

EVALUATING THE EFFICACY OF WHOLE-WATERSHED STREAM
RESTORATION: IMPLICATIONS FOR GROUNDWATER STORAGE, WATER
QUALITY, AND SUSPENDED SOLIDS IN A PIEDMONT URBAN FOREST
WATERSHED

by

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ABSTRACT

DESTINI N. PETITT. Evaluating the Efficacy of Whole-Watershed Stream Restoration: Implications for Groundwater Storage, Water Quality, and Suspended Solids in a Piedmont Urban Forest Watershed. (Under the direction of DR. DAVID VINSON)

The quality and quantity of water in degraded watersheds (i.e.: watersheds undergoing changing land-use, urbanization, agricultural use) may be impaired compared to undisturbed forested watersheds. To enhance the ecosystem function of impaired watersheds, restoration projects have been implemented on a variety of land-use types and at a variety of spatial scales. The 6 km² Reedy Creek urban forested headwater watershed located in Charlotte, North Carolina, has undergone extensive, whole-watershed stream restoration to offset ecosystem degradation caused by historical agricultural use and urban development within the watershed. Streams within the Reedy Creek watershed exhibited channel straightening, disconnection with surrounding floodplains, channel widening, and channel incision. Here, three research topics were investigated with the aim of understanding the efficacy of using restoration to meet stated goals and to reestablish ecological and hydrological functioning within the watershed: 1) if whole-watershed stream restoration was successful at increasing recharge to the thick unsaturated zone and ultimately raising groundwater levels by creating more shallow streams; 2) if the whole-watershed stream restoration approach used in the Reedy Creek watershed in which stream channel morphology and characteristics were altered has changed N retention and seasonal flux; and 3) whether stream restoration has influenced TSS flux within the watershed at both baseflow and stormflow.

Study 1 To examine groundwater storage, groundwater levels were measured continuously from January 2016 – March 2021 in five well transects throughout the

watershed. Post-restoration, groundwater levels in ten wells increased a total of 1.6 m, on average, due to channel relocation. Post-restoration, “flashy” groundwater behavior ended, suggesting that fast pathways were saturated as the water table rose. Stream water levels rose, on average, ~0.3 m at the five studied transects. Both elevated stream water and groundwater levels were attributed to the physical raising of stream channels. Overall, the post-restoration watershed has increased groundwater storage.

Study 2 To examine watershed nitrogen dynamics within the restored watershed, weekly baseflow water samples were monitored for nitrate (NO_3^-), ammonium (NH_4^+), total dissolved nitrogen (TDN), and dissolved organic nitrogen (DON) (January 2016 – March 2021) and seasonal ash-free dry mass and canopy cover data were monitored at the watershed outlet. Data indicate that percent ash free dry mass (AFDM) increased from a pre-restoration average of 1.6% to a post-restoration average of 2.8% and canopy cover decreased from a pre-restoration average of 56 % to a post-restoration average of 50 %. Nitrate concentrations decreased from a pre-restoration average of 0.29 mg N/L to a post-restoration average of 0.08 mg N/L while nitrate flux also decreased from a pre-restoration average of 12.86 mg N/s to a post-restoration average of 5.13 mg N/s. TDN concentrations decreased from a pre-restoration average of 0.43 mg N/L to a post-restoration average of 0.32 mg N/L while flux decreased from a pre-restoration average of 16.96 mg N/s to a post-restoration average of 16.94 mg N/s. DON concentration increased from a pre-restoration average of 0.15 mg N/L to a post-restoration average of 0.21 mg N/L while flux increased from a pre-restoration average of 5.50 mg N/s to a post-restoration average of 977 mg N/s. Here, a significant relationship existed between watershed characteristic changes (i.e.: AFDM and canopy cover) and

ammonium and DON flux. Overall, stream restoration was associated with nitrate and TDN concentrations and flux and increasing DON and ammonium concentrations and flux.

Study 3 To examine changes in baseflow and stormflow TSS concentrations and flux due to changes in channel morphology, bed features, channel depth, and a shift from sand/gravel to clay dominated bank and bed (i.e.: soil cohesion would increase requiring higher shear stress to dislodge and move clay particles), baseflow surface water samples were collected weekly and analyzed for TSS from 2015– 2021 while stormflow samples were collected throughout the study period. TSS concentration and flux were compared before and after restoration at the watershed outlet. Baseflow TSS concentrations decreased from a pre-restoration average of 20 mg/L to a post-restoration average of 7 mg/L while flux decreased from a pre-restoration average of 48.8 kg/ha/year to a post-restoration average of 23.3 kg/ha/year. Stormflow TSS concentrations decreased from a pre-restoration average of 331 mg/L to a post-restoration average of 67.2 mg/L while per storm TSS flux g decreased from a pre-restoration average of 1.9×10^6 g to a post-restoration average of 1.2×10^6 g. Overall, stream restoration led to a decrease in TSS at both baseflow and stormflow which may be an indication of reduced erosion within the channel and a reduction in turbulent flow.

These findings have implications for developing stream restoration plans for degraded watersheds, such as understanding the changes to groundwater storage, watershed-scale hydrologic retention, drought resiliency, and stormwater management. Furthermore, findings may be an indication of increased hyporheic exchange potential in the channel leading to increased denitrification in the post-restoration period, increase in

organic nitrogen source for the stream, and changes in temperature and dissolved oxygen levels altering nitrification potential. Finally, findings may be an indication of increased stream bed and bank stability with decreased TSS in the post-restoration period. Overall the stream restoration approach used within the Reedy Creek headwater watershed was successful at meeting stated restoration goals. Here, stream restoration was successful at meeting the stated restoration goals of enhancing water quality and reducing water pollution within the watershed. Moreover, this restoration project was also successful at increasing groundwater storage – which was not a stated goal of restoration but one that is directly related to restoration at the watershed scale. Overall, the Reedy Creek restoration project was effective at meeting stated goals.

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DEDICATION

To Nacho and our furry and scaly babies

“If there is magic on this planet, it is contained in water”

– Loren Eiseley

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1 INTRODUCTION

Earth has been deemed the Blue Planet due to the abundance of water found on its surface. While 71% of Earth's surface is water covered, less than 1% of that water is readily available fresh surface water (e.g.: rivers, streams, and lakes) (Holden, 2014). In 2017 the EPA evaluated 1.1 million of the nearly 3.5 million freshwater streams in the United States and estimated that half were impaired with low water quality and limited natural functions (EPA, 2017). Stream restoration is an increasingly deployed practice world-wide and is being implemented to offset ecosystem degradation and impairment caused by human alterations of the biosphere (Bernhardt et al., 2005; Jenkinson et al., 2006; Szalkiewicz et al., 2018). Restoration projects are seen in the US (Jenkinson et al., 2006), Europe (Szalkiewicz et al., 2017), Australia (Brooks & Lake, 2007), Netherlands (dos Reis Oliveira et al., 2020), and Asia and are taking place in a variety of land-uses such as urban, suburban, urban-forested, agricultural, and forested watersheds. While restoration is needed and is currently being implemented in all these locations/habitats, little is known about the efficacy of restoration approaches and how they impact long-term watershed functioning. Moreover, even less is known about how large restoration projects (i.e.: over 1 km of stream length restored) affect beneficial watershed function (dos Reis Oliveira et al., 2020) which include, but are not limited to, groundwater, nutrient, and suspended solid dynamics. Beneficial groundwater functions include groundwater storage and precipitation-groundwater interactions within a watershed (i.e.: increase hydrologic retention and groundwater recharge). Nutrient dynamics include the effect of watershed alterations (e.g.: clear cutting, increased light, reconnection of floodplains) on the ability of the watershed to process and retain nutrients. Finally,

suspended solid dynamics include restoration success at returning bank erosion processes to pre-impacted conditions within the watershed.

This dissertation seeks to investigate the efficacy of whole-watershed stream restoration to meet stated restoration goals in a 6km² urban forest watershed in the Piedmont region of North Carolina. To do this, I will investigate: 1) physical hydrology, if restoration has improved hydrologic retention, and if changes to the watershed have led to enhanced precipitation – groundwater dynamics; 2) nutrient retention dynamics, if restoration has led to enhanced nutrient retention and processing; and 3) suspended solid dynamics, if restoration has ameliorated degradation associated with the urban stream syndrome.

1.1 Importance of headwater streams in temperate forested regions

Headwater streams, consisting of zero, first, second, and third order streams (Fritz & Johnson, 2021; Toronto and Region Conservation Authority (TRCA), 2007), represent ~two-thirds of all stream length globally (Leopold et al., 1964) and have the ability to regulate ecosystem functioning of overall river systems (Freeman et al., 2007).

Headwater streams are located in the uppermost areas of watersheds where stream flow begins and furthest from rivers (U.S. EPA, 2015). Non-permanent intermittent streams or non-channeled small watersheds are known as zero-order streams and support runoff during and after precipitation events (Toronto and Region Conservation Authority (TRCA), 2007). According to the Strahler (1957) stream order classification which is based on the type and number of tributary junctions present, a first order stream is defined as a stream that is unbranched and has no tributaries (Fitzpatrick et al., 1998; Toronto and Region Conservation Authority (TRCA), 2007). A second order stream

exists at the confluence of two first order streams (Strahler, 1957; Fitzpatrick et al., 1998; Toronto and Region Conservation Authority (TRCA), 2007). Furthermore, a third order stream exists at the confluence of two second order streams. This study focuses on a headwater watershed and includes data collected from first, second, and third order tributaries within the Reedy Creek watershed.

Headwater streams are hotspots for nutrient processing in which they retain waters, sediments, nutrients, and pollution from the land they drain; and, when functioning, are hydrologically connected and exhibit robust riparian zones, these streams have the ability to process naturally occurring constituent additions to the stream, thus regulating downstream water quality. Moreover, all streams are hydrologically connected, and, therefore, any changes in hydrological flow paths (i.e.: degradation caused by changing land-use) have the potential to impact downstream habitats (Freeman et al., 2007). Hydrologic retention, the processes that increase water residence times within a watershed, is directly responsible for the transport and retention of nutrients (Valett et al., 1996). Examples of processes that contribute to hydrologic retention are flooding, instream pools, large woody debris additions, groundwater recharge, and hyporheic exchange.

Downstream river systems are supported by baseflow derived in headwater watersheds (Alexander et al., 2007; Hofmann et al., 2018) and, therefore, should be an essential focus for restoration and protection. Headwaters are essential for sustaining flow during dry periods as they release stored waters over time (Alexander et al., 2007; Hofmann et al., 2018). With headwater streams making up the majority of stream length, they play an essential role as regulatory bodies as they receive extensive waters and

nutrients from the terrestrial landscape (Bernal et al., 2019; Li et al., 2020). Smaller streams characteristic of headwater watersheds have higher surface area to volume ratios than rivers and therefore, have the potential for increased nutrient processing and retention (Li et al., 2020). Low flow velocities increase residence times in headwater streams and aid in hydrologic retention of nutrients (Alexander et al., 2000; Li et al., 2020). Ultimately, headwaters in forest watersheds (i.e.: natural streams) work to regulate and control downstream water quality by facilitating storage and uptake of solutes (Li et al., 2020) while impacted streams have limited ability to do so.

1.2 Temperate Forest Stream Functioning

In temperate forested regions, unimpacted functioning streams exhibit several characteristics such as providing the necessary waters to support downstream ecosystems, the ability to regulate nutrient concentrations (not too much or too little (Alexander et al., 2007)), and support biologically diverse species. Perennial streams, characteristic of temperate forests, are streams constantly maintained by baseflow between precipitation events (Hornberger et al., 1998). Baseflow, the background low-flow discharge of a stream, is sustained year-round by adequate supplies of groundwater (Hornberger et al., 1998; Holden, 2014). Baseflow works to prevent stream drying and is necessary for maintaining an ecological baseline which sustains conditions needed for ecosystem functioning. Baseflow directly impacts the ability of aquatic and riparian species to survive and thrive, allows for the continuation of biogeochemical cycling throughout the year, and ensures that water is available for downstream ecosystems.

Generally, the water balance of a given system can be expressed as a change in storage equaling the inputs minus the outputs (Equation 1.1) in which the storage is the

groundwater, soil water, and vegetation-held water within the watershed itself, inputs are precipitation, and outputs are evapotranspiration, deep groundwater recharge, and surface runoff. Within this typical headwater watershed in an average year, it can be expected that the annual average change in storage is equal to zero. This is because it is presumed that precipitation is the only input and that the total amount of precipitation in the given watershed will leave as either evapotranspiration or surface runoff. The transient water balance during and after restoration becomes more complex in watersheds that have a history of heavy stream incision; this concept will be discussed in more detail in Chapter 2.

Within headwater watersheds, the movement of water can be substantially more complex and bidirectional than the overall watershed-scale “pipe” model implied by the watershed water balance (Bencala, 1993; Bencala et al., 2011). In a unimpacted, functioning system, potential flow paths include, but are not limited to, lateral subsurface flow in the hyporheic zone, flow from the hyporheic zone into the stream (gaining stream), flow from the stream into the hyporheic zone (losing stream), stream flow from higher to lower elevations (Figure 1.1), flooding onto the surrounding floodplain, and hydrologic retention on multiple time and spatial scales (Bencala, 1993; Bencala et al., 2011). Hyporheic exchange is the movement of water between the subsurface and surface waters at the bank and bed interface and occurs in streams where groundwater mixes with surface water (Tonina & Buffington, 2009; Hester et al., 2017). Groundwater can either be an input or an output to surface water with flow direction changing based on a variety of hydrological (e.g.: changes in head pressures) and morphological (e.g.: meander bends, bedforms) differences along the stream (Tonina & Buffington, 2009; Stonedahl et al.,

2012; Hester et al., 2017). Hyporheic exchange is an important component of nutrient cycling within streams as many of the conditions needed for nutrient transformation and retention happen in the hyporheic zone.

$$\Delta \text{Storage} = \text{Inputs} - \text{Outputs} \quad \text{Equation 1.1}$$

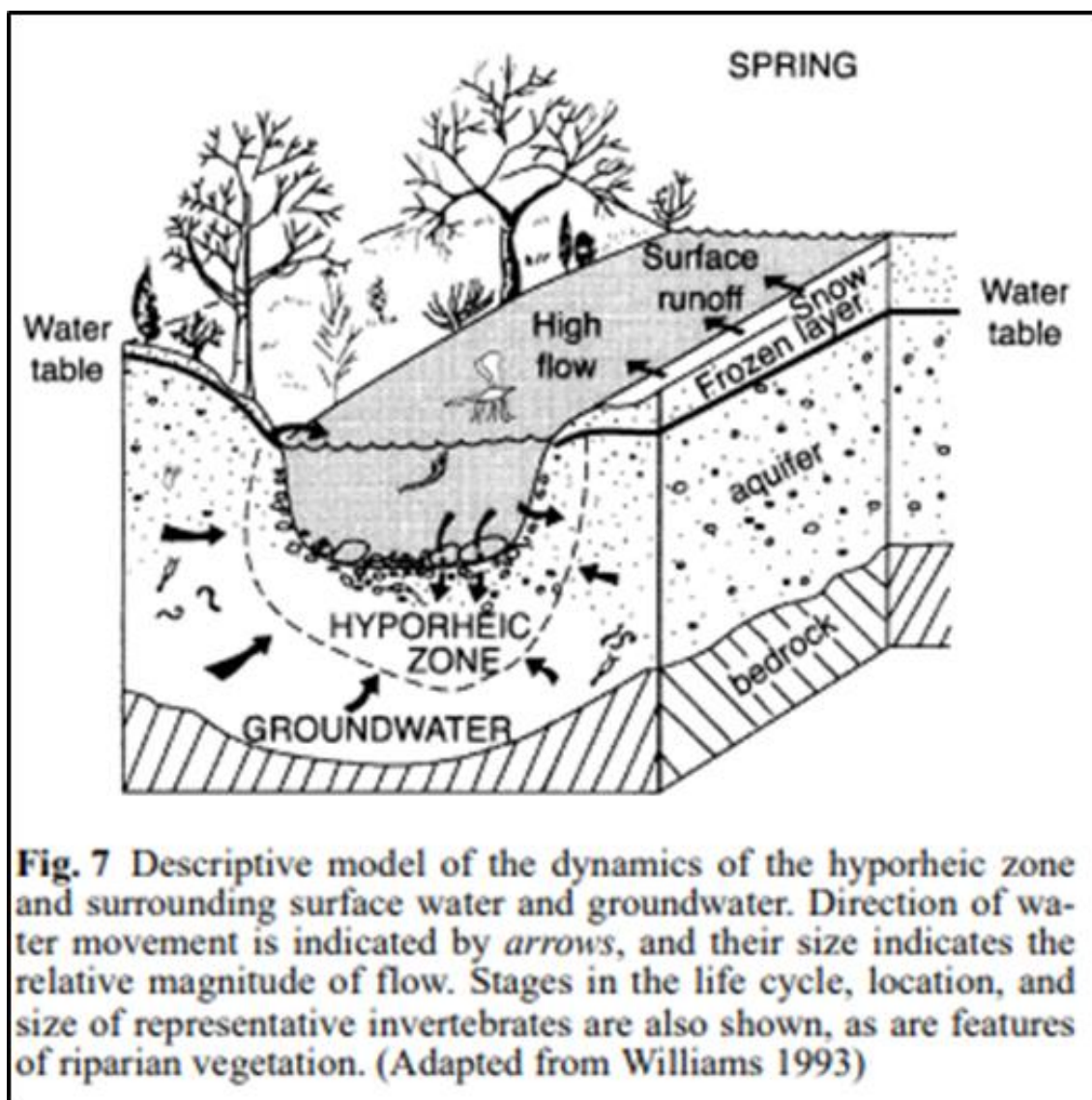


Figure 1.1 Flow and hydraulic exchange directions within a given watershed (Sophocleous 2002)

Whole system hydrologic retention, nutrient retention, and metabolism are regulated by groundwater-surface water (GW-SW) interactions within headwater

streams. Instream nutrient retention processes include storage via assimilation and/or adsorption and removal via transformation (Johnson et al., 2016). Nutrient retention processes are connected to hydrologic retention (i.e.: the ability of a system to slow and retain water before moving to a different reservoir) as the conditions needed for nutrient processing typically occur in areas of increased residence times. For instance, Valett et al. (1997) found that headwater geology, which controls GW-SW interactions, regulates whole-system nutrient dynamics. Depending on the surrounding geology, nutrient exchange from the stream to the hyporheic zone may either be impeded or aided. In cases where the lithology aids exchange, nutrient retention may be increased more than where lithology impedes exchange allowing nutrients to continue spiraling downstream.

Unimpacted headwater stream morphology allows nutrients to be transported from spatially diverse upstream locations to downstream locations where the nutrients become further concentrated (Alexander et al., 2007). Additionally, headwater stream morphology and characteristics (i.e.: sinuosity, riffles, pools, surface area to volume ratios, and woody debris) (Alexander et al., 2007) aids nutrient retention by slowing flows and allowing for more contact with surrounding hyporheic and riparian zones. The critical hyporheic and floodplain connections characteristic of unimpacted streams work to regulate nutrient concentrations by providing essential parameters needed for nutrient processing to occur (i.e.: denitrification, uptake and cycling of nutrients) (Valett et al., 1997; Alexander et al., 2007). In lotic systems (i.e.: streams), nutrient cycling occurs as dissolved nutrients are accumulated by organisms, converted to organic forms, and returned to the water as inorganic forms after organisms and materials degrade – these

inorganic forms are then taken up by downstream organisms which begin the process again (Schlesinger & Bernhardt, 2013).

Pre-European settlement Southern Piedmont watersheds may have begun exhibiting upland erosion with the cultivation of maize by Native Americans centuries before European settlers arrived (Reusser et al., 2015). Disturbance in pre-settlement Piedmont forests include severe weather such as hurricanes, clearing of land by Native Americans, and frequent wildfires caused by lightning (Cowell, 1995). The abundance of streams present within the Piedmont act as natural firebreaks creating the conditions needed for small Piedmont prairies that exhibited an abundance of non-woody plant species (Serrano, 2015). According to Dearman (2018) and Daniels (1987), pre-settlement alluvium suggests that sedimentation and erosion rates were lower than post-settlement rates. In some areas, evidence may suggest that pre-settlement sedimentation may have occurred in wetland complexes created by beaver dams (Dearman, 2018). During the 1800 to mid-1900s, intensive agriculture in the Piedmont led to upland erosion of hillslopes and deposition of sediments in the lowlands (Trimble, 2008). Agriculture in the Piedmont was unsustainable and detrimental to the land with farmers clear-cutting, growing crops, and then abandoning fields for more fertile lands upslope (Trimble, 2008). This led to poor soil quality upslope and deposition of eroded, poor-quality sediments downstream where streams became aggraded, groundwater levels rose, and lowland areas became too wet for agricultural purposes (Trimble, 2008). Moreover, the change from rich soils full of humus that could readily absorb water to poor quality soils led to increasing overland flow, erosion, creating gullies, and ultimately, degrading downstream ecosystems (Trimble, 2008).

Streams in urbanizing environments have been severely altered by humans on a global scale. Channelization, piping, burial, and impoundment of urban streams coupled with increases in impervious surfaces needed to meet human needs have altered the natural biogeochemical processes of streams (Freeman et al., 2007). Alterations of the physical stream coupled with various other anthropogenic land-use and land-maintenance trends (e.g.: increased imperviousness & chemical use) present increasing problems for the immediate urban area (e.g.: flooding & decreased water quality) as well as downstream ecosystems (e.g.: less & more nutrients & increased stormflows) (O'Driscoll et al., 2010; Hibbs & Sharp, 2012). In the immediate urban area, the altered stream exhibits changes that include increased incision, flashy hydrographs, reduced infiltration, increased overland flow, and increased urban pollutants (e.g.: sewage, nutrients, & sediments) (Walsh et al., 2005; O'Driscoll et al., 2010). Increased incision occurs when urban stormflow enters streams within a shortened timeframe and at a higher velocity than in natural systems, effectively scouring the channel bed and banks (Walsh et al., 2005). Higher quantities of urban stormflow with shorter duration to reach the stream are characteristics of a flashy hydrograph. Streams that exhibit this sudden rise in stage are considered flashy streams. Furthermore, reduced infiltration in urban areas leads to an increase in overland flow with stormwater being able to acquire various types of urban pollutants where they enter and are deposited into the stream (Walsh et al., 2005; O'Driscoll et al., 2010). Streams located within urban areas exhibit altered biogeochemical processes such as the ones mentioned above. Collectively, these changes are known as the urban stream syndrome (USS).

1.3 Urban Stream syndrome

Climate change is expected to bring variable storm events with irregular precipitation amounts (Famiglietti & Rodell, 2013) which may alter the availability and quality of water in the future:

Local and regional changes in the character of precipitation depend a great deal on variability in patterns of atmospheric circulation. Some of these observed circulation changes are associated with climate change. A shift in the storm track makes some regions wetter and some—often nearby—drier, leading to complex patterns of change (Famiglietti & Rodell, 2013, p129).

This variable climate, coupled with a fast-growing population and the environmental changes those people cause, set the stage for increasing freshwater stream degradation. The effects of a changing climate and increasing anthropogenic environmental degradation can be readily seen altering stream hydrology, morphology, and chemical processing. Currently, the majority (55%) of the world's population resides in urban areas with more (+13%) predicted to move to urban areas by 2050 (United Nations, 2018). This expansion of human populations in urban areas brings with it increases in demand for places to reside and shop, as well as expansions in roadways and manicured outdoor spaces leading to the addition of impermeable surfaces. Increases in urbanization coupled with predicted variability in intensifications of the frequency and duration of extreme events (Trenberth, 2011; Famiglietti & Rodell, 2013) can cause severe issues for streams and the watersheds within which they reside.

Hydrological, morphological, and chemical degradation become further exacerbated when streams are located in these urban settings; thus, leading to streams being classified as exhibiting the USS. Streams exhibiting these characteristics must

undergo extensive restoration to return the systems to an unimpacted state. Unfortunately, restoration has not been the first approach to offset anthropogenic degradation to urban streams with urban centers adding infrastructure such as retention ponds and stormwater conveyance structures which alter the natural functioning of the ecosystem. For instance, stormwater conveyance structures funnel impervious surface runoff to retention ponds or directly to the stream leading to flashier streams and altering the location of groundwater recharge. Unimpacted systems have the ability to adapt to changing climate as they have in the past. However, urbanization has reduced the ability of these systems to adapt and, therefore, has had a more substantial effect on stream form and function than changes in climate alone.

