12	0.722222222	145
15	0.666666667	143
18	0.638888889	142
23	0.583333333	140
28	0.5	137
33	0.416666667	134
38	0.333333333	131
43	0.291666667	129.5
48	0.222222222	127
53	0.152777778	124.5
58	0.111111111	123
63	0.055555556	121
68	0.055555556	121

To =	ln(0.37)/(-0.036)*60	1657.09	sec

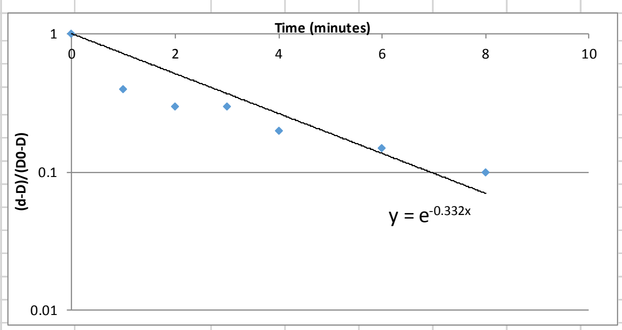


TPL 12

Depth to GW after pumping =	D_0=	118
Depth to GW before pumping =	D=	113

Time	$(d-D)/(D_0-D)$	d
0	1	118
1	0.4	115
2	0.3	114.5
3	0.3	114.5
4	0.2	114
6	0.15	113.75
8	0.1	113.5

To =	ln(0.37)/(-0.332)*60	179.68	sec

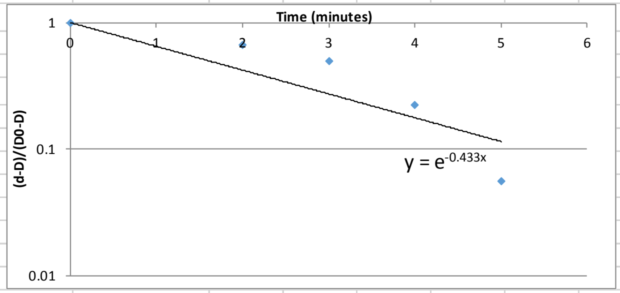


TPL 13

Depth to GW after pumping =	D_0=	115	cm
Depth to GW before pumping =	D=	106	cm

Time	$(d-D)/(D_0-D)$	d
0	1	115
2	0.666666667	112
3	0.5	110.5
4	0.222222222	108
5	0.055555556	106.5

To =	ln(0.37)/(-0.433)*60	137.77	sec



Summary

Well	L	R	r	To	$K_{sat} = (r^2 \cdot \ln(L/R))/(2 \cdot L \cdot To)$	
TPL 3	2	0.125	0.08	191.8	2.5E-05	7.6E-06
TPL 4	2	0.125	0.08	774.7	6.2E-06	1.9E-06
TPL 5	2	0.125	0.08	3977.0	1.2E-06	3.7E-07
TPL 6	2	0.125	0.08	501.3	9.6E-06	2.9E-06
TPL7	2	0.125	0.08	324.2	1.5E-05	4.5E-06
TPL8	2	0.125	0.08	14913.8	3.2E-07	9.8E-08
TPL9	2	0.125	0.08	117.7	4.1E-05	1.2E-05
TPL10	2	0.125	0.08	1807.7	2.7E-06	8.1E-07
TPL11	2	0.125	0.08	1657.1	2.9E-06	8.9E-07
TPL12	2	0.125	0.08	179.7	2.7E-05	8.2E-06
TPL13	2	0.125	0.08	137.8	3.5E-05	1.1E-05
Units	ft	ft	ft	s	ft/s	m/s