Walsh et al. (2005) characterized the USS as altered hydrology, increased temperatures, altered water chemistry, and changes in stream biology. In urban areas, an increase in population density exacerbates the USS and facilitates overland flow, rather than infiltration, leading to increasingly flashy hydrographs (i.e.: larger, more frequent flows that exhibit faster ascending and descending limbs). Walsh et al. (2005) classified flashy flows as those with the potential to erode stream banks, disturb the biota, and cause channel incision. Morphological changes occur via increasing flows and erosion leading to increasing channel width, increasing the depth of pools, scouring of streambeds, and a decreasingly complex channel (Walsh et al., 2005) (Figure 1.2; Figure 1.3). These alterations in channel morphology led to a decrease in floodplain connectivity, reducing the ability of the stream to interact with the subsurface, and reducing nutrient retention and transformation (Figure 1.3). Additionally, the USS leads

to decreased baseflow impacting local and downstream ecosystem functions (Walsh et al., 2005; O'Driscoll et al., 2010; Hibbs & Sharp, 2012).



Figure 1.2 Natural Stream Ecosystem. Adapted from Bell et al. (2012).



Figure 1.3 Urban Stream Ecosystem. Adapted from Bell et al. (2012).

Changes in water chemistry are symptoms of the USS. For example, according to Meyer et al. (2005), increasing concentrations of ammonium (NH_4^+) were seen in urban streams when compared to natural forested streams due to increased input from point and non-point sources. Moreover, Meyer et al., (2005) also found that the uptake rate of NH_4^+ and soluble reactive phosphorus (SRP) decreases with increasing urbanization. Walsh et al. (2005) state that even watersheds with limited urbanization exhibit increased chemical concentrations from sources such as waste water and sewage runoff (Lavelle et al., 2019; Reisinger et al., 2016; Zhou et al., 2017). In a study by Moore et al. (2017), changes in stream chemistry were seen along a forested to urban gradient with major ion concentrations (e.g.: Ca^{2+} , Mg^{2+} , Na^+ , Cl^- , and SO_4^{2-}) increasing on average 27 times along that gradient. Some of the mechanisms that work to alter the chemistry of urban streams include the use of road salts, weathering of impervious surfaces, mobilization of ions from soils to streams, and land development (Kaushal et al., 2014, 2017; Bird et al., 2018).

1.4 Stream Restoration

With their importance for regulating ecosystem functions, it is essential that headwaters become a priority for restoration and management. Restoration of headwater streams has the ability to maintain the retention of nutrients and organic matter (Wohl, 2017; Thompson et al., 2018). Furthermore, restoring headwater streams works to ensure downstream ecosystems have reduced total suspended solids (TSS) inputs (i.e.: maintaining downstream water quality), the necessary supply of nutrients, and waters to warrant proper ecosystem functioning. In urban systems, it is necessary to restore stream processes impacted by human activities (Violin et al., 2011). For instance, reconnecting

streams with historic floodplains provide conditions needed to increased residence times of floodwaters and increases the potential for nutrient exchange (Valett et al., 1997; Alexander et al., 2007). Revegetating riparian buffers and floodplains aids in erosion control and nutrient retention (Jansson et al., 2007) within the watershed by increasing residence times of stormflows. Restoring stream morphology from the characteristic channelized streams to more natural meandering streams provides the conditions needed to support habitats for various aquatic species (Wohl, 2017). Sinuosity aids hyporheic exchange of nutrients and increased residence times provide conditions needed for denitrification (Gomez-Velez et al., 2017).

There are several motivations to restore degraded streams including: mending channel morphology, improving water quality, increasing biological resilience, meeting social values, an increasing need for knowledge, to bring back functioning for personal needs or desires, and to meet the needs of consumers (e.g.: animals & humans) as well as to offset anthropogenic damage (Clewett & Aronson, 2006; Sudduth et al., 2007). According to Clewett & Aronson (2006), restoration tends to occur in silos with a single objective driving the process; successful restoration, however, requires multiple motivations be employed. Moreover, Bernhardt et al. (2005) compiled information from ~37,000 restoration projects and determined that the majority of those projects were small in scale (i.e.: less than one km of stream length being altered) and that nearly 20% of projects assessed had no listed motivations for restoration. Sudduth et al. (2007) in a survey of 860 southeastern restoration projects found that the median stream length repaired was less than 2.5 km. Additionally, Sudduth et al. (2007) determined that 18% of projects within the Southeast had no motivations, 29% had only one project motivation,

27% had two motivations, and 22% had four or more motivations. According to Bernhardt et al. (2005), “the most commonly stated goals for river restoration in the United States are (i) to enhance water quality, (ii) to manage riparian zones, (iii) to improve instream habitat, (iv) for fish passage, and (v) for bank stabilization”. Additionally, it was found that the number one project goal in North Carolina stream restoration projects was to restore water quality followed by riparian management, land acquisition, and channel reconfiguration (Sudduth et al., 2007).

Current approaches to restoration tend to focus on restoring partial reaches to increase stream functioning or aesthetics (Hamilton, 2012) and take the form of bank stabilization projects, changing channel morphology, or using green infrastructure to reduce stormflows entering streams. Projects such as these are small in scale and usually address few ecosystem components while hydrologic function and ecosystem health can, and should, be viewed and assessed at a larger scale (Lewis et al., 1996; Hamilton, 2012). While small-scale restoration may work to lessen the ecosystem stressors, environmental degradation needs to be addressed at the large-scale to ensure long-term rehabilitation (Lewis et al., 1996; Lake et al., 2007; Stoll et al., 2016). To address the larger issue, whole-watershed stream restoration, such as was implemented in the Reedy Creek watershed, can be implemented to recreate a functioning ecosystem through the implementation of various methods. Instead of focusing on a single aspect that needs to be restored within the ecosystem, this approach facilitates combined efforts to recreate function. Projects may include but are not limited to reconnecting streams to historic floodplains that have been revegetated coupled with green infrastructure to reduce stormflow erosion capacity. When working in streams exhibiting USS, the best approach

for restoration would be one that encompasses and addresses the multiple factors contributing to degradation of the ecosystem. It is important to mention that for a project to be considered a watershed restoration project, upland components should be addressed as well. Here, instead of having a singular focus on small-scale stream channel alterations, watershed restoration should address the full extent of stream length contributing to the flow at the restored reach of focus and may also incorporate restoration of terrestrial riparian habitats.

Practitioners within the southeast United States tend to base their designs on the Rosgen natural channel design (NCD) approach (McMillan et al., 2014). The NCD approach uses three reach types, the existing, reference, and potential (Rosgen, 2011). The existing reach is the reach set to undergo restoration. The reference reach is not necessarily pristine but is a reach that is self-sustaining and self-maintaining spatially and temporally. The potential reach is the reach that is designed with characteristics (e.g.: morphology, sediment regimes, hydraulic, and biologic, etc.) of the reference reach (Rosgen, 2011). The NCD considers the fluvial processes exhibited by self-maintained and self-sustained systems over space, time, and successional stages; the approach inherently recognizes that systems are dynamic and change over time (Rosgen, 2011). The NCD is an adaptive approach to restoration that allows for new stream designs to function within larger landscapes and with the conditions those landscapes exhibit (Rosgen, 2011).

1.5 Study Context and Overarching Objectives

Charlotte, North Carolina is built on a network of headwater streams along the watershed divide between the Catawba and Yadkin-Pee Dee river systems. Between

2003-2016, Charlotte-Mecklenburg Storm Water Services actively improved ~48 km of streams (Charlotte-Mecklenburg Storm Water Services (CMSWS), 2021b). The majority of waters within Mecklenburg county, including those within the Reedy Creek watershed where this study has taken place, have been listed as impaired (City of Charlotte, 2022)). Stream degradation within the county is attributed to stream bank erosion from historical stormwater management, the channelization of streams for agricultural purposes, and erosion driven by historical and current increases in impervious cover. Population growth continues to increase the risk to streams now and into the future. According to the US Census Bureau (2019b), the population of the state of North Carolina grew by nearly one million inhabitants from 2010-2019 ranking fourth out 50 states for population increase. Between 2017-2018, Charlotte was fifth in the nation for population growth (US Census Bureau, 2019a) with a growth of ~13,200 individuals. Increases in population density bring with them increased in impervious surfaces. According to the Quality of Life Explorer (2020), impervious surfaces within Mecklenburg county increased 1.4% from 2011-2018. To offset and counteract the degradation caused by historical stream management and current/future population growth, Charlotte-Mecklenburg Storm Water Services has ~26 restoration projects ongoing or nearing completion (Charlotte-Mecklenburg Storm Water Services (CMSWS), 2021b).

This study focuses on the headwaters of the Reedy Creek watershed located in Northeast Charlotte, North Carolina (35°15'32.0"N 80°42'42.4"W; Figure 1.4). The watershed consists of five sub-watersheds draining areas with differing land-use characteristics (historically agriculture, developed, deciduous forest; Figure 1.4; Table 1.1) and has a drainage area of 6 km². Throughout the study monitoring period, land-uses

remained the same within the watershed. The watershed consists of 12 km of stream length. From 2015 -2021, mean yearly precipitation in the watershed was ~1285 mm (US Geological Survey, 2022b). From 2015-2021, the mean yearly temperature of the region was ~16°C (State Climate Office of North Carolina, 2021). The Reedy Creek watershed is in the Piedmont region of North Carolina (Figure 1.4) and is part of the Yadkin-Pee Dee river basin which consists of over 18,000 km² of drainage area. The Yadkin-Pee Dee river basin is the second largest in the state.

The watershed consists of a housing development (Figure 1.5, Development) and land that historically was cleared for agricultural uses (Figure 1.5, Agricultural) which have led to the watershed exhibiting characteristics of the USS. USS characteristics caused pre-restoration tributaries to have flashy hydrology that led to erosion and incision of stream banks and beds (Figure 1.6). The incised streams flowed over sand, saprolite, gravel beds, and bedrock. Historic channelization and straightening of streams within Charlotte for agriculture most likely contributed to incision.

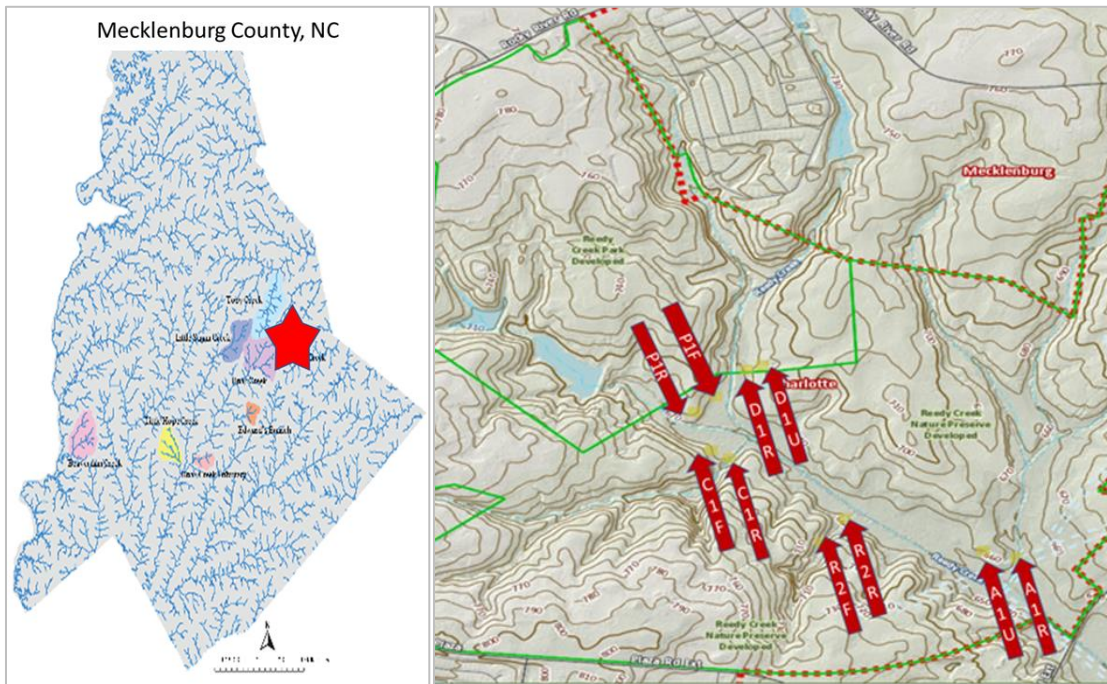


Figure 1.4 Location of the Reedy Creek headwater watershed within Mecklenburg County, North Carolina (D. Vinson, unpublished document) as well as approximate well locations within Reedy Creek headwater watershed (modified from Mecklenburg County Polaris, 2022).

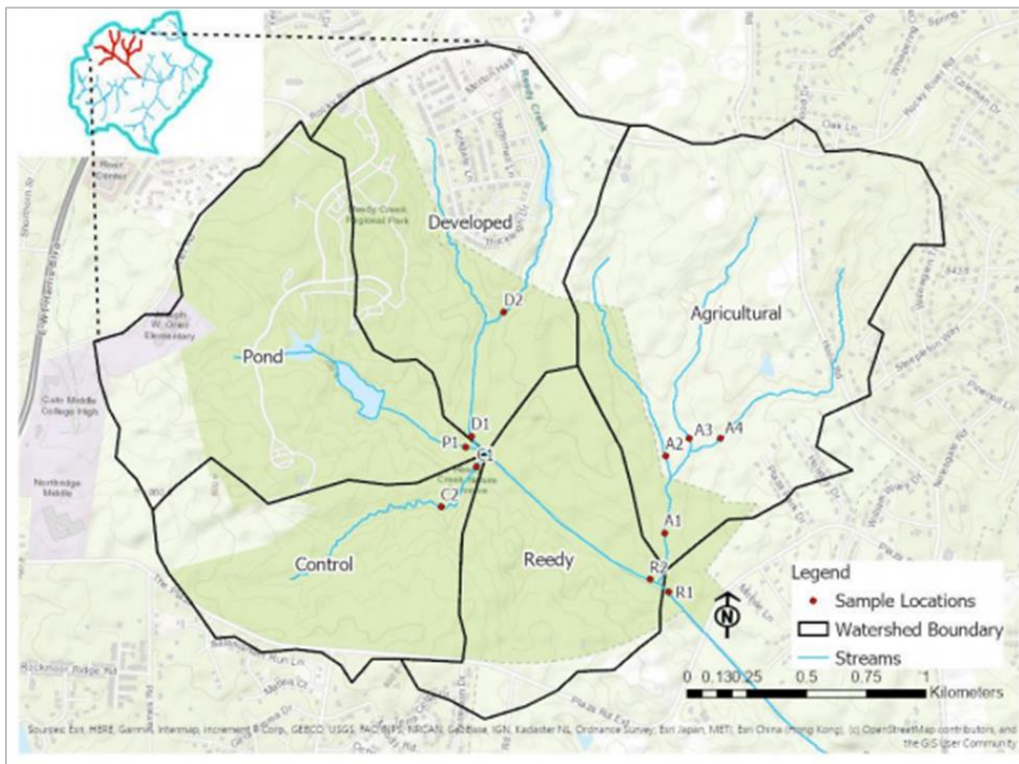


Figure 1.5 The Reedy Creek headwater watershed with land-use types and surface water reaches (J. Hartman, UNC Charlotte, unpublished map).



Figure 1.6 Pre-restoration stream conditions in A) R1 reach, B) C1 reach, and C) D2 reach

With heavily incised streams, the Reedy Creek watershed was determined to be a good candidate for restoration. Restoration broke ground in late December 2017 and continued into early January 2019. The restoration was classified as a priority one restoration with changes in tributary morphology and reconnecting the tributaries to their floodplains (Doll et al., 2021). Priority one restoration projects emphasize reestablishing stream channels on existing floodplains by construction of new channels based on reference channel pattern and profiles (Rosgen, 2011). Old channels were backfilled to an elevation level with the historic floodplain (Rosgen, 2011). The project goals for this restoration were to: 1) stabilize banks, 2) restore floodplains to natural function, 3) return streams to a more natural pattern, 4) improve habitat, 5) enhance water quality, and 6) reduce pollution in Reedy Creek and its tributaries. Monitoring has taken place in the watershed since 2013 at each tributary within the five sub-watersheds (Figure 1.5) (McMillan & Clinton, 2013). Each sub-watershed was monitored for surface water, standard water quality parameters, and stream chemistry. Furthermore, each sub-watershed has a riparian well and upland well where groundwater levels, groundwater temperature, and groundwater chemistry was monitored.

Restoration began at the end of December 2017 at the uppermost reaches of the Reedy Creek headwater watershed (Table 1.1; Figure 1.4, D2). Two years of pre-restoration monitoring was carried out for each site from the restoration start dates. Additionally, two years post-restoration was carried out for each site from the restoration end dates. These two dates, two years PRE and two years POST, are used to define the comparison period for each site. The timeframe from two years PRE to Start Restoration dates were used to define the pre-restoration period for each site. The timeframe from

two years POST to end restoration dates were used to define the post-restoration period. The study was bound between the years of 2015 and 2021 (Table 1.2).

Three overarching objectives aim to better understand the efficacy of stream restoration projects to meet their stated goals. In Chapter 2, I quantified changes in groundwater from pre-restoration levels to post-restoration levels (Objective 1). To answer research questions within this study, a set of groundwater monitoring wells at sub-watershed outlets were monitored since 2016 (Figure 1.4 labeled sites). In Chapter 3, I quantified changes in baseflow nitrogen from pre-restoration levels to post-restoration levels (Objective 2). To answer research questions within the study, baseflow surface water samples were collected since 2016 at the watershed outlet (Figure 1.5, R1). In Chapter 4, I quantified changes in baseflow and stormflow total suspended solids from pre-restoration levels to post-restoration levels (Objective 3). To answer research questions within the study, baseflow and stormflow surface water total suspended solid samples were collected since 2016 at the watershed outlet (Figure 1.5, R1). All objectives aim to better understand the efficacy of stream restoration projects to meet their stated objectives (objectives 2 & 3) as well as investigating changes that may occur that are not stated as restoration goals (objective 1).

Table 1.1 Land-use characteristics and watershed area of each sub-watershed. Yellow highlights indicate tributaries with stream-well transects.

Site	Land-use	Watershed Area (km ²)	Start Restoration	End Restoration	Two years PRE	Two years POST
D2	Development	0.27	12/06/2017	05/06/2018	12/06/2015	05/06/2020
D1	Development	1.15	12/20/2017	07/08/2018	12/20/2015	07/08/2020
P1	Pond Influenced	1.18	06/13/2018	09/10/2018	06/13/2016	09/10/2020
C1	Deciduous Forest	0.77	06/12/2018	07/08/2018	06/12/2016	07/08/2020
C2	Deciduous Forest	0.68	CONTROL	CONTROL	CONTROL	CONTROL
A4	Historical/Current Agriculture	0.55	02/06/2018	06/07/2018	02/06/2016	06/07/2020
A3	Historical/Current Agriculture	0.6	03/31/2018	06/07/2018	03/31/2016	06/07/2020
A2	Historical/Current Agriculture	0.48	02/06/2018	04/05/2018	02/06/2016	04/05/2020
A1	Historical/Current Agriculture	1.9	01/19/2018	07/08/2018	01/19/2016	07/08/2020
R2	Mainstem	3.69	08/22/2018	12/13/2018	08/22/2016	12/13/2020
R1	Mainstem (watershed outlet)	5.7	12/19/2018	02/15/2019	12/19/2016	02/15/2021

Table 1.2 Restoration dates for the pre, during, and post-restoration periods used for statistical analysis for this study.

Restoration Period	Start Date	End Date
Pre-Restoration	December 2015	November 2017
During Restoration	December 2017	December 2018
Post-Restoration	January 2019	February 2021

2 WHOLE-WATERSHED STREAM RESTORATION-INDUCED CHANGES ON GROUNDWATER STORAGE AND SEASONAL WATER TABLE CYCLING

2.1 Introduction

Groundwater is an essential component of the hydrologic cycle and is recharged by infiltrating soil water, subsurface stormflows, and groundwater-surface water interactions (GWSWIs) (Winter et al., 1998; Healy & Scanlon, 2010; Holden, 2014). Groundwater makes up nearly 30% of all freshwater on Earth (Hornberger et al., 1998; Griebler & Avramov, 2015) making it a vital reservoir for meeting hydrologic and ecologic needs as well as the needs of ~20% of the world's population (Famiglietti, 2014; Holden, 2014; Griebler & Avramov, 2015). According to Holden (2014), roughly 10% of groundwater is consumed unsustainably while providing one-half of all irrigation water (Famiglietti, 2014). This unsustainable use, coupled with climate change, stream incision, and increasing demands from population growth has strained groundwater supplies and led to increasing conflicts over water resources (Famiglietti, 2014; Holden, 2014). Groundwater supplies become further threatened in watersheds that exhibit degradation such as streams becoming disconnected from surrounding floodplains and increased incision of stream channels. Alteration of groundwater at a global scale calls for effective management plans and practice (Healy & Scanlon, 2010). For instance, whole-watershed stream restoration may be used to offset degradation and incision of streams and, ultimately, lead to an increase in groundwater storage potential. Here, this study aims to determine the

efficacy of using the whole-watershed stream restoration approach to increase groundwater levels in the Reedy Creek watershed.

2.1.1 Groundwater's Role in Maintaining Stream Ecosystems

Groundwater plays a vital role in maintaining stream and fluvial ecosystems (Woessner, 2000). These ecosystems are defined as groundwater-dependent ecosystems and are responsible for maintaining ecological communities and the biophysical processes needed for the ecosystem to thrive (Kath et al., 2018). Groundwater is an essential component of ecosystem functioning and provides a plethora of ecosystem services such as maintaining baseflow, the transformation and retention of nutrients, retention and storage of water, flood buffering, and various cultural services (Griebler & Avramov, 2015; Kath et al., 2018). Hyporheic exchange is the physical process, occurring within the banks and beds of streams, by which surface waters and groundwater mix (Hester & Gooseff, 2010; Hester et al., 2017). This process supports many ecosystem services (Mendoza-Lera & Datry, 2017) and, therefore, is an important mechanism by which water quality is regulated and is essential for supporting invertebrate diversity.

2.1.1.1 Groundwater's Role in Physical Hydrological Processes

Typical forest streams within the Piedmont region can be characterized by their pool/riffle/run sequences (N.C. Stream Functional Assessment Team, 2013). These streams tend to be moderately sinuous, have moderate gradients, exhibit substrates with fine to moderate coarseness (Bisson et al., 2006), are less constrained than mountain streams (Tonina & Buffington, 2009), and, in the recent past, have become increasingly incised (Wondzell & Swanson, 1999) due to various anthropogenic alterations of land

use. Human impacts have further altered water quality with the addition of nutrients from urban/agricultural runoff.

The areal extent and degree of groundwater-surface water interactions within Piedmont streams are regulated by the stream characteristics mentioned above. Pool/riffle sequences exhibited by the streams enhance bed hyporheic exchange through head gradient changes (Hester & Gooseff, 2011). Sinuosity of these streams, though moderate, increases lateral hyporheic exchange as stream water enters the hyporheic zone along the outside of the bend where it comes into contact with biologically active sediments and eventually leaves the hyporheic zone and reenters the stream at the beginning of the following bend (Hester & Gooseff, 2011). The moderate gradients exhibited in these channels reduces the vertical mixing when compared to mountain streams. Moreover, these streams also exhibit fine to moderate substrates which are not as successful at changing head gradients as the substrates (i.e.: larger substrates such as cobbles) found in mountain streams. These streams are less confined than mountain streams allowing them to have an increased spatial impact on the surrounding floodplains (Tonina & Buffington, 2009). Incision, which is a characteristic of streams within the Piedmont region that has experienced continuous land-use changes, has been shown to decrease the areal extent of hyporheic mixing (Wondzell & Swanson, 1999).

Biological and biogeochemical processes rely heavily on hyporheic exchange in these streams. Having a functioning and connected hyporheic zone is vital in these streams as excess nutrients are present in these streams from urban/agricultural runoff. Moreover, these streams tend to have an adequate supply of sediment (Bisson et al., 2006) available for biochemical transformations. However, according to Lenat &

Crawford (1994), soils within this region are easily eroded which can lead to clogged beds, reduced hyporheic exchange, and have adverse impacts on aquatic invertebrates.

Hydraulic conductivity is the complex relationship between hydraulic gradient, discharge, and the porosity and permeability of the material through which water moves (Hornberger et al., 1998; Bierman & Montgomery, 2014; Holden, 2014). In other words, it is the ability of water to move through a porous material (Hornberger et al., 1998). The hydraulic conductivity of a given area is directly related to the geology of that area. For instance, the hydraulic conductivity of a stream dominated by gravel/sand would be higher than in a stream dominated by clay beds as porosity and permeability of gravel/sand is higher than clay (Bierman & Montgomery, 2014). The natural bed substrate of these streams consists of gravel (Bisson et al., 2006), cobble, and sand (N.C. Stream Functional Assessment Team, 2013). According to Winter et al. (1998), these streams tend to be gaining.

In the Piedmont region where incision is intensive due to increased urbanization, heavily incised channels have little to no exchange with their hyporheic zones. In extremely incised channels, incision may have reached bedrock or clay-rich weathered bedrock (saprolite), which effectively stops vertical hyporheic exchange as bedrock has low/no permeability. In cases such as this, the stream bed is no longer biologically active. Incision also scours the stream bed creating a homogenous stream bed with little to no hydraulic head changes. Incised streams tend to be channelized removing any possibility of bank exchange. These streams are inherently flashy, reducing the time waters may be in contact with incised banks. Moreover, if incision is severe enough, floodplain inundation is impossible which reduces the areal extent of hyporheic exchange.

Groundwater aids streamflow and maintains baseflow conditions necessary for aquatic life, especially in gaining streams (Winter et al., 1998; Figure 2.1A).

Groundwater can also be impacted by streamflow with losing streams contributing to groundwater (Winter et al., 1998); Figure 2.1B). Within the Reedy Creek watershed, streams tend to be net gaining due to the humid climate with streams relying on groundwater for flow. Groundwater also plays an important role in maintaining wetland functioning as wetlands exist where the groundwater level and wetland surface hydrology are uniform (Devito et al., 1996). Moreover, groundwater levels are also related to flooding in that as floods occur, water moves from the stream channel into the banks (bank storage) and can result in transient increases in groundwater level within the watershed (Winter et al., 1998).

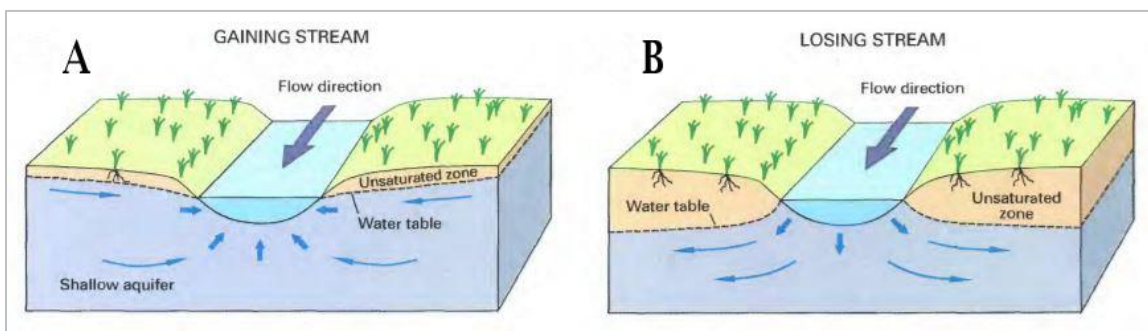


Figure 2.1 Cross-sectional view of gaining (A) and losing (B) streams. (Winter et al., 1999)

Within groundwater-dependent ecosystems, groundwater that entered the watershed as precipitation throughout the year is slowly discharged into the stream (Winter et al., 1998). In humid regions, this slow discharge maintains stream baseflow (Kath et al., 2018;) throughout the drier months. Moreover, groundwater recharge contributes to hydrologic retention and longer watershed-scale residence times. Hydrologic retention and transient storage refer to the short term storage of water within the stream channels themselves or the streams' hyporheic zones (Bukaveckas, 2007).

Transient storage and retention are important as they increase the time waters may be in contact with surrounding sediments, thus increasing the potential for interactions needed for nutrient transformations as well as storing flood waters and decreasing the energy of stormflows which work to reduce stream channel degradation.

Shallow groundwater is an essential component of hyporheic exchange (Woessner, 2000; Tonina & Buffington, 2009; Hester et al., 2017; Magliozzi et al., 2018) and is necessary to ensure functioning of biogeochemical processes such as maintaining floodplain carbon and nitrogen exchange, altering hydrologic residence times, denitrification, and nutrient transport from soils to streams (Wallis et al., 1981; Jansson et al., 2007; Harvey & Gooseff, 2015; Johnson et al., 2016; Yabusaki et al., 2017; Magliozzi et al., 2018). Hydrologic retention, in the form of hyporheic exchange, is essential for nutrient processing as the conditions needed for processes such as denitrification exist in regions exhibiting low oxygen levels and increased retention times (Grimm et al., 2003; Durand et al., 2011; Zarnetske et al., 2011b; Xia et al., 2018). Furthermore, stream features such as pools also provide the conditions needed for increasing hydrologic retention (Bukaveckas, 2007).

2.1.1.2 Groundwater's Role in Water Quality

The quality and availability of groundwater is important for stream water quality and ecosystem health both in the chemistry of groundwater inputs to streams from surrounding landscapes and the role that groundwater plays in processing nutrients from downwelled stream water. Nutrients can be sourced from a variety of landscapes and land-uses including urban and suburban areas, agricultural areas, urban-forested areas, and more natural forested areas. Water that flows over these areas picks up nutrients and

delivers them to nearby streams via overland flow or delivers them to groundwater sources via infiltration and percolation. Here, groundwater acts as a buffer for floodwaters and the nutrients those floodwaters contain by slowing flows, increasing hydrological retention times, and providing the conditions needed for nutrient processing. Nutrient processing such as denitrification could happen more easily in the hyporheic zone where water has downwelled than instream where conditions are not ideal.

Holistically, the quality of groundwater is directly related to the physical, chemical, and biological attributes of the groundwater basin. Physical components of groundwater include the volume, speed, and direction of groundwater flow which work together to dictate discharge patterns, stream chemistry, and hyporheic exchange within a given reach. The quality of groundwater is also related to the groundwater-surface water interactions within the watershed. Movement of water between the surface and subsurface creates conditions needed for nutrient processing (Tonina & Buffington, 2009; Hester & Gooseff, 2010; Hester et al., 2017). For instance, hyporheic exchange between groundwater and stream channels could create the conditions needed for the instream processing of nutrients by preventing the stream from drying, moderating stream water temperatures, and increasing subsurface denitrification. When nutrients are in excess of stream processing capacities, pollution becomes a threat to groundwater and ecosystem health as chemicals added to soil leach into the subsurface and make their way downstream and negatively affect downstream ecosystems (Bennett et al., 2001). Groundwater is also biologically active and contains organisms from bacteria to viruses and protozoans to macroinvertebrates (U.S. EPA, 2008; Stein et al., 2010). Groundwater organisms, such as nitrifying and denitrifying bacteria (Pinder et al., 2013), are

considered to be involved in chemical processes (e.g.: breaking down of organic matter, chemical cycling of nitrogen, sulfur, and iron) and are extremely sensitive to changes (e.g.: changes in oxygen and redox conditions, pollution, heavy metals, and organic matter) in their environments (Robertson et al., 2009; Stein et al., 2010; Korbel et al., 2022). Little is known about how these groundwater organisms respond to being stressed with research indicating they may be sensitive to groundwater abstraction and may increase in abundance and decrease in diversity due to the addition of nutrients (Robertson et al., 2009). It can be assumed that if stressed, these organisms may not be able to complete the processing of chemical constituents or may process chemical constituents in excess which may lead to degraded groundwater quality. If the physical, chemical, and biological components of groundwater are unstressed, the ecosystem should be able to regulate itself without the need of human interference.

2.1.1.3 Groundwater's Role in Watershed Ecosystem Services

Groundwater is a fundamental source of freshwater that impacts the functioning of the critical zone by interacting with watershed ecosystem services such as food production, regulating water quality and quantity, flood buffering, groundwater-surface water interactions, maintaining stream baseflows, and nutrient cycling (Griebler & Avramov, 2015; Mendoza-Lera & Datry, 2017; Kath et al., 2018; Griebler et al., 2019; Qiu et al., 2019). For example, the quantity of groundwater may elicit different responses within the ecosystem such as shallow groundwater availability increasing food yields as water is available to plant roots during dry years but conversely, during wet years, plant roots may become stressed due to a decrease in root oxygen availability (Soylu et al., 2014; Zipper et al., 2015; Qiu et al., 2019). Moreover, groundwater is important for the

buffering of floodwaters as wet soils are able to absorb more precipitation than their dry-soil counterparts (Griebler et al., 2019). Essential groundwater ecosystem services are far-reaching and are at risk due to climate change, anthropogenic overuse, and contamination (Griebler et al., 2019). Groundwater also has the potential to dampen climate change effects via the sequestration of inorganic carbon (Lal, 2004).

Groundwater plays an important role in maintaining riparian phreatophyte flora (Kath et al., 2018) which are essential for supporting nutrient and sediment dynamics and reducing erosion (Jansson et al., 2007). Ensuring there is an adequate supply of groundwater for riparian plants is imperative as plants in and along streams provide a buffer for overland flow, help aid in bank stabilization, and control sediment and nutrient dynamics (Jansson et al., 2007). Jansson et al. (2007) found that groundwater discharge areas were able to support more riparian plant species at higher densities than non-discharge areas. Without an adequate supply of groundwater to maintain baseflow, aquatic species life cycles would be disrupted (Winter et al., 1998).

Groundwater is also important for regulating and maintaining hyporheic organisms (e.g.: microbes and fungi) which directly impact nutrient processing of carbon, nitrogen, and phosphorus (Marmonier et al., 2012; Griebler & Avramov, 2015; Griebler et al., 2019). These groundwater organisms are essential as they provide bioturbation of the subsurface creating interstitial pathways which increase soil permeability allowing for nutrient cycling (Griebler et al., 2019).

2.1.2 Watershed-scale Water Balance

Unimpacted Systems - Section 1.2 briefly discussed the water balance of an unimpacted forested watershed. In a small-forested watershed with functional GW-SW

interactions, it is expected that there is no change in water storage during an average year as all inputs leave the watershed as outputs (Equation 1, Figure 2.2A). For groundwater, as Figure 2.2A illustrates, precipitation allows for groundwater recharge (input) via infiltration through the soil zone and percolation reaching the water table (Healy & Scanlon, 2010). Precipitation can also flow as subsurface stormflow through the unsaturated zone towards the stream and across the surface as overland flow. Once stored as groundwater, water can flow in a variety of paths and become outputs. Outputs for groundwater include evapotranspiration, subsurface and groundwater flows out of the watershed, and baseflow discharge as spring and stream flow (Healy & Scanlon, 2010; Hiscock & Bense, 2014) (Figure 2.2A). The watershed outlet has a singular output which is surface runoff, assuming that there is no subsurface flow to adjacent watersheds.

Studies by Hardison et al. (2009) and Rose & Peters (2001) provide evidence that groundwater in unimpacted systems exhibit water table levels that are shallower (Figure 2.2A) when compared to urbanized counterparts as these urban streams tend to be deeper (more incised) and have floodplains underlain by large unsaturated zones (Figure 2.2B). Moreover, this is evidence that unimpacted systems have increased groundwater storage and thinner unsaturated zones. In an unimpacted system, groundwater levels vary between seasons (i.e.: the growing season from April to September and the dormant season from October through March) with winter having a higher water table and summer having a lower water table (Zimmer & McGlynn, 2017) (Figure 2.2A). However, in both instances, the groundwater level is apparently able to maintain adequate baseflow to meet the ecological needs of the system. The depth of the unsaturated zone to the water table also plays an important role in groundwater recharge. For instance, in regions with

thin unsaturated zones, groundwater may infiltrate quickly in response to episodic events and may be readily influenced by transpiration (Healy & Scanlon, 2010; Hunt et al., 2018). Alternatively, thick unsaturated zones may be less likely to exhibit recharge responses to episodic events (Healy & Scanlon, 2010).

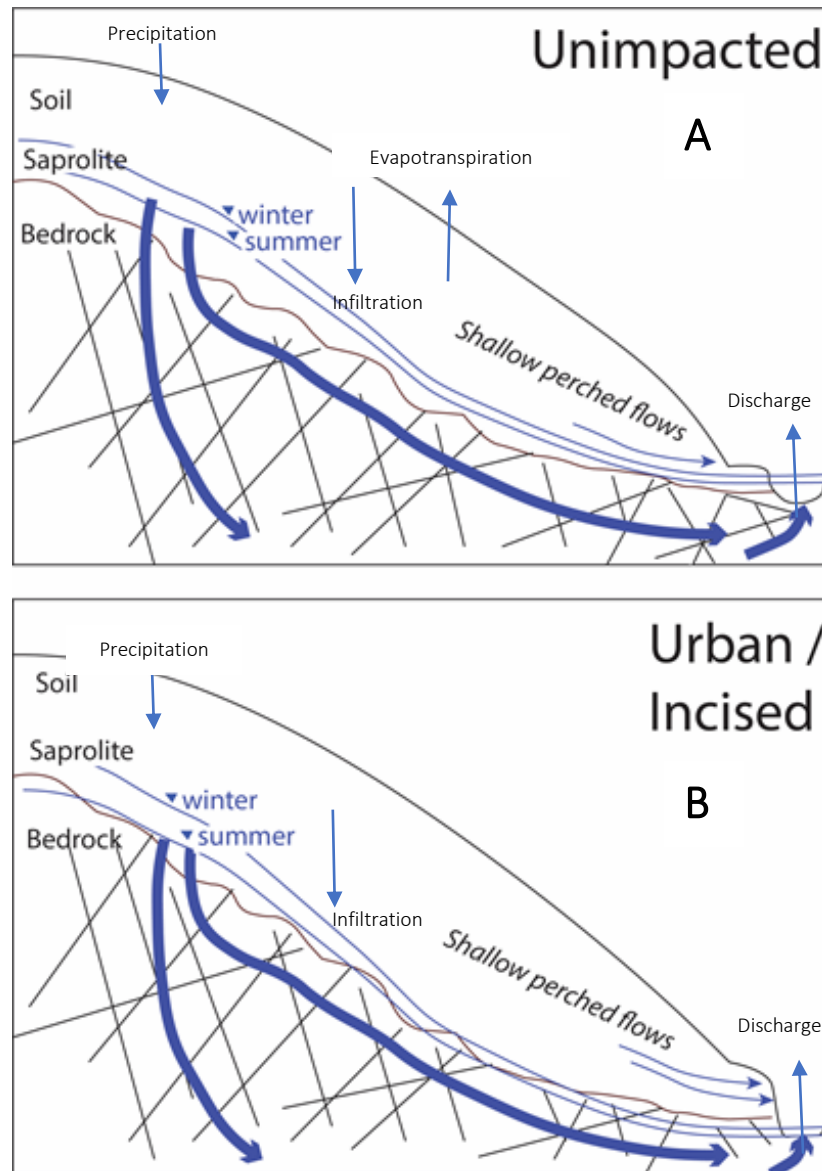


Figure 2.2 Diagram illustrating the water balance of a) unimpacted and b) urban/incised watershed. (modified from D. Vinson, unpublished document).

Impacted Systems – Within the Reedy Creek watershed, streams exhibited characteristics of degradation including extreme incision, disconnection of streams from

surrounding floodplain-like surfaces, and reduced GW-SW interactions. It is expected that as streams become more incised within impacted watersheds there will be less transient water storage as groundwater once stored in the watershed leaves as baseflow due to changes in hydraulic gradients causing a flushing of groundwater or a reduction in groundwater recharge (Hunt et al., 2018). According to Hunt et al. (2018), these changes in groundwater levels near incised channels are similar to groundwater pumping cones of depression with the stream itself acting as the pump. Impacted systems still have water inputs in the form of precipitation, but groundwater recharge and storage are limited to a narrower area as the impacted stream has a thick unsaturated zone due steep hydraulic gradients (Figure 2.2B). For instance, if a stream exists in its natural pattern where flood waters are able to inundate the surrounding floodplain, flood waters are able to access more area in which the potential for groundwater recharge increased (Healy & Scanlon, 2010).

Groundwater flow paths in impacted sites are typically more steep and deeper than in unimpacted systems. Outputs for groundwater include subsurface and groundwater flow and discharge as stream flow (Figure 2.2B). Due to the deeper water table, it can be assumed that transpiration may not impact the water table in the impacted system. Like in the unimpacted system, precipitation can flow as subsurface stormflow through the unsaturated zone towards the stream and across the surface as overland flow. Groundwater in impacted systems (Figure 2.2B) tends to exhibit water table levels that are deeper than their unimpacted counterparts (Figure 2.2A) (Rose & Peters, 2001; Hardison et al., 2009).

In a highly impacted system with deeply incised streams, water that is input in the system follows steeper and deeper flow paths than water flowing through an unimpacted system. However, in some instances, urbanized areas can have increased groundwater levels caused by leaking water pipes and irrigation (O'Driscoll et al., 2010). Much like in unimpacted systems, these impacted systems exhibit differences in groundwater levels between seasons with winter having a higher water table and summer having a lower water table (Zimmer & McGlynn, 2017) (Figure 2.2B). However, in contrast to unimpacted systems, impacted systems may exhibit larger fluctuations in groundwater levels with levels in summer being too low to maintain consistent baseflow (Figure 2.2B) and an overall decrease in groundwater storage within the system.

2.1.3 The Impacts of Restoration on Groundwater and Hyporheic Dynamics and their Role in Hydrologic Retention

When considering the water balance of a restored watershed, the system has the potential for a change in groundwater levels as old, incised channels that once acted as a groundwater drain are backfilled and new channels are formed. Generally, urbanization and historic land-use practices, such as are seen in the Reedy Creek watershed with a switch from forested to agricultural or forested to urban landscapes, have led to a decrease in baseflow by changing plant composition and increasing impervious surfaces which may reduce infiltration capacity. Furthermore, these land-use practices led to incision of stream channels, increasing the depth of the unsaturated zone and, in turn, increasing groundwater drainage from the riparian water table into the stream (Groffman et al., 2003; Schilling et al., 2004; Hunt et al., 2018) (Figure 2.2).

When restoration of incised channels creates shallower stream channels, the system has the potential to increase storage and to simulate unimpacted conditions through decreasing hydraulic gradients, increasing recharge potential, allowing floodwaters to access the floodplain, and increasing hydrologic retention. This potential increase in storage can take place via infiltration and recharge as the thick unsaturated zone, representative of the impacted system, begins to fill (Figure 2.2A). Moreover, post-restoration streams can dissipate flood volumes by providing access to the surrounding floodplains which is a goal of restoration. Once the floodplain is inundated, floodwaters have longer residence times allowing for increased infiltration potential when compared to incised reference streams (Hammersmark et al., 2008; Healy & Scanlon, 2010; Hunt et al., 2018). In situations such as these, the change in storage would be positive as groundwater storage increases post-restoration much like what was discovered by Hunt et al. (2018) and Hammersmark et al. (2008). Alterations of stream morphology can lead to increases in groundwater levels in the near-stream and upland areas such as was discovered by Moorhead et al. (2008) in their investigation of restoration re-meandering a previously channelized stream. Furthermore, a study by Welsh et al. (2020) found that the addition of stream geomorphological features such as riffle-pool sequences and cross vanes increased groundwater levels in the areas surrounding the channel. Changes in baseflow may be seen after restoration such as a decrease in baseflow caused by increased evapotranspiration (Hammersmark et al., 2008) or an increase in baseflow driven by increased water availability.

Recharge of groundwater is an increasingly important consideration for water managers looking to prepare watersheds for future potential climate shifts (Healy &

Scanlon, 2010). Through restoration, impaired watersheds have the potential to regain their natural forms and functions. Generating changes in impaired watersheds that aid groundwater will help balance nutrient and sediment dynamics, provide habitat for various flora and fauna, and increase groundwater storage and retention potential.

Increased groundwater levels brought by restoration can have important implications for aquatic and terrestrial wildlife species as well as for human uses. For wildlife, increasing groundwater storage with restoration can help maintain sufficient baseflow necessary for local flora and fauna to thrive. Understanding that restoration of degraded streams can increase watershed-scale water retention, including groundwater storage, can provide benefits such as stormwater and flood control, and can enhance exchange of nutrients with the floodplain which is essential for watershed managers hoping to prepare their watershed for future climate change.

2.2 Study Objectives and Research Questions

Few published studies exist that examine the effect of stream restoration on groundwater storage and recharge (Hammersmark et al., 2008; Moorhead et al., 2008; Hunt et al., 2018; Welsh et al., 2020) and none investigate the efficacy of using the whole-watershed stream restoration approach (i.e.: a large spatial scale) to increase groundwater levels throughout restored watersheds.

The sample period for this study was bound from January 2016 through December 2020 as this time period represents the two-year pre-post restoration period. The pre-restoration period includes data from January 2016 – December 2017. The post-restoration period includes data from January 2019 – December 2020 (Table 1.2). The during-restoration period incorporates the period of January 2018 – December 2018 and

is not included in any statistical analyses as this period represents when the watershed was undergoing restoration to its tributaries. Here, the objective of this study was to determine if whole-watershed stream restoration was successful at increasing recharge to the thick unsaturated zone, either by direct recharge or GW-SW interactions, and ultimately raising groundwater levels by creating more shallow streams. Here, I quantified changes in groundwater levels following whole-watershed stream restoration. For this research study I asked four questions:

(RQ2.1) *Does groundwater storage increase between the pre- and post-restoration watershed? If so, to what extent?* I hypothesized that:

Hypothesis	Hypothesized mechanism
Restoration would influence groundwater storage with overall storage in the watershed being higher post-restoration.	This is due to post-restoration changes in the watershed such as the raising of stream channel elevations allowing for the thick unsaturated zone present during the pre-restoration period to become saturated and thus increasing groundwater storage and retention.

(RQ2.2) *How does the seasonal groundwater level fluctuation and hillslope hydraulic gradient differ between the pre- and post-restoration stream?* I hypothesized

Hypothesis	Hypothesized mechanism
Pre-restoration groundwater levels and hydraulic gradients would change seasonally, with levels peaking and gradients being steepest during the dormant season. Levels will be lowest and gradients gentlest during the growing season. Post-restoration, groundwater levels and hydraulic gradients would exhibit the same trends but at a shallower level. Generally, post-restoration groundwater levels would be closer to the land surface. Overall, as groundwater levels rose (as is expected after restoration), gradients from the upland wells to the riparian wells and to the stream would become less steep.	This hypothesized decrease in gradients is due to the assumed increase in groundwater and stream water levels at monitoring transects allowing for a more gentle gradient from the upland well to the stream.

(RQ2.3) *Does baseflow increase between the pre- and post-restoration stream?* I

hypothesized that:

Hypothesis	Hypothesized mechanism
Baseflow at the watershed outlet will increase.	Pre-restoration, baseflow at the watershed outlet would be lower as there will be less groundwater retained in the watershed to support robust flows and more flashy flow (e.g.: stormflow-dominated). Post-restoration, there would be an increase in groundwater storage which will lead to an increase in baseflow amounts and baseflow consistency at the watershed outlet. Here, groundwater should be directly responsible for baseflow in the study watershed.

(RQ2.4) *Does the seasonality of near-stream shallow groundwater change*

between the pre- and post-restoration stream? I hypothesized that:

Hypothesis	Hypothesized mechanism
Seasonal cycling of groundwater levels would not significantly change in amplitude (i.e.: summer low groundwater levels and winter high groundwater levels) but would continue cycling at a level closer to the land-surface (i.e.: the water table would be higher).	This is due to the assumption that seasonal characteristics (i.e.: precipitation amounts and vegetation effects) would remain the same throughout the short study period. Seasonal near-stream groundwater level cycling would continue into the post-restoration period at a higher elevation (i.e.: closer to the land surface).

To test these hypotheses, groundwater levels and stream water levels were monitored continuously using pressure transducers.

2.3 Methods

2.3.1 Field Methods

Onset HOBO (#U20L-01 and #U20-001-01) continuous recording pressure transducers were deployed at five surface water and groundwater transects within the RC watershed encompassing five stream sites and ten monitoring wells for continuous water level measurements. A data logger was deployed at the watershed outlet (site R1) where

it was placed near the USGS monitoring station and recorded barometric pressure. Stream loggers were encased in 1.8-meter PVC pipe housings with slits to ensure adequate stream flow and a removable cap was placed on the top to allow for ease of access when downloading data. The PVC pipes were installed in the stream reach with the attempt to create an approximately perpendicular straight-line transect with the riparian and upland wells. However, not all transects were exactly perpendicular due to limitations in placing the stream logger (see discussion with hydraulic gradients below). Due to lessons learned with regard to scour in the pre-restoration period, post-restoration loggers were installed more effectively. Here, using a hand auger, the pipe tips were inserted into the stream bed deep enough to allow sensors to be covered with water at baseflow. Pipes were held in place with fence posts and were secured with worm clamps. Each data logger was secured to the top cap using eyehooks and a known length of string. In-situ well loggers were encased in metal and PVC pipe casings. Wells average five meters deep (see Table 2.1). Measurements of meters above sea level were taken from the top of the casing (TpOC) at all logger locations. Loggers were set to record temperature and pressure every 10 minutes. Riparian well, upland well, and stream logger data were recorded from 2016 through 2020 (at reaches A1, R2, D1, P1, and C1; Figure 1.5). During restoration, the upland well and the riparian well located at C1 and the riparian well located at A1 were damaged and moved slightly (Table 2.1).

Using HOBOWare software (Onset Computer Corporation, 2022), loggers were downloaded manually approximately every six weeks. Loggers were removed from their casings, stopped, data were downloaded, and loggers were set to relaunch at 1400 hours the same day as downloading took place. Prior to Spring 2019, data were recorded in

local time according to the default time settings of the Windows laptop used for downloading. However, the HOBO loggers do not recognize daylight savings time and therefore, beginning in Spring 2019 Eastern Standard Time was used throughout the remainder of the project. A known limitation of this study is that daylight savings time may or may not have been recognized early in the project, and therefore, times associated with the HOBO loggers may be off by plus or minus one hour.

Table 2.1 Well locations and depths (m). New wells (C1R, C1U, A1R) are wells that had to be replaced after restoration. Stream locations are post-restoration coordinates. Well characteristics were provided by Wildlands Engineering INC.

Site	Longitude (N)	Latitude (W)	Top of casing elevation above sea level (m)	Land surface elevation above sea level (m)	Logger elevation above sea level (m)	Well depth (m below land surface)
D1-RIP	35°15'50.7"	80°42'40.8"	204.5	203.6	199.9	4.2
D1-UP	35°15'50.0"	80°42'40.5"	206.9	206.0	200.3	6.3
D1-STREAM	35°15'49.0"	80°42'41.7"				
P1-RIP	35°15'48.4"	80°42'45.1"	203.9	203.0	200.0	3.9
P1-UP	35°15'49.3"	80°42'44.3"	203.7	202.8	199.7	5.1
P1-STREAM	35°15'47.2"	80°42'44.2"				
C1-RIP	35°15'44.8"	80°42'41.1"	203.4	202.6	199.6	3.6
C1-RIP new	35°15'44.408"	80°42'41.164"	204.0	203.1	200.1	3.6
C1-UP	35°15'45.6"	80°42'42.7"	203.7	202.9	200.0	3.9
C1-UP new	35°15'45.017"	80°42'42.530"	204.6	203.8	200.9	3.9
C1-STREAM	35°15'44.4"	80°42'40.7"				
A1-RIP	35°15'32.2"	80°42'09.7"	198.3	197.5	194.8	4.5
A1-RIP new	35°15'31.862"	80°42'09.066"	198.1	197.2	194.5	3.6
A1-UP	35°15'32.0"	80°42'10.7"	200.9	199.5	195.5	3.6
A1-STREAM	35°15'31.8"	80°42'08.6"				
R2-RIP	35°15'29.6"	80°42'12.5"	198.4	197.7	192.8	5.1
R2-UP	35°15'28.8"	80°42'12.7"	197.9	197.1	196.1	5.1
R2-STREAM	35°15'30.4"	80°42'11.4"				

2.3.2 Data Sources

Several external data sets were utilized for the analyses in this study. First, precipitation data were used from the USGS rain gauge at Reedy Creek Nature Center (US Geological Survey, 2022b). This monitoring station records precipitation every five minutes. Second, stream discharge data were utilized from the USGS Reedy Creek stream stage-discharge gauge (US Geological Survey, 2022a) located at the outlet of the

Reedy Creek watershed (site R1). Third, specific conductance of weekly precipitation data were taken from the National Atmospheric Deposition Program (2022) (station NC34) which was used for the precipitation endmember for the hydrograph separation calculations. Fourth, hourly stream specific conductance data were taken from the Charlotte-Mecklenburg Storm Water Services Continuous Monitoring and Alert Notification Network (CMANN, 2022) monitoring station located at the outlet of the headwaters of the Reedy Creek Watershed.

2.3.3 Data Processing

To analyze data retrieved from the HOBO pressure transducers, HOBOWare, in conjunction with Excel and R were used. In HOBOWare, data were plotted using pressure in kilopascals and temperature in degrees Celsius. Data were downloaded as CSV files and any spurious data points (i.e.: data points in which the logger recorded while it was out of the water, resulting in a low-pressure spike) were removed. These time series datasets were analyzed in chronological order. After removing spurious data points, data were re-imported into HoboWare, further inspected, exported as CSV into Excel for conversion of date formats, then re-exported as CSV and finally imported into R and analyzed for changes in groundwater level. Time series points where no data were recorded were treated as gaps with data points being left blank.

For all calculations, results were reported in meters above sea level. To calculate elevation above sea level for all loggers, Equation 2.1 was used. For this formula, elevation TpOC is the elevation above sea level at the top of the well casings. The string length is the length of the string used to suspend the data logger in the well. These two numbers are subtracted to determine the data logger's elevation above sea level.

$$\text{Elevation TPOC} - \text{String Length}$$

Equation
2.1

$$= \text{Logger Elevation Above Sea Level}$$

To calculate groundwater level from barometric pressure in kilopascals (kPa), Equation 2.2 was used. For this formula, kpa well is the pressure recorded (in kilopascals) by the data logger in the well. The kpa barometric logger is the pressure recorded (in kilopascals) by the data logger at the watershed outlet. These two numbers are subtracted to determine the corrected well pressure (in kilopascals). In other words, this calculation is completed to determine water pressure in the well. In cases of missing barometric data, the average of barometric pressure from the barometric logger was used for barometric compensation.

$$\text{kpa Well} - \text{kpa Barometric Logger}$$

Equation
2.2

$$= \text{Corrected Well Pressure (kpa)}$$

To calculate groundwater levels in meters above sea level, Equation 2.3 was used which incorporates corrected well pressure, TPOC elevation, string length, and the hydrostatic equation. For this calculation, the hydrostatic equation $(0.145038 \times 0.703 \times 3.28)$ was used to convert well pressure from kilopascals (kPa) to feet of pressure.

$$(TpOC - String Length + (Corrected Well Pressure (kpa) * (0.145038 * 0.703 * 3.28)) * 0.3048 = Well Water Level Meters in Above Sea Level \quad \text{Equation 2.3}$$

To calculate hydraulic gradients, I used Equation 2.4. Here, ΔH is the change in water elevation from the stream to the riparian well and, ultimately, to the upland well. L is the distance between the three points. Stream, riparian, and upland water levels were collected via pressure transducers described above and gradient calculations took place on monthly average groundwater levels. Gradient calculations took place using vertical and horizontal distances collected via a total station survey. As transects were not completely perpendicular, reported hydraulic gradients should be considered minimum gradients, as actual gradients could be slightly steeper along a perpendicular transect.

$$\Delta H / \Delta L = \text{hydraulic gradient} \quad \text{Equation 2.4}$$

A standard two-component hydrograph separation (Equation 2.5) was used to calculate the relative contribution of old and new water to the overall hydrograph (Cartwright, 2022; Cartwright & Miller, 2021; Hornberger et al., 1998; Pellerin et al., 2008; W. Yang et al., 2019). Here, Q_o is discharge of old water, Q_t is total stream flow, C_n is tracer concentration of new water, C_o is the tracer concentration of old water, and C_t is the measured tracer value.

$$Q_o = Q_t [(C_n - C_t) / (C_n - C_o)] \quad \text{Equation 2.5}$$

Time series decomposition was used to determine if restoration-induced changes in seasonal groundwater cycling occurred using the forecast R package (Hyndman et al., 2022) (Figure 2.3). Here, time series data were broken into four components (e.g.:

observed, trend, seasonal, and random) and graphed. The observed data are raw data collected from groundwater pressure transducers. Trend data are the data that represent the moving average (i.e.: this package calculates a moving average monthly for each month throughout the year). Seasonal data represent seasonal cycling. Finally, random data are those that are left after the trend and seasonal data are removed. These random data are those that are unable to be explained by either the trend or the seasonal fluctuations.

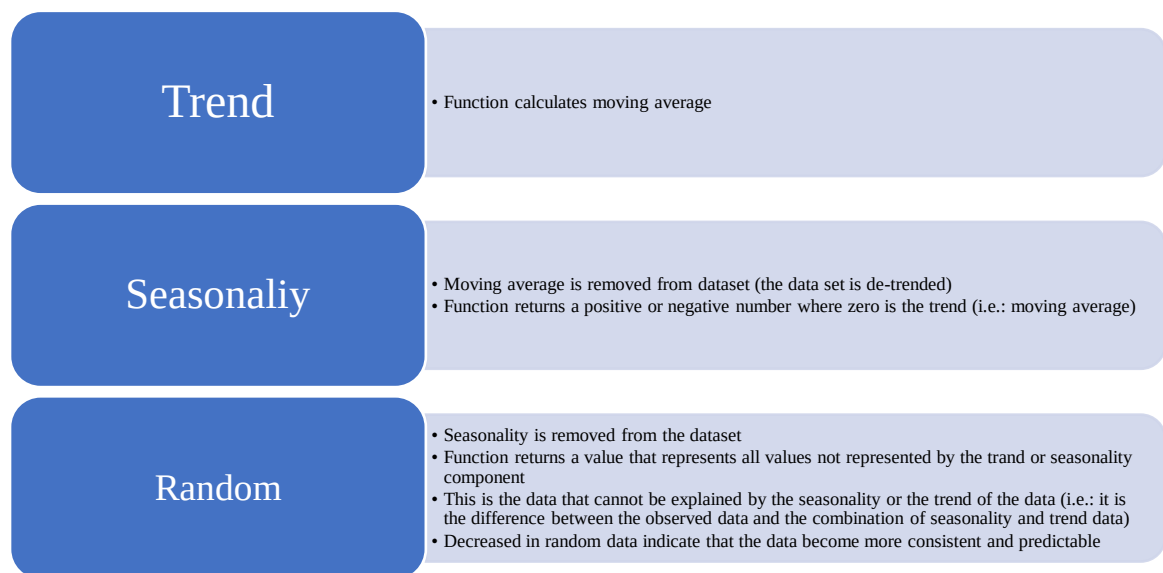


Figure 2.3 Time series decomposition steps

2.3.4 Statistical Analysis

Basic descriptive statistics (e.g.: mean, median, mode, standard error, standard deviation, range, minimum, and maximum) were calculated for pre- and post-restoration WLs. Calculations were completed for each of the five well transects, five stream sites, and precipitation data. All statistical analyses were conducted in R (R Core Team, 2019). R packages used include base R, BINCOR (Polanco-Martínez et al., 2018), ggpubr (Kassambara, 2020) and forecast (Hyndman et al., 2022). BINCOR was used to

determine correlations between irregularly spaced time series data sets (Polanco-Martínez et al., 2018). ggpubr was used for general correlation analyses (Kassambara, 2020).

forecast was used to determine seasonality of groundwater levels (Hyndman et al., 2022).

To answer RQ2.1, *does groundwater storage increase between the pre- and post-restoration watershed*, two levels of data analysis took place. First, time series plots were constructed and visually examined for differences in groundwater levels over time. Second, the time series were separated into pre-and post-restoration periods for comparison. Here, data were separated based on construction start and end dates for each study sub-watershed and all data for pre-restoration were collected before the start of restoration in the headwaters (Table 1.2). Moreover, the during-restoration period encompasses the time any of the streams were undergoing restoration. Finally, post-restoration data were collected after all streams within the watershed were completely restored. Non-parametric Mann-Whitney tests were used to assess differences in WLs before and after treatment (i.e.: restoration; Whitlock & Schluter, 2009) where the W score represents the rank of the sample and the p value indicates the significance of results.

To answer RQ2.2, *how do the seasonal hydraulic gradients differ between the pre- and post-restoration stream*, time series plots were constructed and visually examined for seasonality. The time series was separated into pre-and post-restoration periods for comparison. Non-parametric Mann-Whitney tests were used to assess differences in gradients before and after treatment (i.e.: restoration) for each month (Whitlock & Schluter, 2009).

To answer RQ2.3, *does baseflow increase between the pre- and post-restoration stream*, two-component storm hydrograph separations were conducted following established methods (Cartwright, 2022; Cartwright & Miller, 2021; Hornberger et al., 1998; Pellerin et al., 2008; W. Yang et al., 2019) to determine relative differences in baseflow. Here, the hydrograph was separated into the new water (i.e.: precipitation additions) and old water (i.e.: baseflow) end members using a two-endmember hydrograph separation model. Once separated, baseflow discharge at R1 was divided into pre- and post-restoration periods (excluding the during-construction periods). Non-parametric Mann-Whitney tests were used to assess differences in baseflow before and after treatment (i.e.: restoration; Whitlock & Schluter, 2009). Furthermore, a baseflow index score was calculated from daily average discharge. The baseflow index, which is defined as the total baseflow Q divided by the total Q , was calculated using an automated filtering method (Nathan & McMahon, 1990) on daily average USGS Q data to determine if there was a restoration-induced effect on flashiness at the watershed outlet. The filter was run forward-backward-forward using the recommended parameter $\alpha=0.925$ to ensure consistency of results (Nathan & McMahon, 1990; Schwartz & Smith, 2014).

To answer RQ2.4, *does the seasonality of near-stream shallow groundwater change between the pre- and post-restoration stream*, groundwater data sets were compared across the pre-restoration and post-restoration periods. First, seasonal decompositions (i.e.: separating the seasonal trends from random background noise and general groundwater level trends) were computed using the *forecast* (Hyndman et al., 2022) decomposition function with a frequency of 12 (i.e.: monthly data). Second, decomposed seasonal time series plots were visually examined for differences over time

and were separated into pre-and post-restoration periods for comparison. Third, descriptive statistics were calculated to determine differences in seasonal cycling from decomposed data for the pre- and post-restoration periods. Fourth, Mann-Whitney tests were used to assess differences in WLs before and after treatment (i.e.: restoration; Whitlock & Schluter, 2009).

2.4 Results

2.4.1 (RQ2.1) Does groundwater storage increase between the pre- and post-restoration watershed? If so, to what extent?

2.4.1.1 D1 – Developed sub-watershed

Water levels from D1U (RCMW5) and D1R (RCMW5A) are reported in Figure 2.4. Annual statistics are presented in Table 2.2 and Table 2.3. The riparian well D1R was adjacent to the deeply incised, straight, pre-restoration channel and is located along the newly meandering post-restoration channel. The upland well D1U is a hillslope well, located significantly uphill from D1R and away from the channel. This hillslope setting is reflected in the deeper water tables in D1U (commonly more than five meters deep pre-restoration; Figure 2.4). Along with A1U, D1U exhibits a true hillslope setting rather than being on a flat or floodplain-like surface as was seen with the riparian wells, C1U, P1U, and R2U.

Water levels between the upland and riparian well differed seasonally in both the pre- and post-restoration periods (Figure 2.4). During winter, the upland well exhibited higher water levels than the riparian well. During summer, the upland well and riparian well had similar groundwater levels. This suggests that in winter, groundwater is added to storage. Stormflow events are evident in the groundwater level data, especially in pre-

restoration D1U data where pulses appear to represent recharge events – alternatively, these pulses may represent a transitory pressure effect driven by storms. The steep tall spikes of higher groundwater levels (including sudden rises more than 1.5 meters above the seasonal water level) may indicate that subsurface stormflow reached the well. Post-restoration, the steep tall spikes were replaced by rises and falls more typical of groundwater recharge at only 0.3-0.6 meters above the seasonal water table level. These findings suggest that areas of high hydraulic conductivity became part of the saturated zone rather than pathways for stormflows in the shallow unsaturated zone as groundwater levels increased.

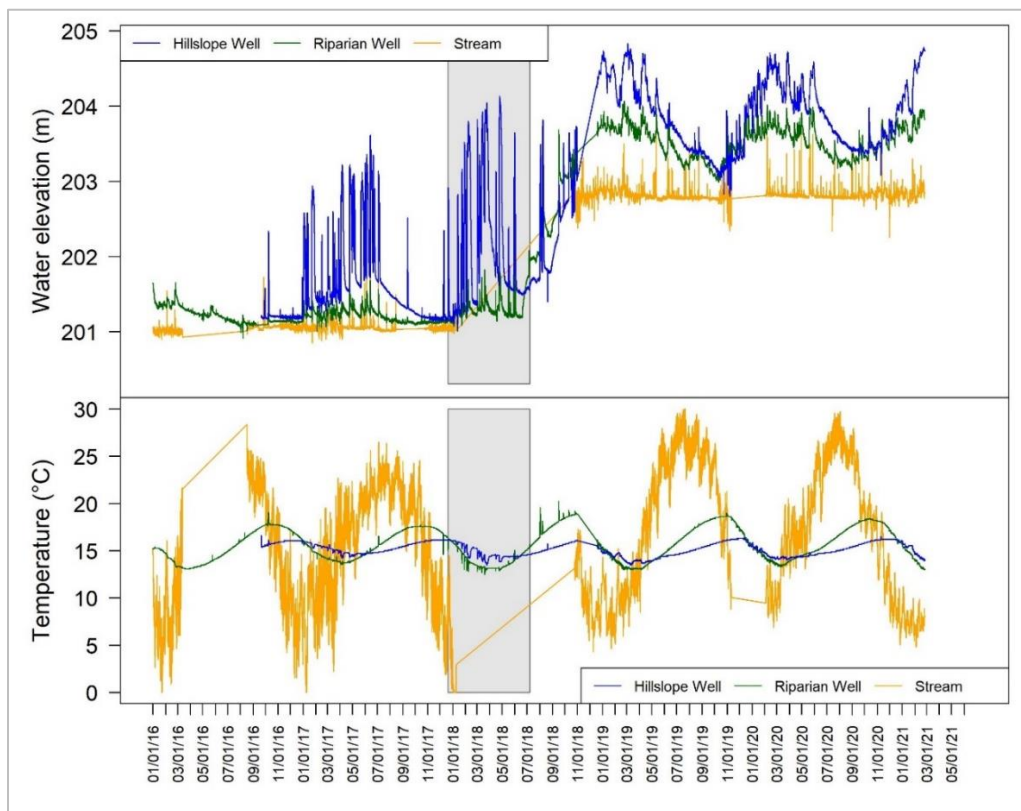


Figure 2.4 Water elevation (m above sea level) and temperature (°C) for the D1 well transect site. The blue line is the hillslope well, green line is the riparian well, and orange line is the stream. Data represent entire sampling period from January 2016 – March 2021. Data are in ten-minute intervals. Grey bar represents the during-restoration period.

D1R and D1U were monitored continuously during the construction period. Both wells exhibited sharp increases in water levels between summer 2018 and winter 2019 (i.e.: during the construction period), in which D1R (Figure 2.5) and D1U (Figure 2.6) water levels rose 1.8-2.4 meters. D1R water levels shifted from 2.7-3.0 meters below the land surface pre-restoration to 0.6-1 meters below the land surface post-restoration (Figure 2.5). D1U water levels shifted from 3-5.5.0 meters below the land surface pre-restoration to 1.8-3.6 meters below the land surface post-restoration (Figure 2.6).

The water level difference between the riparian well and the stream water level increased post-restoration. Pre-restoration, D1R's water level was 0.0-0.6 meters higher than the stream water level in winter/spring and not higher than the stream water level in summer/fall, while post-restoration, D1R's water level was at least two meters higher than the stream water level in winter/spring and at least 1.5 meters higher than the stream water level in summer/fall.

Results of a Mann-Whitney test indicate that there was a statistically significant difference in stream water levels between the pre-restoration and post-restoration periods ($W = 73006534$, $p < 2.2 \times 10^{-16}$). Results of a Mann-Whitney test suggest that there was a statistically significant difference in upland well water levels between the pre-restoration and post-restoration periods ($W = 7183855$, $p < 2.2 \times 10^{-16}$). Furthermore, findings from a Mann-Whitney test indicate that there was a statistically significant difference in riparian well water levels between the pre-restoration and post-restoration periods ($W = 0$, $p < 2.2 \times 10^{-16}$).

2.4.1.1.1 Condition at end of study period

By the end of the study period, both the riparian and upland wells had reached a new stable state, rising and falling seasonally. The upland well exhibited uniformly higher water levels than the riparian well implying a potential for robust baseflow year-round, with the largest relative difference being seen in summer, protecting the stream from drying. These trends are complicated by the fact that the stream channel was moved laterally away from the D1R and D1U wells during restoration, which is consistent with groundwater flow from the hillslope to the stream.

2.4.1.1.2 Inferred restoration effect

It is noteworthy that restoration apparently caused D1U, distant and uphill from D1R, to have similar increase in water levels as a result of restoration as was seen in the streamside well D1R. This demonstrates how stream restoration can affect groundwater levels some distance from the channel. This situation is typical of topographic-driven groundwater flow. Due to the presence of hilly topography near the stream, there is a source of groundwater having higher head than the riparian groundwater. The rise of water tables could affect not only riparian vegetation but also the wooded hillslope beyond the area cleared for the restoration project. Overall, the relationship between upland and riparian water levels suggests seasonally variable baseflow occurred before and occurs after restoration.

While baseflow is evident both pre- and post-restoration, it is intriguing that stormflow signals became muted post-restoration in D1U (note the post-restoration disappearance of tall sharp spikes in Figure 2.6). These signals may be attributed to pressure effects driven by changes in pressure during the storm. Subsurface stormflow is

associated with runoff generation after storm events. If this effect occurred across the watershed more broadly, that would imply an increase in hydrologic retention. Overall, the stream appears to be less likely to dry due to the increased baseflow through the year.

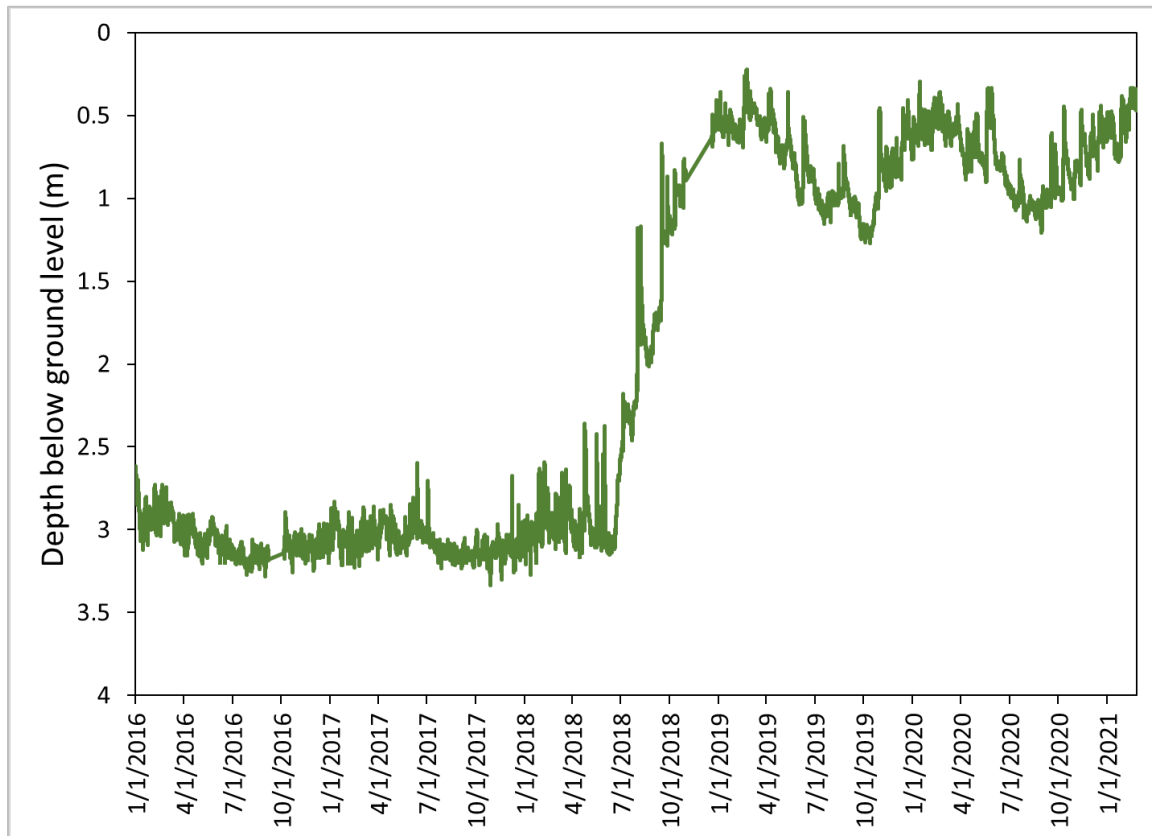


Figure 2.5 Depth to water table below land surface (m) in the riparian well D1R. Data represent the interval from January 2016 – March 2021.

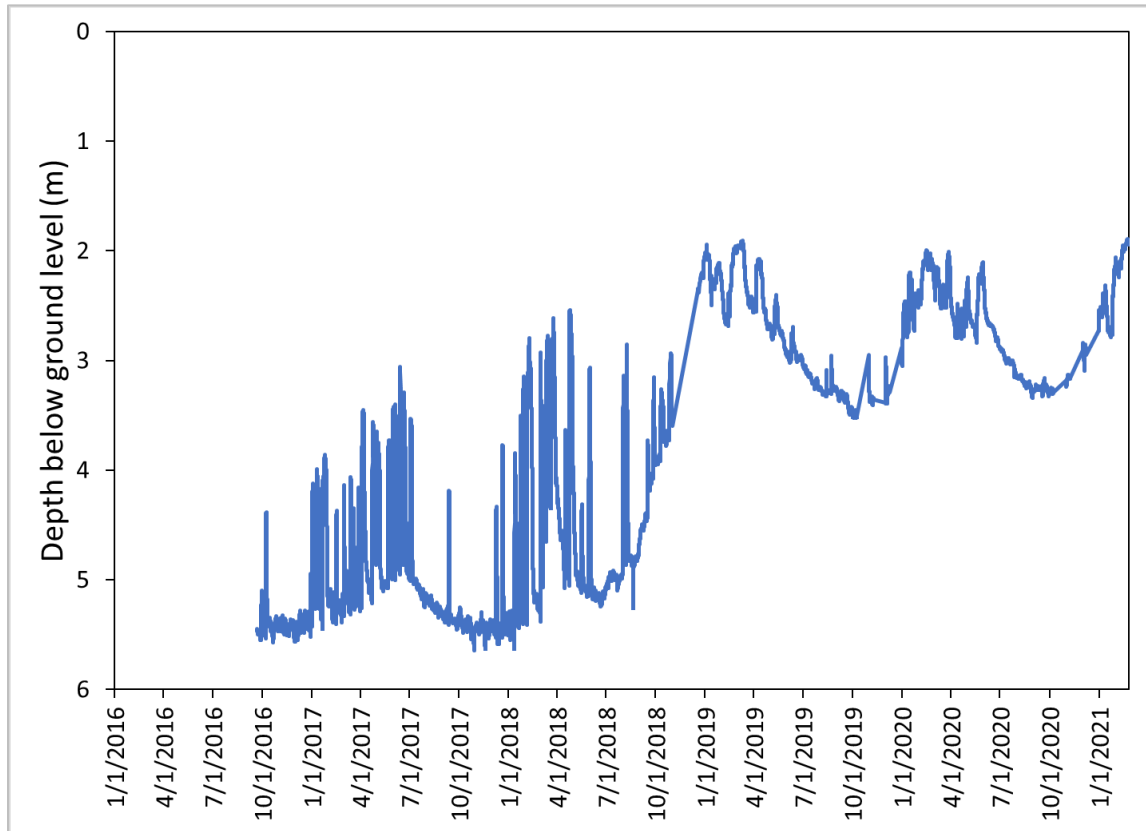


Figure 2.6 Depth to water table below land surface (m) in the upland well D1U. Data represent the interval from January 2016 – March 2021.

Table 2.2 Water level analysis of D1R riparian groundwater. Asterisk *events represent episodic recharge/pressure spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	200.99	1/1/2016	200.32	9/1/2016	0.66	200.56
2017	201.01	6/13/2017*	200.27	10/29/2017	0.74	200.54
2018	203.20	12/28/2018	200.33	1/12/2018	2.86	201.25
2019	203.38	2/23/2019	202.33	10/12/2019	1.05	202.82
2020	203.31	1/14/2020	202.40	8/30/2020	0.91	202.85

*Table 2.3 Water level analysis of D1U upland groundwater. Asterisk * events represent episodic recharge/pressure spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	201.60	10/9/2016	200.42	10/9/2016	1.19	200.59
2017	202.93	6/14/2017*	200.34	6/14/2017	2.59	200.99
2018	204.00	12/31/2018	200.37	12/31/2018	3.63	201.64
2019	204.08	3/11/2019	202.36	3/11/2019	1.72	203.13
2020	203.99	2/15/2020	202.40	2/15/2020	1.59	203.20

2.4.1.2 P1 - Pond controlled / park development sub-watershed

P1R is a riparian well along the Upper Reedy Creek tributary downstream from a park dam. P1U is distant from the channel but it is located on the flat, floodplain-like surface rather than being a true “upland” well. Water levels from P1U (RCMW4) and P1R (RCMW4A) are reported in Figure 2.7. Annual statistics are presented Table 2.4 and Table 2.5. Overall, P1R and P1U exhibit a similar amount of seasonal variability. Pre-restoration, the seasonal variability was 0.6-1.0 meters (Table 2.4 and Table 2.5), but it is difficult to quantify this variability due to the prevalence of event spikes that interrupt the water table records (Figure 2.7). Figure 2.7 exhibits periods during which the stream water elevation appears higher than the well water elevation implying that the stream water elevation is slightly higher than groundwater levels during the summer dry period and suggest that the stream is perched above the height of the adjacent riparian groundwater during summer. However, the apparent difference could be explained by study design factors. First, the well-to stream transects are not exactly perpendicular to the stream, which could cause stream water to appear higher than adjacent groundwater if the stream logger were slightly upstream from the well. Second, the observed differences

within a few inches are within the likely real-world degree of accuracy for height measurements in this study, including vertical survey errors, inconsistent placement of the well cap to which the logger string was attached, and possible ambiguity in other measured heights such as string lengths. In addition, the P1 stream logger was hit by a small tree during the pre-restoration period and had to be moved slightly and was not re-surveyed. Overall, we can confidently state that, year-round groundwater levels were higher than stream water levels post-restoration and that pre-restoration winter groundwater levels were also higher than stream water levels.

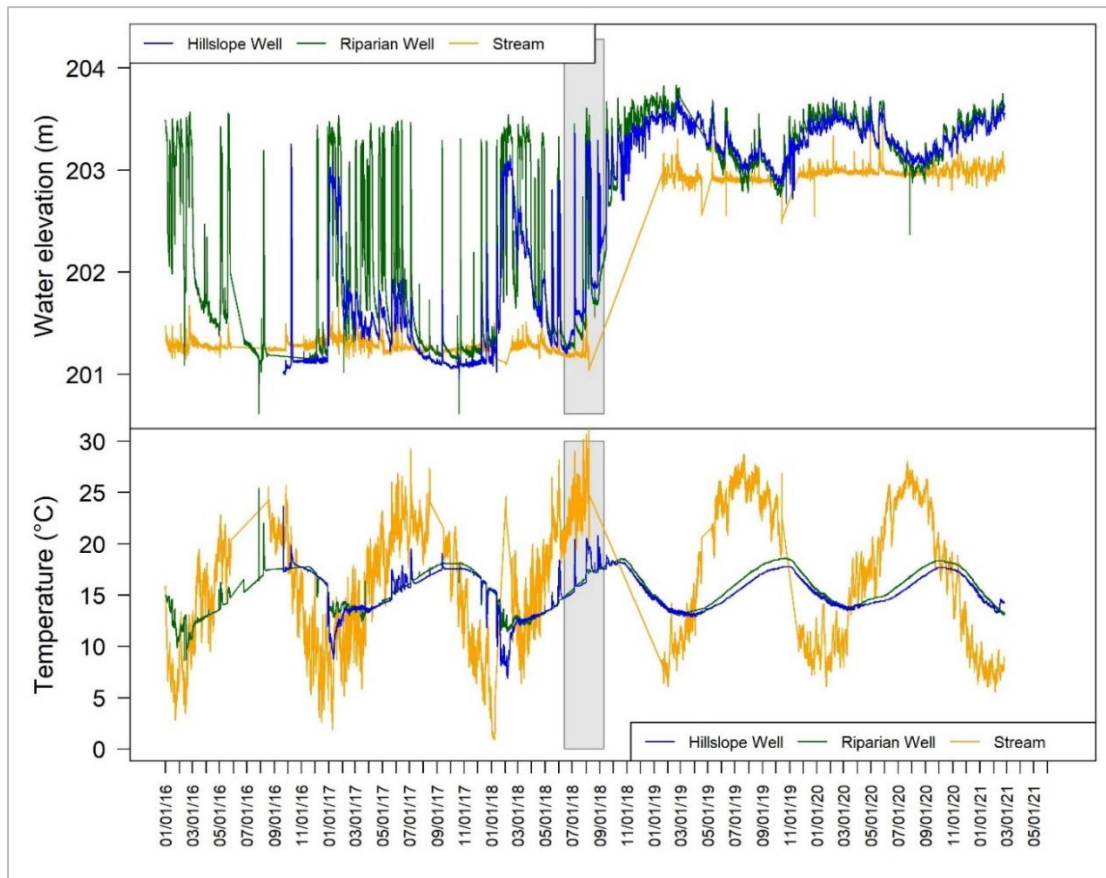


Figure 2.7. Water elevation (m) and temperature (°C) for the P1 well transect site. The blue line is the hillslope well, green line is the riparian well, and orange line is the stream. Data represent entire sampling period from January 2016 – March 2021. Data are in ten-minute intervals. Grey bar represents the during-restoration period.

P1R and P1U were monitored continuously throughout the construction period. Both wells exhibited a sharp increase in water levels during the construction period between summer 2018 and winter 2018, in which P1R (Figure 2.8) and P1U (Figure 2.9) water levels rose by at least 1.8 meters. Water levels in P1R rose from 1.8-2.5 meters below land surface pre-restoration to being within one meter of the land surface post-restoration (Figure 2.8). Water levels in P1U rose from 0.6-2.5 meters below the land surface to being within 0.6 meters of the land surface post-restoration (Figure 2.9). These water levels were essentially within 0.3 meters of the land surface during the first two winters post-restoration. At P1, the water level in the riparian well rose much more than the stream water level due to restoration. P1R's pre-restoration water level, while irregular, was at least 0.3 meters higher than the stream in winter/spring, and not higher than the stream in summer/fall. Post-restoration, the P1R water level was 1.8-2.0 meters higher than the stream level in winter/spring and 1.2-1.5 meters higher than the stream level in summer/fall, implying potential for robust baseflow. The largest difference may be in summer, protecting the stream from drying post-restoration.

Results of a Mann-Whitney test indicate that there was a statistically significant difference in stream water levels between the pre-restoration and post-restoration periods ($W = 525291746, p < 2.2 \times 10^{-16}$). Results of a Mann-Whitney test suggest that there was a statistically significant difference in upland well water levels between the pre-restoration and post-restoration periods ($W = 17009711, p < 2.2 \times 10^{-16}$). Furthermore, findings from a Mann-Whitney test indicate that there was a statistically significant difference in riparian well water levels between the pre-restoration and post-restoration periods ($W = 2146540472, p < 2.2 \times 10^{-16}$).

2.4.1.2.1 Condition at end of study period

At the end of the study period, both the riparian and upland wells were rising and falling seasonally, indicating a new stable state had been achieved. Water levels in the riparian and upland wells, in addition to being much closer to the land surface, are very close to each other during the post-restoration period, suggesting similar controls on water levels including groundwater-surface water interactions and growing season effects such as consumption by trees. It should also be noted that P1U is now closer to the post-restoration D1 stream and P1R is farther from the post-restoration P1 stream. Therefore, P1U was not in an upland-like setting either before or after restoration, and both wells have experienced changes in their relationship to the stream channel.

2.4.1.2.2 Inferred restoration effect

It is important to note that even with the relocation of the streams at this confluence, both P1R and P1U exhibited similar increases in water levels as a result of restoration and exhibit very similar water levels post-restoration suggesting similar controls on both wells. It is apparent that restoration can affect groundwater levels at locations distant from the stream channels. Both P1R and P1U occupy a floodplain-like area that is capable of seasonal soil saturation and potentially capable of being inundated at high flows, whereas before restoration P1R and P1U were capable of periodic soil saturation after rain events and not capable of being inundated.

Much like at the D1 reach, stormflow signals that were present at P1U and P1R pre-restoration became muted in the post-restoration wells. These subsurface stormflows may be associated with post-storm runoff generation or may be attributed to pressure effects driven by changes in pressure during the storm. Post-restoration, the highly

conductive pathways to wells P1U and P1R may have been within the saturated zone due to the rise of water tables, causing the highly conductive paths to no longer be conduits of rapid to the well screens. This effect may also imply an of increase hydrologic retention or less “flashy” hydrology in the system. Overall, due to the increased baseflow throughout the year, the stream appears to be less likely to dry.

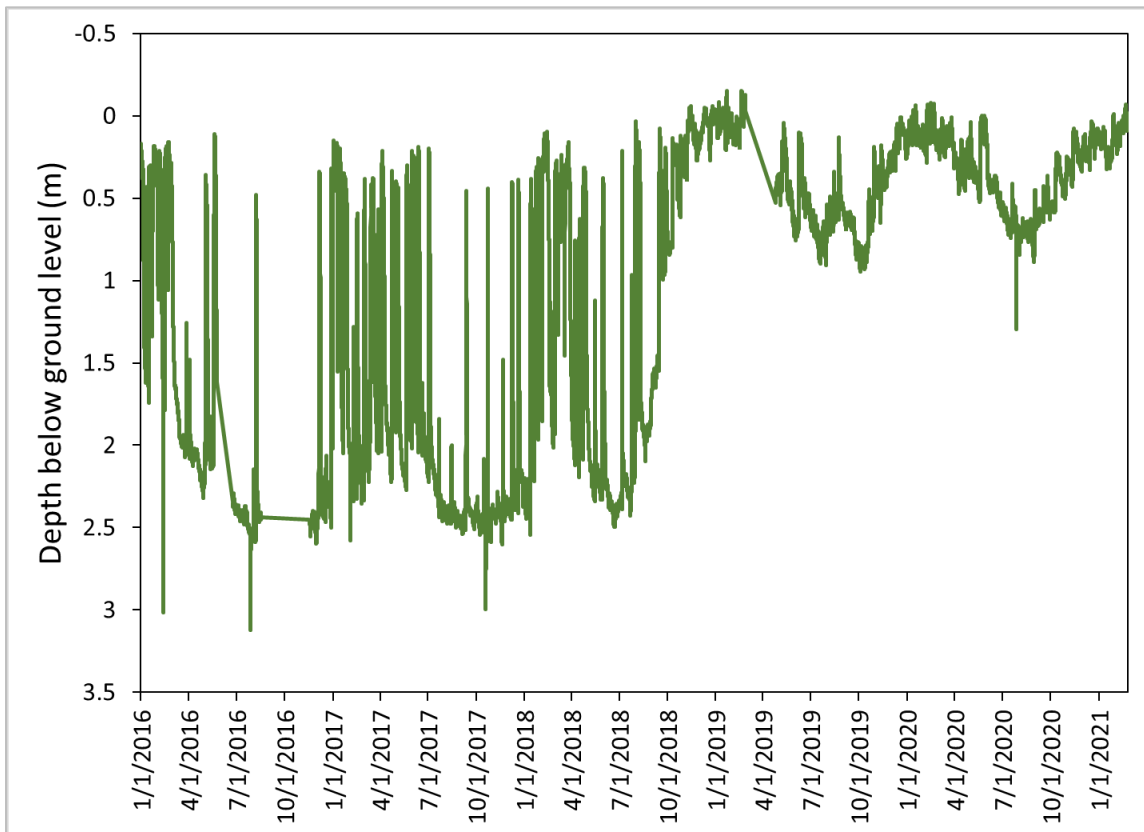


Figure 2.8 Depth to water table below land surface (m) in the riparian well P1R. Data represent the interval from January 2016 – March 2021.

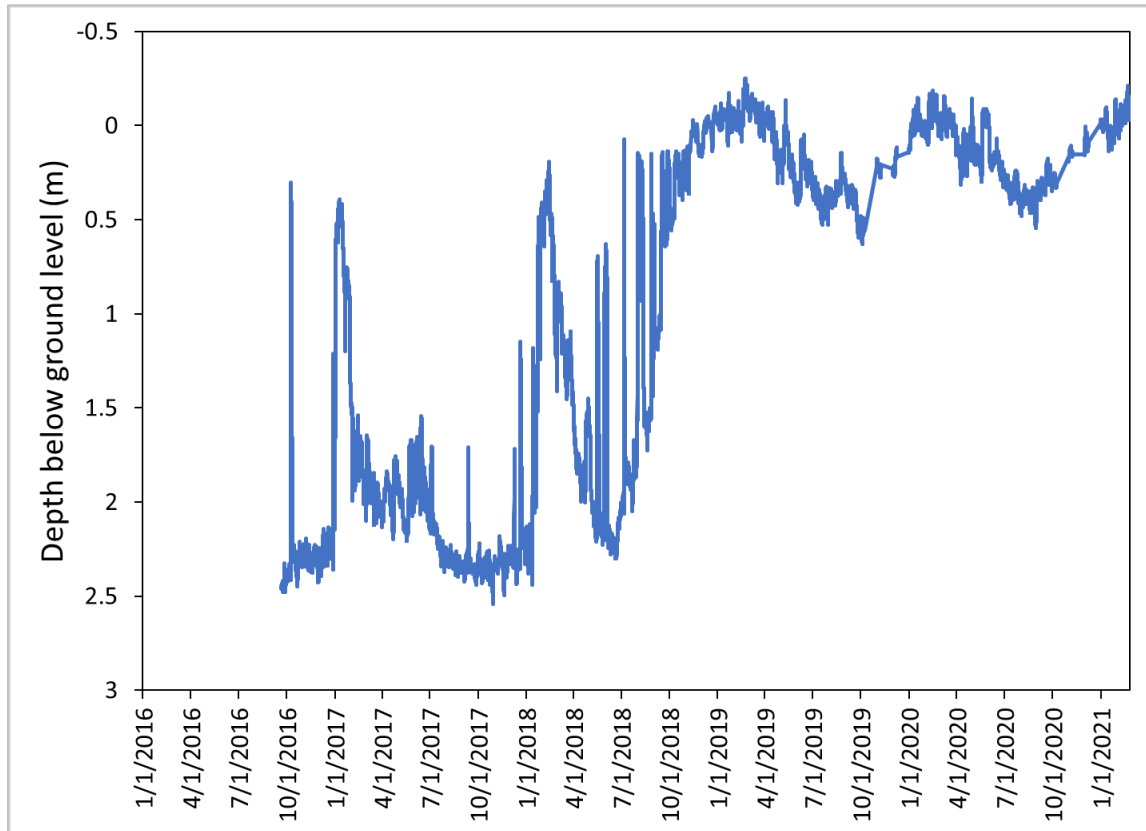


Figure 2.9 Depth to water table below land surface (m) in the upland well P1U. Data represent the interval from January 2016 – March 2021.

Table 2.4 Water level analysis of P1R riparian groundwater. Asterisk *events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	202.88	5/20/2016*	199.87	7/28/2016	3.01	201.31
2017	202.84	1/2/2017	200.00	10/19/2017	2.85	201.12
2018	203.05	11/14/2018	200.45	1/12/2018	2.61	201.80
2019	203.15	2/19/2019	202.05	10/4/2019	1.10	202.57
2020	203.07	2/15/2020	201.70	7/27/2020	1.37	202.66

*Table 2.5 Water level analysis of P1U upland groundwater. Asterisk *events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	202.48	10/8/2016	200.31	9/24/2016	2.18	200.55
2017	202.40	1/9/2017	200.25	10/29/2017	2.15	200.79
2018	202.89	12/26/2018	200.34	1/12/2018	2.54	201.70
2019	203.04	2/22/2019	202.15	10/4/2019	0.88	202.60
2020	202.97	2/15/2020	202.24	8/30/2020	0.73	202.64

2.4.1.3 C1 - Control sub-watershed

C1R and C1U are both located on the flat, floodplain-like surface adjacent to the stream, similar to the setting of the P1R-P1U well pair. Water levels from C1U (RCMW3) and C1R (RCMW3A) are reported in Figure 2.10. Annual statistics are presented Table 2.6 and Table 2.7. Overall, C1U exhibited slightly more seasonal variation in groundwater levels than did C1R during the pre-restoration period. C1R exhibited a 1.8 meter seasonal variation (Table 2.6) while C1U exhibited one meter of variation seasonally (Table 2.7). During the post-restoration period, both C1R and C1U showed about 0.9 meters of groundwater level variation seasonally. In general, seasonal variation decreased in both the riparian and upland wells.

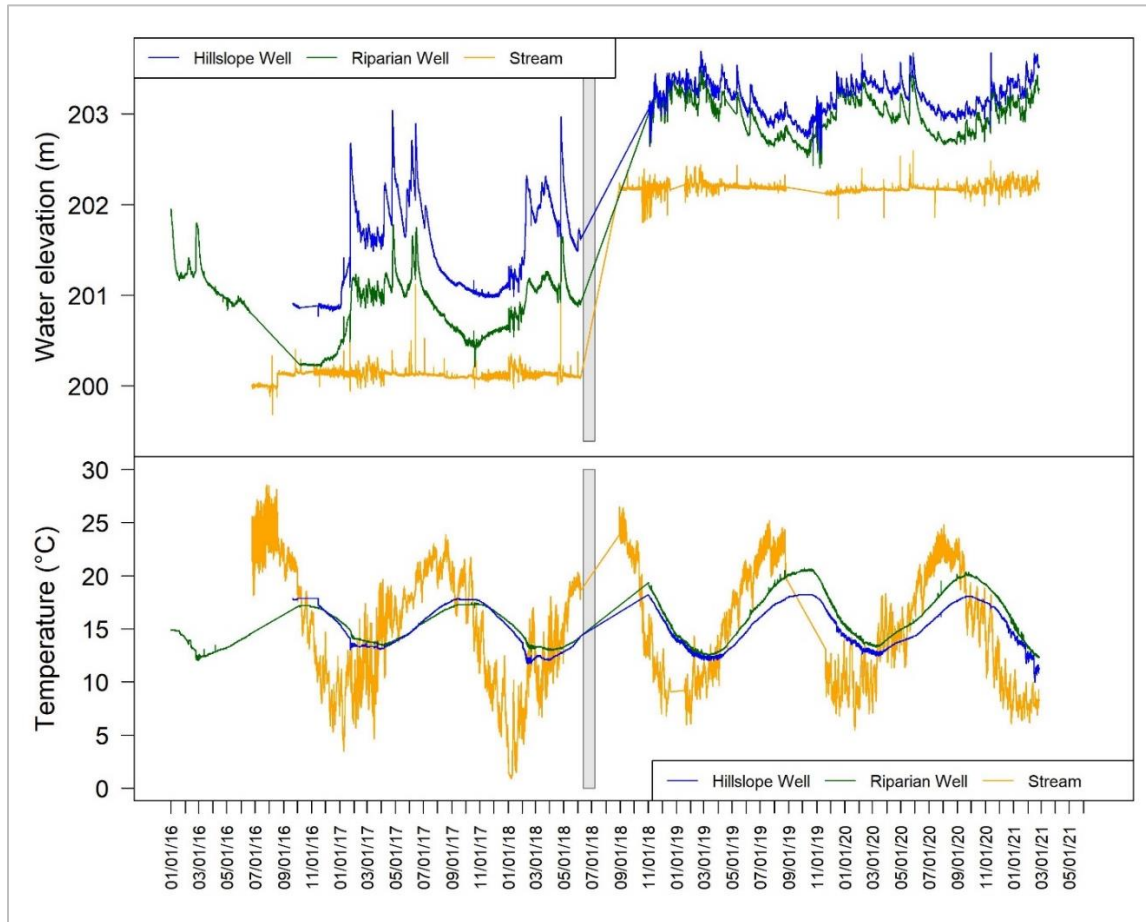


Figure 2.10 Water elevation (m) and temperature (°C) for the P1 well transect site. The blue line is the hillslope well, green line is the riparian well, and orange line is the stream. Data represent entire sampling period from January 2016 – March 2021. Data are in ten-minute intervals. Grey bar represents the during-restoration period.

C1R and C1U were unmonitored during the construction period because both wells were offline and required replacement in approximately the same locations as the pre-restoration wells. Both wells exhibited sharp increases in water levels between summer 2018 and winter 2019, in which C1R water levels rose about 1.5 meters (Figure 2.11) and C1U water levels rose 0.6 meters (Figure 2.12). In the post-restoration period, water levels in C1R were 0.6-1.0 meters below the land surface, compared to 1.8-3.0 meters below the land surface in the pre-restoration period (Figure 2.11). In C1U, water levels were 0.9-1.5 meters below the land surface post-restoration, compared to 0.9-2.7

meters below the land surface pre-restoration (Figure 2.12). It must be emphasized that C1R and C1U were re-drilled in approximately the same location during the restoration construction period but may not represent the same environment before and after restoration. Finally, the water level difference between the riparian well and the stream increased as a result of restoration. Pre-restoration, the riparian well level was 0.6-0.9 meters higher than the stream in winter/spring and 0.0-0.3 meters higher than the stream in summer/fall. Post-restoration, the riparian well level was 1.8-2.0 meters higher than the stream in winter/spring and 1.5 meters higher than the stream in summer/fall. This suggests a stronger potential for year-round baseflow with the largest change potentially occurring in summer/fall, protecting the stream from drying.

Results of a Mann-Whitney test indicate that there was a statistically significant difference in stream water levels between the pre-restoration and post-restoration periods ($W = 250818410, p < 2.2 \times 10^{-16}$). Results of a Mann-Whitney test suggest that there was a statistically significant difference in upland well water levels between the pre-restoration and post-restoration periods ($W = 507497118, p < 2.2 \times 10^{-16}$). Furthermore, findings from a Mann-Whitney test indicate that there was a statistically significant difference in riparian well water levels between the pre-restoration and post-restoration periods ($W = 693818, p < 2.2 \times 10^{-16}$).

2.4.1.3.1 Condition at end of study period

A new stable state had been achieved at the end of the study period which is evident by both the riparian and upland wells rising and falling seasonally in a similar pattern as one another. The riparian well exhibited slightly higher water levels than the upland well, which is not what would be expected in hillslope-to-stream groundwater

flow. However, well C1U is close to the newly restored Upper Reedy Creek channel and may be behaving more like a riparian well post-restoration. Indeed, C1R and C1U exhibit parallel post-restoration groundwater levels, suggesting similar controls on groundwater levels.

2.4.1.3.2 Inferred restoration effect

It is important to note that groundwater levels changed in both the riparian and upland wells indicating that restoration is able to impact groundwater levels some distance from the stream. Much like the P1 study reach, both the riparian and upland wells are located on a floodplain-like surface with no true “upland” area. However, the C1 study reach is not quite as saturated as the P1 reach which may be due to elevation differences.

Groundwater levels in both the riparian and upland wells became more variable post-restoration which implies that restoration has led to a more dynamic system with increased groundwater-surface water connections. Restoration has led to increased water table heights throughout the study area. Overall, in the post-restoration period, increased baseflow throughout the year appears to create the conditions needed to make the stream less likely to dry.

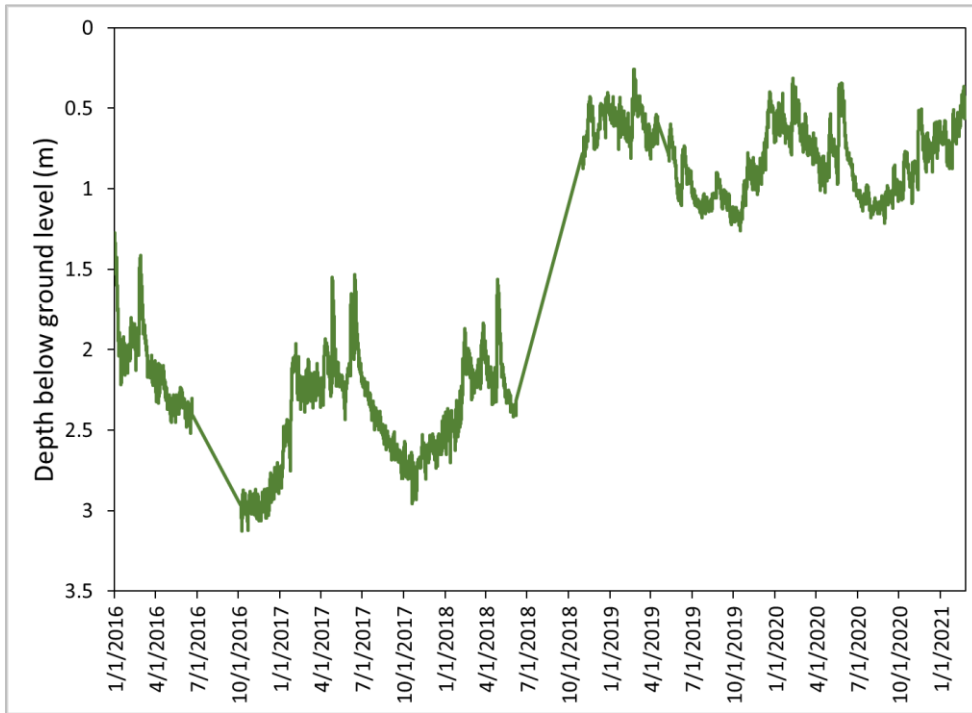


Figure 2.11 Depth to water table below land surface (m) in the riparian well C1R. Data represent the interval from January 2016 – March 2021.

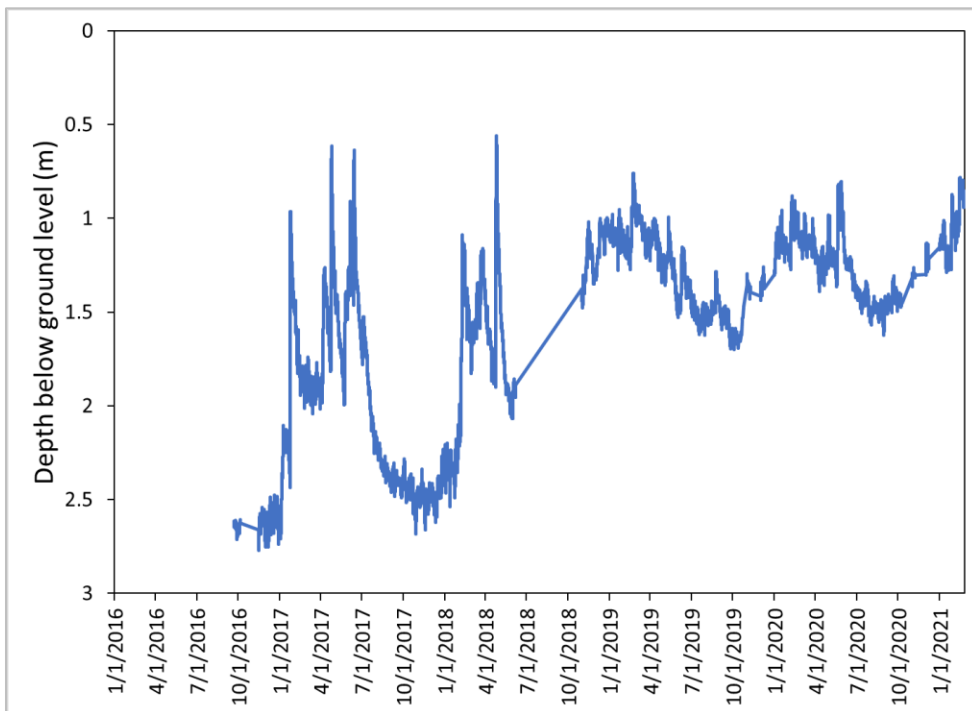


Figure 2.12 Depth to water table below land surface (m) in the upland well C1U. Data represent the interval from January 2016 – March 2021.

*Table 2.6 Water level analysis of C1R riparian groundwater. Asterisk * events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	201.30	1/1/2016	199.44	10/8/2016	1.85	200.23
2017	201.03	6/15/2017*	199.61	10/20/2017	1.42	200.19
2018	202.17	12/26/2018	199.87	1/12/2018	2.30	200.80
2019	202.86	02/22/2019	201.73	11/07/2019	1.13	202.28
2020	202.82	05/27/2020	201.99	08/03/2020	0.83	202.31

*Table 2.7 Water level analysis of C1U upland groundwater. Asterisk * events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	200.37	12/19/2016	200.08	11/15/2016	0.30	200.22
2017	202.23	4/25/2017	200.14	1/3/2017	2.10	200.85
2018	202.29	4/25/2018	200.31	1/12/2018	1.98	201.27
2019	203.02	02/22/2019	201.89	11/07/2019	1.13	202.47
2020	203.00	11/12/2020	202.18	11/10/2020	0.82	202.53

2.4.1.4 A1 - Agricultural sub-watershed

Pre-restoration, A1R was located at the edge of the hillslope and along the bank of the deeply incised stream channel as it approached its confluence with Reedy Creek. The A1R well was replaced during the construction period because the well location conflicted with the eventual stream location. Post-restoration, A1R is located on the flat, floodplain-like post-restoration surface adjacent to the newly reconstructed stream. A1U is a true upland well that is located on a hillslope adjacent to the stream. Water levels from A1U (RCMW1) and A1R (RCMW1A) are reported in Figure 2.13. Annual statistics

are presented Table 2.8 and Table 2.9. Overall, A1R exhibited more seasonality in water table levels than D1R during the pre-restoration period. In A1R, the seasonal variation was approximately 1.5 meters pre-restoration and approximately 0.9-1.0 meters post-restoration (Table 2.8). In A1U, seasonal variation was approximately 1.0-1.5 meters pre-restoration and approximately 1.5 meters post-restoration (Table 2.9). Throughout the year, the groundwater level in the upland well was up to 0.3 meters higher than the riparian well, consistent with down-gradient groundwater flow from the hillslope to the stream. Pre-restoration, this upland to riparian gradient was fairly constant during the year. Post-restoration, the upland well exhibits a larger head difference with the riparian well during the winter/spring wet period, and much less of water level difference in the summer 2020 dry period, suggesting a stronger gradient of groundwater flow from hillslope to stream in winter than in summer. Likewise, the water level difference between the riparian well and the stream is greater in the winter/spring wet period than the summer/fall dry period (Figure 2.13). This pattern is consistent with baseflow being greater in winter than summer.

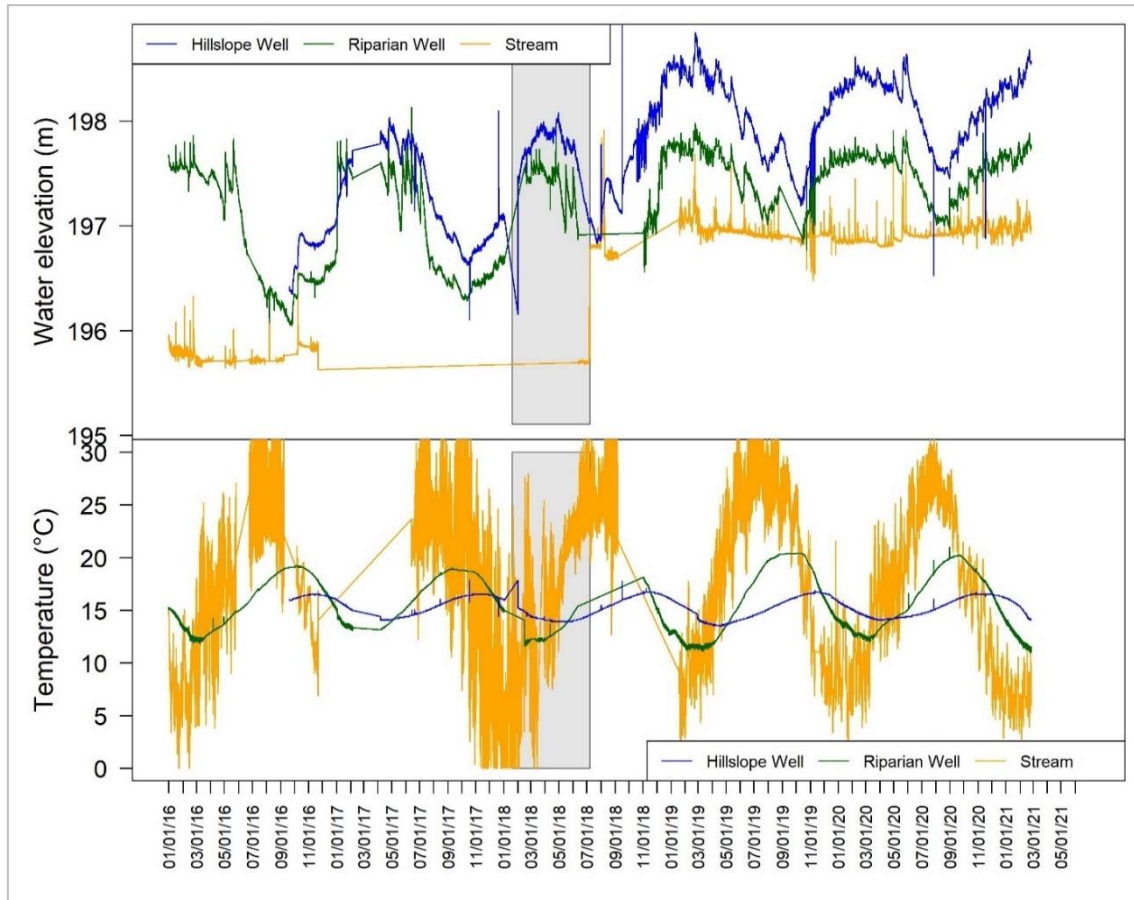


Figure 2.13 Water elevation (m) and temperature (°C) for the A1 well transect site. The blue line is the hillslope well, green line is the riparian well, and orange line is the stream. Data represent entire sampling period from January 2016 – March 2021. Data are in ten-minute intervals. Grey bar represents the during-restoration period.

A1U was monitored continuously during the construction period. A1R was decommissioned during the construction process but was re-drilled in approximately the same location or slightly upstream. Both A1U and A1R exhibited systematic increases in water levels between summer 2018 and winter 2019, in which A1R water levels rose approximately 0.6 meters (Figure 2.14) and A1U water levels also rose approximately 0.6 meters (Figure 2.15). This is not as large an increase as observed at the sites further upstream in the Reedy Creek watershed, D1, P1, and C1. It is possible that larger water

level increases occurred at the three upstream sites due to incision being deeper and more focused in the riparian corridor, whereas the vicinity of A1 features a broader riparian corridor and steeper slopes. Pre-restoration, water levels were 0.6-1.8 meters below the land surface in A1R and 2.0-3.3 meters below the land surface in A1U. Post-restoration, water levels were 0.0-0.9 meters ft below the land surface in A1R and 1.8-3.0 meters below the land surface in A1U (Figure 2.14-Figure 2.15), suggesting that restoration added the potential for seasonally saturated conditions in the riparian corridor near A1R. As with the other sites, the water level difference between the riparian well and the stream water elevation increased post-restoration. Pre-restoration, the A1R water level was up to 1.5 meters higher than the stream water level in winter/spring, and 0.0-0.3 meters above the stream water level in summer/fall, suggesting stronger baseflow in winter/spring. Post-restoration, the A1R water level was up to 1.8 meters higher than the stream in winter/spring and 0.6-0.9 meters higher than the stream in summer/fall, suggesting that the potential for robust baseflow actually increased as a result of restoration, especially in summer. If this effect occurred site-wide, the stronger summer gradient to the stream would protect the stream from drying with beneficial effects on aquatic life.

Results of a Mann-Whitney test indicate that there was a statistically significant difference in stream water levels between the pre-restoration and post-restoration periods ($W = 3459666143$, $p < 2.2 \times 10^{-16}$). Results of a Mann-Whitney test suggest that there was a statistically significant difference in upland well water levels between the pre-restoration and post-restoration periods ($W = 751445522$, $p < 2.2 \times 10^{-16}$). Furthermore, findings from a Mann-Whitney test indicate that there was a statistically significant

difference in riparian well water levels between the pre-restoration and post-restoration periods ($W = 3472343078$, $p < 2.2 \times 10^{-16}$).

2.4.1.4.1 Condition at end of study period

By the end of the study period, a new stable state had been achieved which can be seen by the riparian and upland wells rising and falling seasonally and synchronously. The upland well exhibited slightly higher water levels than the adjacent riparian well which is expected in hillslope-to-stream driven groundwater flow that supports stream baseflow, especially during the winter/spring wet period (Figure 2.13).

2.4.1.4.2 Inferred restoration effect

An overall effect of restoration at the A1 reach is that there was an increase in water levels in both the riparian and upland wells. This provides evidence that stream restoration can have an impact on groundwater levels, not only in the near-stream zone, but also at some distance from the restored channel and up-gradient from the riparian corridor. This situation is common in topographic-driven groundwater flow. Effects from the newly increased water table could be seen in the newly vegetated riparian corridor (A1R) as well as in the upland, hillslope wooded area (A1U) which lie beyond the region cleared for the restoration project. This suggests that vegetation clearing alone is not responsible for the rise in water tables but fundamentally is controlled by the elevation of the stream channel. The relationship between the riparian and upland wells suggests the potential for seasonally variable baseflow after restoration. Overall, the stream is less likely to dry in the post-restoration period due to increases in baseflow.

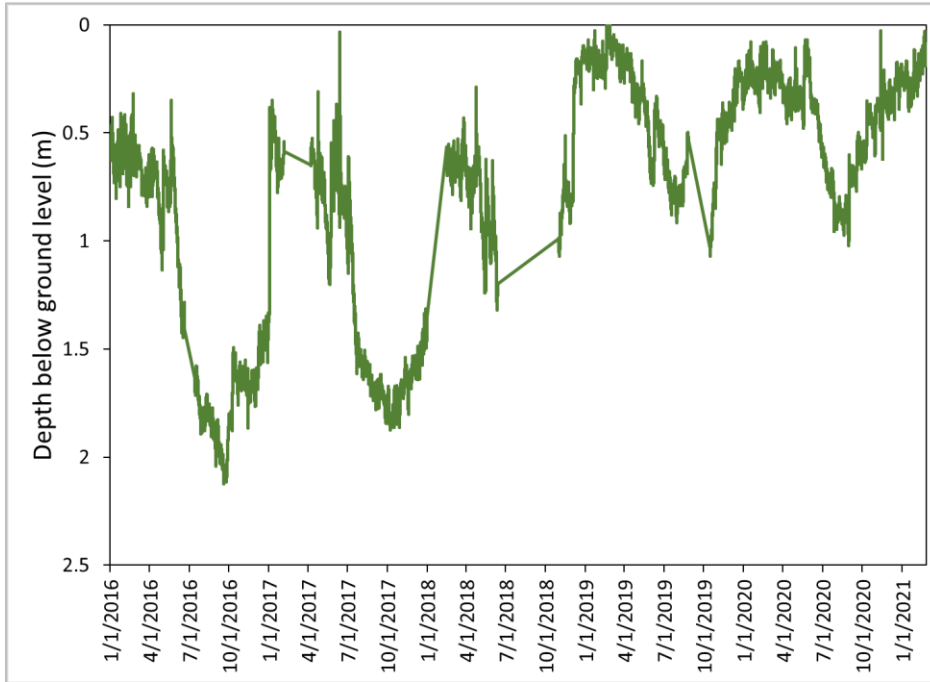


Figure 2.14 Depth to water table below land surface (m) in the riparian well A1R. Data represent the interval from January 2016 – March 2021.

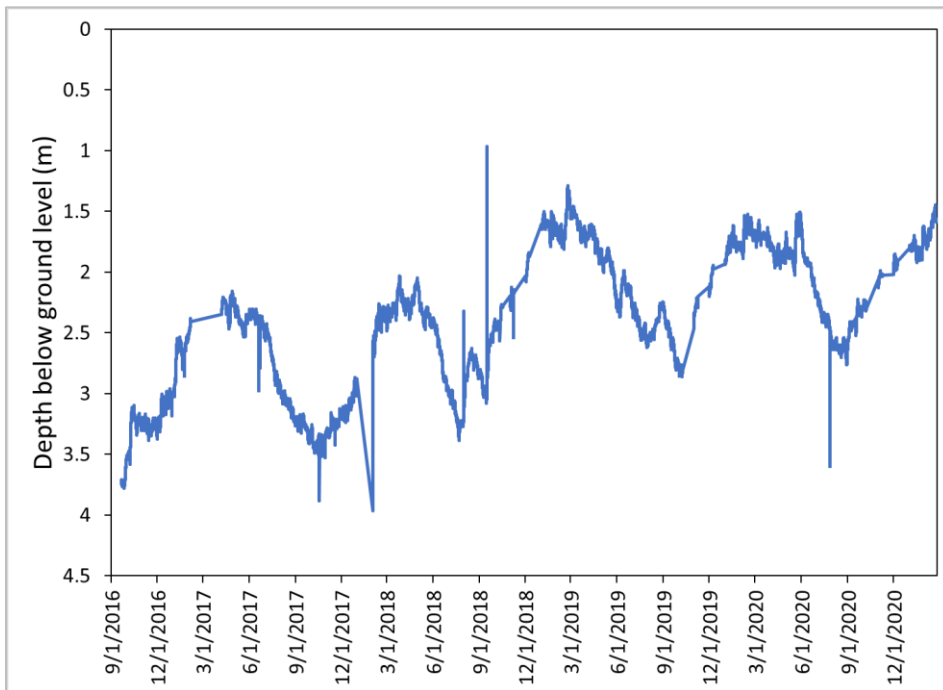


Figure 2.15 Depth to water table below land surface (m) in the upland well A1U. Data represent the interval from January 2016 – March 2021.

*Table 2.8 Water level analysis of A1R riparian groundwater. Asterisk * events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	197.18	2/23/2016	195.38	9/19/2016	1.81	196.27
2017	197.47	6/13/2017*	195.63	10/8/2017	1.84	196.26
2018	197.41	12/26/2018	196.11	1/1/2018	1.29	196.78
2019	197.33	02/22/2019	196.12	10/24/2019	1.21	196.86
2020	197.26	05/27/2020	196.30	07/27/2020	0.96	196.84

*Table 2.9 Water level analysis of A1U upland groundwater. Asterisk * events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	196.50	12/26/2016	195.68	9/26/2016	0.82	196.19
2017	197.39	12/20/2017	195.58	10/18/2017	1.80	196.60
2018	198.50	9/16/2018*	195.50	2/1/2018	3.00	197.06
2019	198.17	2/23/2019	196.51	10/16/2019	1.66	197.38
2020	197.96	5/29/2020	195.87	7/27/2020	2.09	197.45

2.4.1.5 R2 - Main-stem watershed

R2R is a riparian well along the lower Reedy Creek tributary downstream from a park. R2U is distant from the channel but it is located on the flat, floodplain-like surface rather than being a true “upland” well. R2U is also located in the middle of a wetland, with seasonally pooled water both pre- and post-restoration. Water levels from R2U (RCMW2) and R2R (RCMW2A) are reported in Figure 2.16. Annual statistics are presented Table 2.10 and Table 2.11. As with site P1, pre-restoration data suggest periods during which the riparian well water level was lower than the stream water level, implying periods that the stream water level was perched above the groundwater level.

While it is possible to have periods of stream water perched above poorly connected groundwater, the apparent difference may also be explained by the degrees of accuracy of the study design and survey techniques.

Overall, R2R and R2U exhibited a similar amount of seasonal variation in groundwater levels during the pre-restoration period. Pre-restoration, the seasonal variation in both wells was about 0.6 meters (Figure 2.16). Post-restoration, R2R exhibited more variable groundwater levels than R2U (Figure 2.16). The riparian well exhibited about 0.6 meters of seasonal groundwater variation post-restoration while the upland well exhibited less than 0.3 meters of seasonal variation, showing nearly constant groundwater levels except for a slight drop during the summer/fall dry period.

The water level difference between the riparian and upland wells differed seasonally. During the winter and summer periods pre-restoration, the upland well exhibited consistently higher groundwater levels than the riparian well. During the post-restoration period, the riparian well closely followed the upland well and exhibited only slightly lower groundwater levels than the upland well in winter/spring. The riparian well exhibited 0.3-0.6 meters lower water levels than the upland well in the summer/fall dry period.

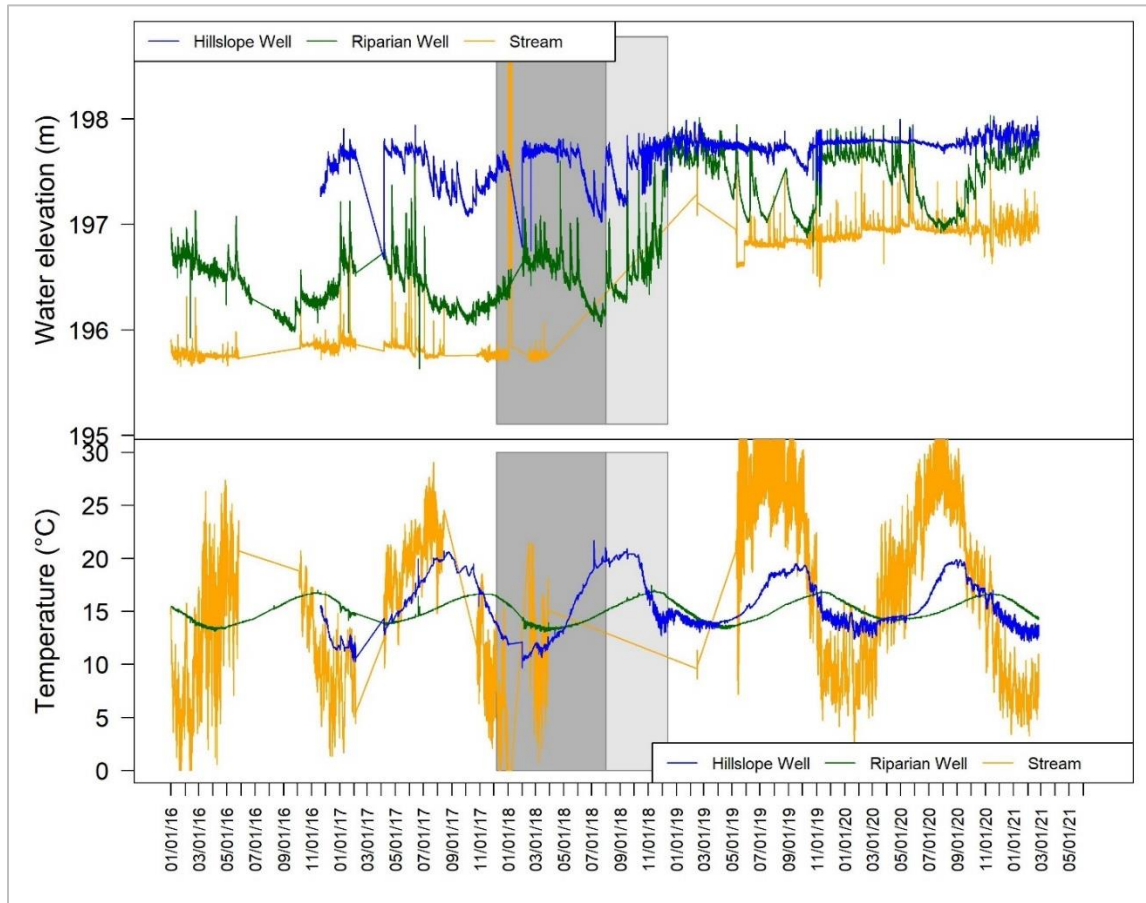


Figure 2.16 Water elevation (m) and temperature ($^{\circ}\text{C}$) for the R2 well transect site. The blue line is the hillslope well, green line is the riparian well, and orange line is the stream. Data represent entire sampling period from January 2016 – March 2021. Data are in ten-minute intervals. Grey bar represents the during-restoration period.

R2R and R2U were monitored continuously during the construction period. The riparian well exhibited a sharp increase in water levels between summer 2018 and winter 2019, in which the water level rose about 1.2 meters (Figure 2.17). The upland well exhibited a water level increase of up to 0.6 meters depending on season (Figure 2.18). R2R water levels shifted from up to 2.5 meters below the land surface pre-restoration to 0-0.9 meters below the land surface post-restoration (Figure 2.17). R2U water levels shifted from 0-0.6 meters below the land surface pre-restoration to 0-0.3 meters post-restoration (Figure 2.18). The difference in water levels between the riparian well and the

stream increased post-restoration. Pre-restoration, the R2R water level was up to 0.3 meters above the stream water level, or not above the stream water level as discussed above. Post-restoration, the R2R water level was about 1.2 meters higher than the stream water level in winter/spring and about 0.6 meters higher than the stream water level in summer/fall. This implies that the potential for baseflow has increased post-restoration, particularly during summer/fall when there could have been little or no potential for groundwater flow between R2R and the pre-restoration stream. In the headwaters sites D1, P1, and C1, this enhanced groundwater-stream gradient may protect the stream from drying during drought. Farther downstream along the Reedy Creek main stem, the stream likely has more established perennial flow, even pre-restoration, but the potential seems enhanced for groundwater baseflow to contribute to the stream.

Results of a Mann-Whitney test indicate that there was a statistically significant difference in stream water levels between the pre-restoration and post-restoration periods ($W = 2501928175$, $p < 2.2 \times 10^{-16}$). Results of a Mann-Whitney test suggest that there was a statistically significant difference in upland well water levels between the pre-restoration and post-restoration periods ($W = 1387711199$, $p < 2.2 \times 10^{-16}$). Furthermore, findings from a Mann-Whitney test indicate that there was a statistically significant difference in riparian well water levels between the pre-restoration and post-restoration periods ($W = 75450858$, $p < 2.2 \times 10^{-16}$).

2.4.1.5.1 Condition at end of study period

By the end of the study, it was evident that the riparian well had reached a new stable state, falling and rising seasonally. In the upland well, after restoration there was a shift from a pre-restoration seasonal wet and dry periods to a constantly wet period post-

restoration. The upland well exhibited a near-constant wet state where the wetland surrounding the upland well was constantly wet. Overall, the upland well tended to have higher water table levels (Figure 2.16) than the riparian well which is consistent with groundwater flow from the hillslope to the stream. After restoration, R2R exhibited water levels within 0.6 meters of the land surface year-round and less than 0.3 meters from the land surface in winter/spring (Figure 2.17). This demonstrates new potential for seasonal saturation in the riparian corridor at R2R. After restoration, R2U exhibited water levels at the land surface nearly year round (Figure 2.18), confirming a nearly perennial saturated conditions post-restoration, whereas the site was only seasonally saturated pre-restoration.

2.4.1.5.2 Inferred restoration effect

It is important to note that restoration apparently led to the wetland-surrounded, hillslope R2U well having similar increases in groundwater levels as the near-stream R2R well. This implies that restoration of the stream channel has the ability to induce changes in groundwater levels, not only in the near-stream riparian well, but also in the well some distance from the stream. Both R2R and R2U are located on the floodplain-like surface that is capable soil saturation and capable of being inundated by high flows. Throughout the study reach, restoration has led to increased water table heights and more robust seasonal baseflow. Overall, due to the increased in baseflow, the stream is less likely to dry in the post-restoration period.

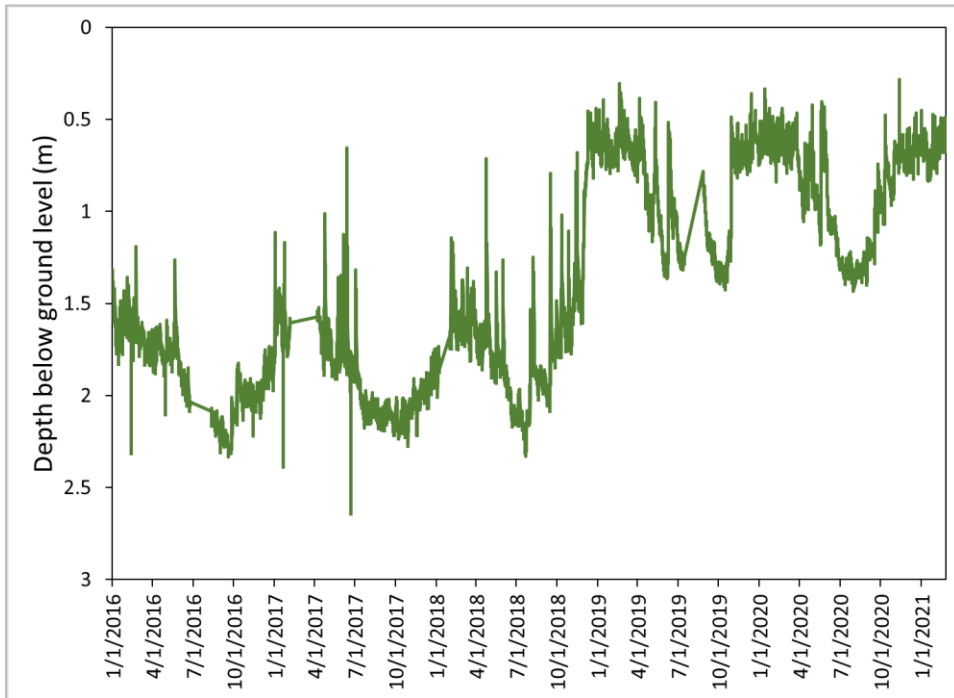


Figure 2.17 Depth to water table below land surface (m) in the riparian well R2R. Data represent the interval from January 2016 – March 2021.

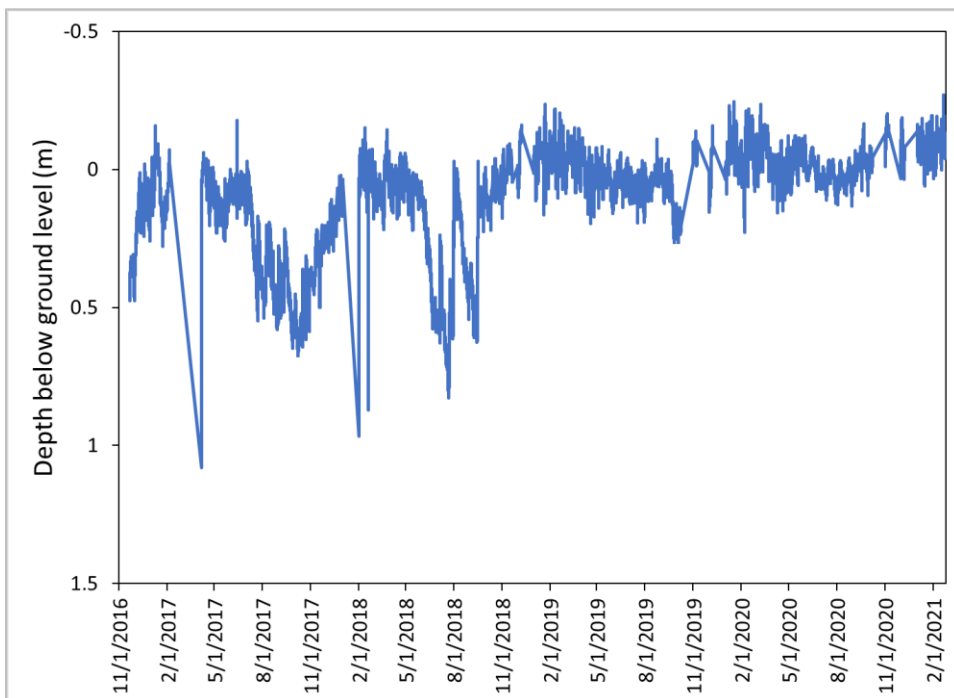


Figure 2.18 Depth to water table below land surface (m) in the upland well R2U. Data represent the interval from January 2016 – March 2021.

*Table 2.10 Water level analysis of R2R riparian groundwater. Asterisk * events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	196.46	2/23/2016	195.32	9/19/2016	1.14	195.79
2017	196.99	6/13/2017*	195.01	6/22/2017	1.98	195.75
2018	197.21	12/28/2018	195.32	7/22/2018	1.89	196.00
2019	197.35	2/19/2019	196.22	10/16/2019	1.12	196.78
2020	197.37	11/12/2020	196.22	7/31/2020	1.15	196.77

*Table 2.11 Water level analysis of R2U upland groundwater. Asterisk * events represent episodic recharge spikes discussed in text. However, overall average groundwater levels were highest during the winter/spring period. GW elev = groundwater elevation (m asl). Year = calendar year.*

Year	Yearly maximum gw elev (m asl)	Yearly maximum date	Yearly minimum groundwater elev (m asl)	Yearly minimum date	Annual gw elev range (m)	Mean annual gw elev (m asl)
2016	197.11	12/19/2016	196.61	11/21/2016	0.50	196.90
2017	197.27	6/13/2017*	196.01	4/7/2017	1.26	196.85
2018	197.26	12/26/2018	196.12	2/1/2018	1.14	196.97
2019	197.33	1/22/2019	196.82	10/12/2019	0.51	197.08
2020	197.36	11/19/2020	196.86	2/7/2020	0.50	197.12

2.4.1.6 Watershed changes in water levels

Overall, changes in water levels throughout the watershed were consistent across all well transects. Water levels increased from the pre-restoration levels to post-restoration levels at all riparian and upland well transects (Table 2.12). In all cases, the increase in water levels were statistically significant (Table 2.12). These findings are consistent with the hypotheses (RQ2.1).

Table 2.12 Changes in groundwater levels from the pre-restoration to the post-restoration period at all well locations. Statistically significant ($p < 0.05$) difference according to the Mann-Whitney test is indicated by *.

Site	Change from pre to post	Statistical Significance
D1U	Increased	*
D1R	Increased	*
P1U	Increased	*
P1R	Increased	*
C1U	Increased	*
C1R	Increased	*
A1U	Increased	*
A1R	Increased	*
R2U	Increased	*
R2R	Increased	*

2.4.2 (RQ2.2) How do the seasonal hydraulic gradients differ between the pre- and post-restoration stream?

2.4.2.1 D1 – Developed sub-watershed

Average annual D1 gradients from the upland well to the stream increased from a pre-restoration gradient of 0.01 to a post-restoration gradient of 0.03 (Figure 2.19) (Table 2.13). Throughout the monitoring period, the D1 transect exhibited seasonality with increased gradients in the summer periods and decreased gradients in the winter periods (Figure 2.19). However, the pre-restoration period exhibited larger seasonal variations in gradients when compared to the post-restoration period with the seasonal variation being 0.09 pre-restoration and 0.07 in the post-restoration period (Table 2.13). Minimum gradients shifted from the dormant season pre-restoration to the growing season post-restoration (Table 2.14). Maximum gradients switched from the growing season pre-restoration to the dormant season post-restoration period (Table 2.14). Results of a Mann-

Whitney test indicate that there was a statistically significant difference in total transect hydraulic gradients between the pre-restoration and post-restoration periods ($W = 1.11 \times 10^9$, $p < 2.2 \times 10^{-16}$).

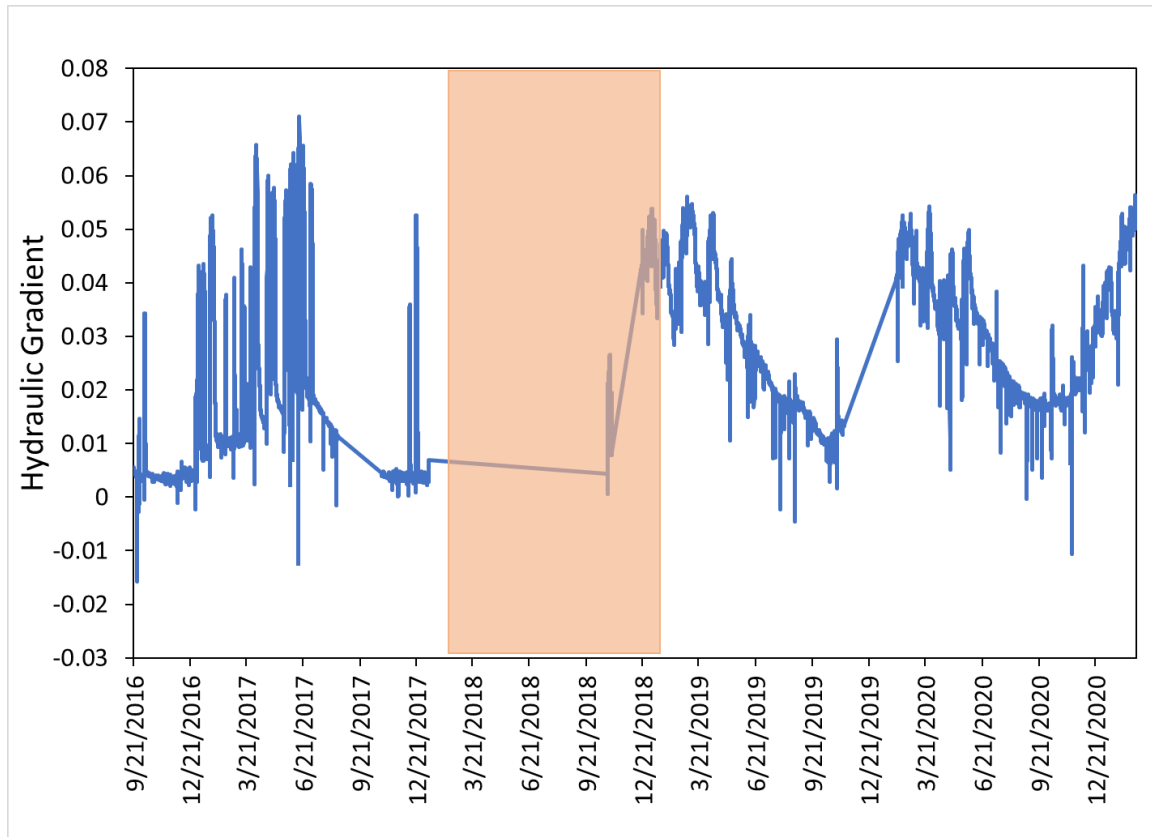


Figure 2.19 Total (upland to stream) hydraulic gradient at the D1 well transect. Study period September 2016 – March 2021. Orange bar represents the during-construction period.

Table 2.13 Descriptive statistics for D1 well transect total hydraulic gradient (upland to stream) in the pre- and post-restoration periods.

	Pre-Restoration (Jan 2016-Dec 2017)	Post-Restoration (Jan 2019 – Mar 2021)
Mean	0.01	0.03
Standard Error	0.00	0.00
Median	0.01	0.03
Mode	0.00	0.02
Standard Deviation	0.02	0.01
Range	0.09	0.07
Minimum	-0.02	-0.01
Maximum	0.07	0.06

Table 2.14 Gentlest and steepest monthly total (upland to stream) hydraulic gradient for each year at the D1 well transect.

	Gentlest gradient	Month	Steepest gradient	Month	Annual Mean
2016	/	/	/	/	/
2017	1.00×10^{-04}	Nov	7.11×10^{-02}	June	1.99×10^{-02}
2018	6.30×10^{-04}	Oct	5.22×10^{-02}	Dec	2.61×10^{-02}
2019	5.66×10^{-04}	Aug	5.61×10^{-02}	March	2.94×10^{-02}
2020	-4.82×10^{-05}	Aug	5.42×10^{-02}	March	2.98×10^{-02}
2021	/	/	/	/	/

2.4.2.2 P1 – Pond controlled / park developed sub-watershed

Overall, average annual P1 gradients from the upland well to the stream increased from a pre-restoration gradient of 0.003 to a post-restoration gradient of 0.008 (

Table 2.15). Throughout the monitoring period, the P1 transect exhibited seasonality with increased gradients in the summer periods and decreased gradients in the winter periods (Figure 2.20). Minimum gradients shifted from the growing season pre-restoration to the dormant season post-restoration (Table 2.16). Maximum gradients switched from the dormant season pre-restoration to the dormant and growing seasons post-restoration period (Table 2.16). Results of a Mann-Whitney test indicate that there was a statistically significant difference in total transect hydraulic gradients between the pre-restoration and post-restoration periods ($W = 1.29 \times 10^9$, $p < 2.2 \times 10^{-16}$).

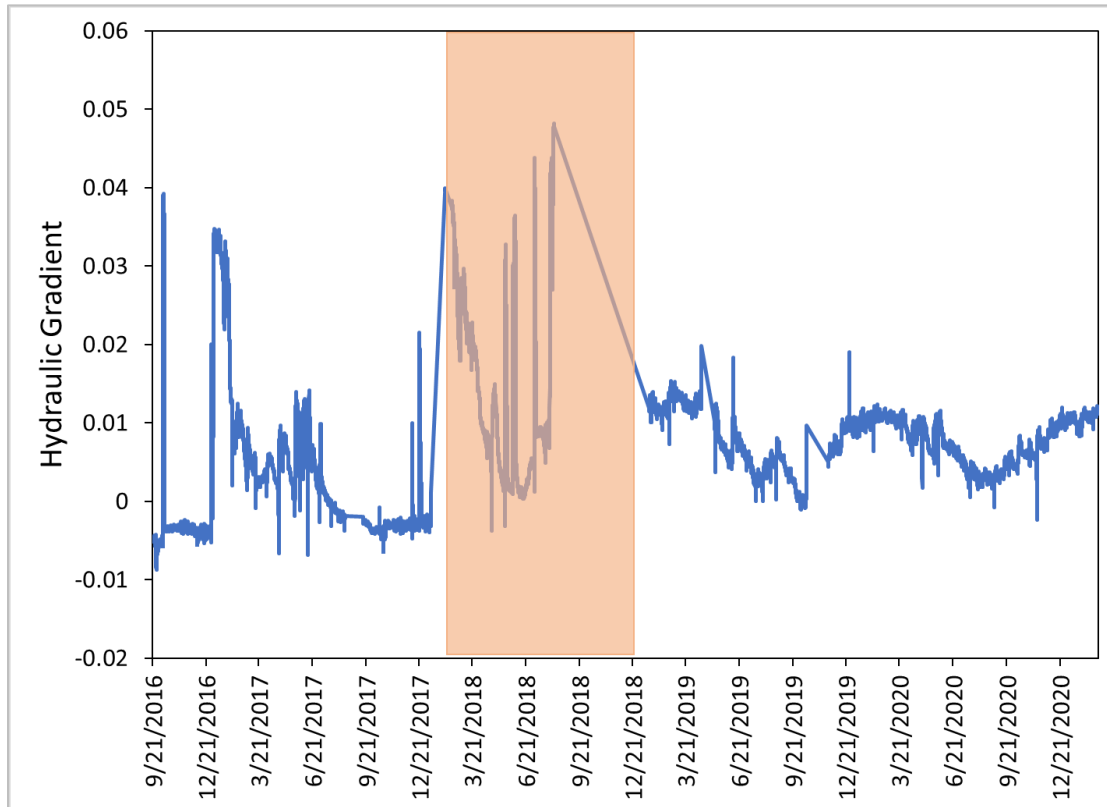


Figure 2.20 Total (upland to stream) hydraulic gradient at the P1 well transect. Study period September 2016 – March 2021. Orange bar represents the during-construction period.

Table 2.15 Descriptive statistics for P1 well transect total hydraulic gradient (upland to stream) in the pre- and post-restoration periods.

	<i>Pre-Restoration (Jan 2016- Dec 2017)</i>	<i>Post-Restoration (Jan 2019 – Mar 2021)</i>
Mean	3.16×10^{-03}	7.80×10^{-03}
Standard Error	3.78×10^{-05}	1.03×10^{-05}
Median	9.71×10^{-05}	7.98×10^{-03}
Mode	-3.47×10^{-03}	1.08×10^{-02}
Standard Deviation	9.22×10^{-03}	3.30×10^{-03}
Range	4.80×10^{-02}	2.22×10^{-02}
Minimum	-8.74×10^{-03}	-2.35×10^{-03}
Maximum	3.93×10^{-02}	1.99×10^{-02}

Table 2.16 Gentlest and steepest monthly total (upland to stream) hydraulic gradient for each year at the P1 well transect.

	Gentlest gradient	Month	Steepest gradient	Month	Annual Mean
2016	/	/	/	/	/
2017	4.54×10^{-07}	July	3.48×10^{-02}	Jan	4.53×10^{-03}
2018	-2.00×10^{-04}	July	4.82×10^{-02}	Aug	1.29×10^{-02}
2019	-1.07×10^{-06}	Oct	1.99×10^{-02}	April	7.71×10^{-03}
2020	5.74×10^{-05}	Oct	1.23×10^{-02}	Jan	7.46×10^{-03}
2021	/	/	/	/	/

2.4.2.3 C1 – Control sub-watershed

Overall, average annual C1 gradients from the upland well to the stream decreased from a pre-restoration gradient of 0.021 to a post-restoration gradients of 0.016 (Table 2.17). Throughout the monitoring period, the C1 transect exhibited seasonality with increased gradients in the summer periods and decreased gradients in the winter periods (Figure 2.21). Minimum gradients shifted from the dormant season pre-restoration to the dormant and growing season post-restoration (Table 2.18). Maximum gradients switched from the growing season pre-restoration to the dormant season post-restoration period (Table 2.18). Results of a Mann-Whitney test indicate that there was a statistically significant difference in total transect hydraulic gradients between the pre-restoration and post-restoration periods ($W = 3.86 \times 10^{09}$, $p > 2.2 \times 10^{-16}$).

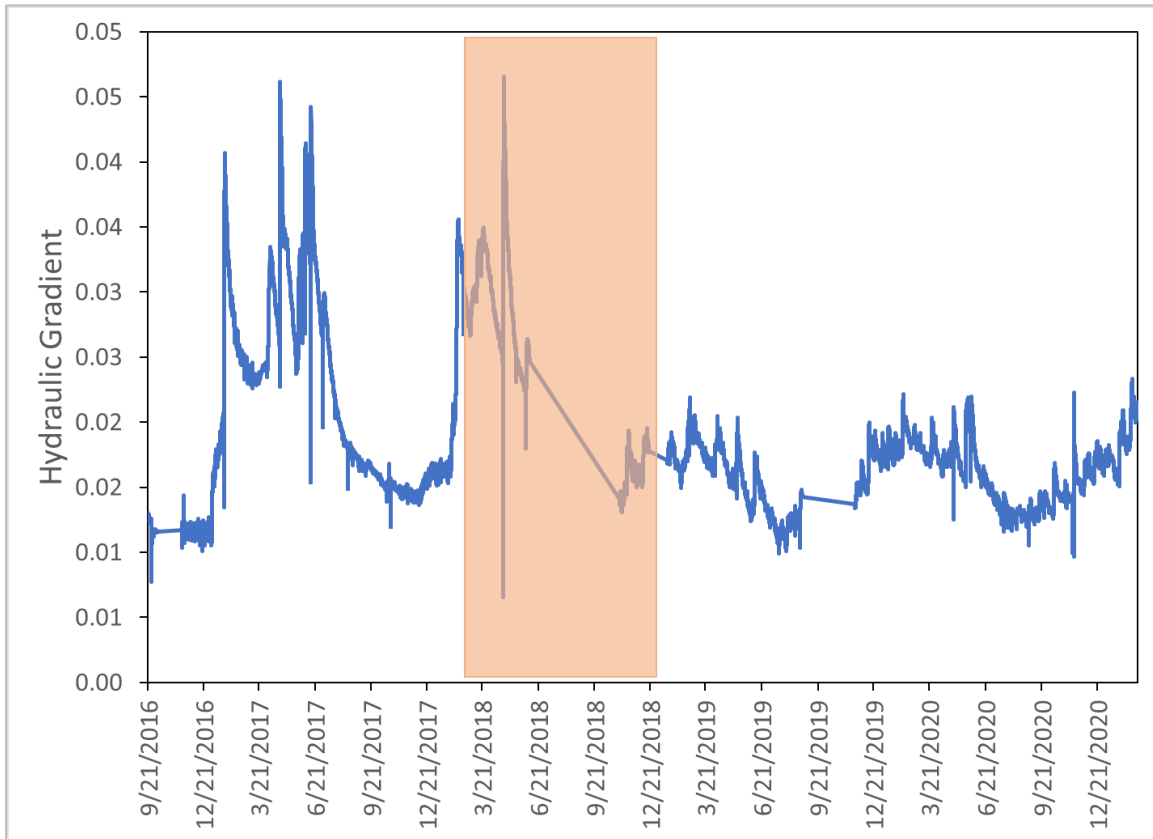


Figure 2.21 Total (upland to stream) hydraulic gradient at the C1 well transect. Study period September 2016 – March 2021. Orange bar represents the during-construction period.

Table 2.17 Descriptive statistics for C1 well transect total hydraulic gradient (upland to stream) in the pre- and post-restoration periods.

	Pre-Restoration (Jan 2016-Dec 2017)	Post-Restoration (Jan 2019 – Mar 2021)
Mean	2.11×10^{-02}	1.61×10^{-02}
Standard Error	3.19×10^{-05}	7.76×10^{-06}
Median	1.82×10^{-02}	1.64×10^{-02}
Mode	1.68×10^{-02}	1.69×10^{-02}
Standard Deviation	7.84×10^{-03}	2.43×10^{-03}
Range	3.84×10^{-02}	1.37×10^{-02}
Minimum	7.73×10^{-03}	9.68×10^{-03}
Maximum	4.61×10^{-02}	2.34×10^{-02}

Table 2.18 Gentlest and steepest monthly total (upland to stream) hydraulic gradient for each year at the C1 well transect.

	Gentlest gradient	Month	Steepest gradient	Month	Annual Mean
2016	/	/	/	/	/
2017	1.05×10^{-02}	Jan	4.61×10^{-02}	April	2.26×10^{-02}
2018	6.55×10^{-03}	April	4.65×10^{-02}	April	2.43×10^{-02}
2019	9.90×10^{-03}	July	2.19×10^{-02}	Feb	1.58×10^{-02}
2020	9.68×10^{-03}	Nov	2.23×10^{-02}	Nov	1.60×10^{-02}
2021	/	/	/	/	/

2.4.2.4 A1 – Agricultural sub-watershed

Pre-restoration data for the A1 transects includes data only from the year of 2016.

Results from the A1 upland well to the stream indicate that gradients increased from a pre-restoration average of 0.017 to a post-restoration average of 0.024 (Table 2.19).

Minimum gradients were seen during the growing and dormant seasons and while maximum gradients were seen in the dormant season post-restoration period (Table 2.20).

Overall, variation in monthly gradients was lower during the pre-restoration period (1.93×10^{-2}) when compared to post-restoration (4.45×10^{-2}) (Table 2.19). Furthermore, findings from a Mann-Whitney test indicate that there was a statistically significant difference in hydraulic gradients from the riparian well and the stream between the pre-restoration and post-restoration periods ($W = 3.07 \times 10^{-8}$, $p = 2.2 \times 10^{-16}$).

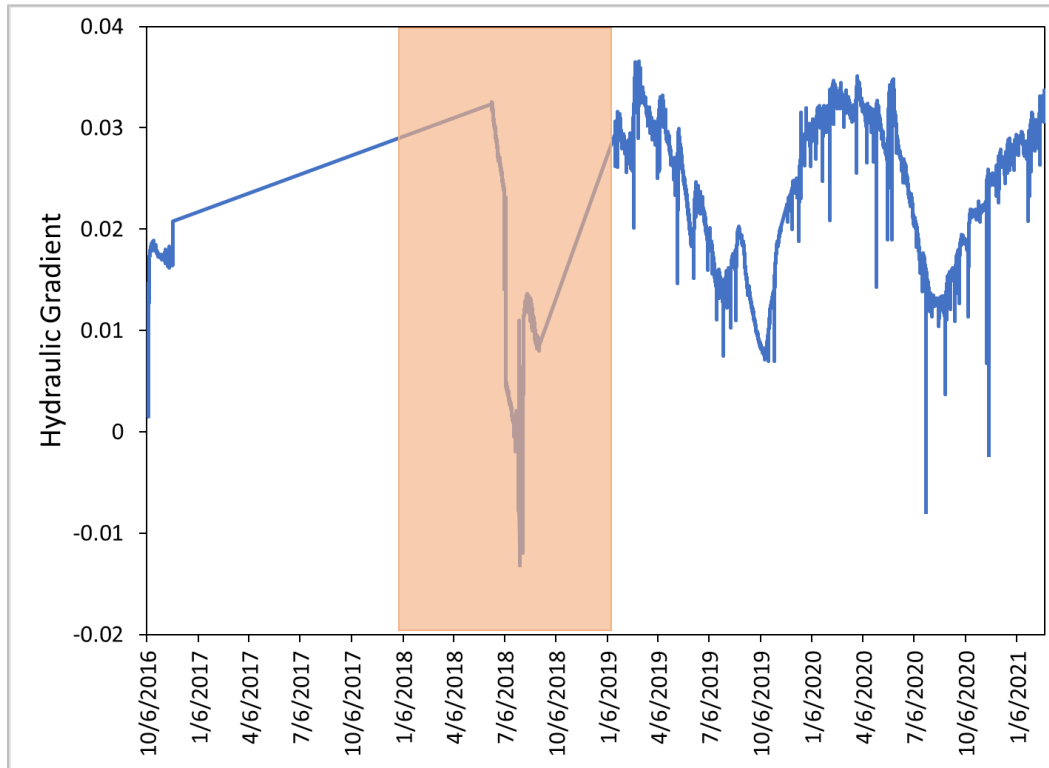


Figure 2.22 Total (upland to stream) hydraulic gradient at the A1 well transect. Study period September 2016 – March 2021. Orange bar represents the during-construction period.

Table 2.19 Descriptive statistics for A1 well transect total hydraulic gradient (upland to stream) in the pre- and post-restoration periods.

	Pre-Restoration (Jan 2016-Dec 2017)	Post-Restoration (Jan 2019 – Mar 2021)
Mean	1.72×10^{-02}	2.43×10^{-02}
Standard Error	1.37×10^{-05}	2.15×10^{-05}
Median	1.74×10^{-02}	2.62×10^{-02}
Mode	1.74×10^{-02}	2.80×10^{-02}
Standard Deviation	1.58×10^{-03}	7.10×10^{-03}
Range	1.93×10^{-02}	4.45×10^{-02}
Minimum	1.50×10^{-03}	-7.88×10^{-03}
Maximum	2.08×10^{-02}	3.66×10^{-02}

Table 2.20 Gentlest and steepest monthly total (upland to stream) hydraulic gradient for each year at the A1 well transect.

	Gentlest gradient	Month	Steepest gradient	Month	Annual Mean
2016	/	/	/	/	/
2017	/	/	/	/	/
2018	-4.07×10^{-07}	Sept	3.26×10^{-02}	June	1.27×10^{-02}
2019	6.98×10^{-03}	Oct	3.66×10^{-02}	March	2.24×10^{-02}
2020	2.08×10^{-02}	June	3.37×10^{-02}	Feb	2.96×10^{-02}
2021	/	/	/	/	/

2.4.2.5 R2 – Main-stem sub-watershed

Overall, average annual R2 gradients from the upland well to the stream decreased from a pre-restoration gradient of 0.029 to a gradient toward the stream in the post-restoration period with a gradient 0.023 (Table 2.21). Throughout the monitoring period, the R2 transect exhibited seasonality with increased gradients in the summer periods and decreased gradients in the winter periods (Figure 2.23). However, the pre-restoration period exhibited larger seasonal variations in gradients when compared to the post-restoration period with the seasonal variation being 0.038 pre-restoration and 0.039 in the post-restoration period (Table 2.21). Minimum gradients shifted from the growing season pre-restoration to the dormant season post-restoration (Table 2.22). Maximum gradients remained in the growing season for both the pre-restoration and post-restoration periods and included the dormant season post-restoration (Table 2.22). Results of a Mann-Whitney test indicate that there a statistically significant difference in total transect hydraulic gradients between the pre-restoration and post-restoration periods ($W = 1.1081 \times 10^{10}$, $p > 2.2 \times 10^{-16}$).

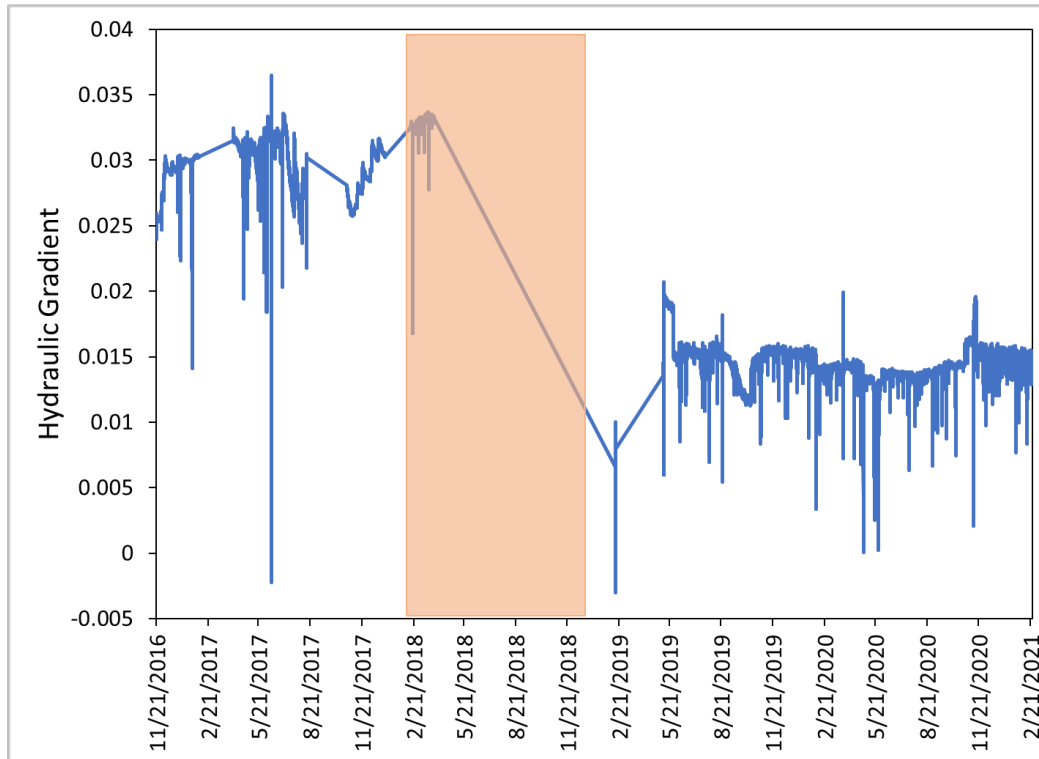


Figure 2.23 Total (upland to stream) hydraulic gradient at the R2 well transect. Study period September 2016 – March 2021. Orange bar represents the during-construction period.

Table 2.21 Descriptive statistics for R2 well transect total hydraulic gradient (upland to stream) in the pre- and post-restoration periods.

	Pre-Restoration (Jan 2016-Dec 2017)	Post-Restoration (Jan 2019 – Mar 2021)
Mean	2.96×10^{-02}	2.33×10^{-02}
Standard Error	5.76×10^{-06}	1.75×10^{-05}
Median	3.00×10^{-02}	2.72×10^{-02}
Mode	3.00×10^{-02}	3.32×10^{-02}
Standard Deviation	1.97×10^{-03}	8.11×10^{-03}
Range	3.88×10^{-02}	3.95×10^{-02}
Minimum	-2.25×10^{-03}	-3.02×10^{-03}
Maximum	3.65×10^{-02}	3.65×10^{-02}

Table 2.22 Gentlest and steepest monthly total (upland to stream) hydraulic gradient for each year at the R2 well transect.

	Gentlest gradient	Month	Steepest gradient	Month	Annual Mean
2016	/	/	/	/	/
2017	-2.70×10^{-04}	June	3.65×10^{-02}	June	2.98×10^{-02}
2018	1.68×10^{-02}	Feb	3.37×10^{-02}	March	3.29×10^{-02}
2019	3.79×10^{-04}	Feb	2.07×10^{-02}	May	1.49×10^{-02}
2020	7.69×10^{-03}	Nov	1.58×10^{-02}	Oct	1.47×10^{-02}
2021	/	/	/	/	/

2.4.2.6 Watershed changes in hydraulic gradients

Overall, changes in water levels throughout the watershed were not consistent across all transects. Hydraulic gradients increased in three upland-stream transects (D1, P1, and A1) and decreased in two of five transects (C1, R2) (Table 2.23). In all transects, the changes in hydraulic gradients were statistically significant (Table 2.23). These findings are not consistent with the hypotheses (RQ2.2).

Table 2.23 Changes in hydraulic gradients throughout the watershed from the pre-restoration to the post-restoration period. Statistically significant ($p < 0.05$) difference according to the Mann-Whitney test is indicated by *.

Site	Change from pre to post	Statistical Significance
D1	Increase	*
P1	Increase	*
C1	Decrease	*
A1	Increase	*
R2	Decrease	*

2.4.3 (RQ2.3) Does baseflow increase between the pre- and post-restoration stream?

Here, storms were classified based on precipitation amounts and length of storm event. Hydrograph separations were attempted on 1) high intensity, short-duration, 2) high intensity, long-duration, 3) low intensity, short-duration, and 4) low intensity, long-duration storms. High intensity storms encompassed storms with total storm precipitation was greater than 16 mm while low intensity storms encompassed events from one to 15 mm. Long-duration storms were classified by lasting six or more hours while short-duration storms were shorter than six hours (Table 4.11).

Pre-restoration, low intensity storm flows (both long and short duration (Figure 2.24; Figure 2.26)) were dominated by old water endmember discharge with percentages ranging from 86% to 101% (Table 2.24). Here, low intensity storms were represented by changes in SC ranging from 10 $\mu\text{S}/\text{cm}$ to 25 $\mu\text{S}/\text{cm}$. High intensity storms (both long and short duration (Figure 2.25; Figure 2.27)) pre-restoration were dominated by old water endmember discharge and were represented by changes in SC ranging from 55 $\mu\text{S}/\text{cm}$ to 60 $\mu\text{S}/\text{cm}$ (Table 2.24). In the post-restoration period, all storms were dominated by old water endmember discharge with percentages ranging from 57% to 87% and with changes in SC ranging from 30 $\mu\text{S}/\text{cm}$ to 70 $\mu\text{S}/\text{cm}$ (Figure 2.28; Figure 2.29; Figure 2.30; Figure 2.31). Results of a Mann-Whitney test indicate that there was not a statistically significant difference in old water endmember percentage contributions ($W = 15$, $p = 0.05$) or new water endmember percentage contributions ($W = 1$, $p = 0.05$) pre- to post-restoration. Major endmember percentage contributions did not change from pre-restoration to post-restoration as stormflows were dominated by old water discharge

during both time periods. Precipitation decreased from the pre-restoration to post-restoration period (see chapter 4, RQ4.1), while simultaneously decreasing the percentage of old water endmember contributions to stormflows. Here, restoration led to an increase in new water endmember contributions to stormflows over time.

Comparing pre-restoration to post-restoration endmember discharge amounts, results indicate that the old water endmember discharge amounts (i.e.: baseflow) were on average higher in the post-restoration period (Table 2.25). Pre-restoration, average baseflow amounts from the hydrograph separation was $0.04 \text{ m}^3/\text{s}$ while baseflow amounts increased in the post-restoration period to $0.12 \text{ m}^3/\text{s}$ (Table 2.25). Findings from a Mann-Whitney test indicate that there was a statistically significant difference in old water endmember amounts between the pre-restoration and post-restoration periods ($W = 4586729$, $p < 2.2 \times 10^{-16}$). Results indicate that there was an increase in the new water endmember amounts (i.e.: precipitation addition) in the post-restoration period (Table 2.25). Pre-restoration, average stormflow amounts from the hydrograph separation was $1.93 \times 10^{-03} \text{ m}^3/\text{s}$ while baseflow amounts increased in the post-restoration period to $0.05 \text{ m}^3/\text{s}$ (Table 2.25). Furthermore, findings from a Mann-Whitney test indicate that there was a statistically significant difference in old water endmember amounts between the pre-restoration and post-restoration periods ($W = 2958435$, $p < 2.2 \times 10^{-16}$).

Table 2.24 Percentage of endmember contributions for each storm and relative change in stream SC ($\mu\text{s/cm}$) driven by precipitation events. SC data were from the CMANN station (MY13A) located at the USGS (0212427947) monitoring station where discharge data were also taken.

	Date	Q Old Percentage	Q New Percentage	Change in SC ($\mu\text{s/cm}$)	Storm category
Pre-restoration	4/22/2016	101	-1	~25	low intensity long duration
	6/3/2016	91	9	~60	high intensity short duration
	6/30/2016	97	3	~10	low intensity short duration
	1/13/2018	86	14	~55	high intensity long duration
Post-restoration	11/25/2019	75	25	~30	low intensity short duration
	12/11/2019	57	43	~70	low intensity long duration
	1/25/2020	75	25	~60	high intensity long duration
	12/6/2020	87	13	~45	high intensity short duration

Table 2.25 Descriptive statistics for hydrograph separation of the old and new end member amounts in the pre- and post-restoration period. Q = discharge (m^3/s). Pre = pre-restoration (January 2016-December 2017). Post = post-restoration (January 2019 – March 2021).

	<i>Pre Q old</i>	<i>Post Q old</i>
Mean	0.04	0.12
Standard Error	7.42×10^{-04}	3.40×10^{-03}
Median	0.04	0.05
Mode	0.07	0.05
Range	0.53	2.42
Minimum	0.01	0.02
Maximum	0.54	2.44
	<i>Pre Q new</i>	<i>Post Q new</i>
Mean	0.00193	0.05
Standard Error	2.59×10^{-04}	2.49×10^{-03}
Median	4.13×10^{-04}	0.01
Mode	-1.07×10^{-02}	2.28×10^{-04}
Range	0.40	1.56
Minimum	-0.03	-0.01
Maximum	0.37	1.55
Number of storms	4	4

Furthermore, results from a baseflow index indicate that flashiness at the watershed outlet negligibly decreased over time (i.e.: the stream did not become significantly more baseflow-dominated pre- vs. post-restoration). Here, calculation using daily average discharge for the pre-restoration period resulted in a baseflow index score of 0.58, indicating that 58% of runoff at the watershed outlet occurred as baseflow. In the post-restoration period, daily average discharge resulted in a baseflow index score of 0.57, indicating that 57% of runoff at the watershed outlet occurred as baseflow.

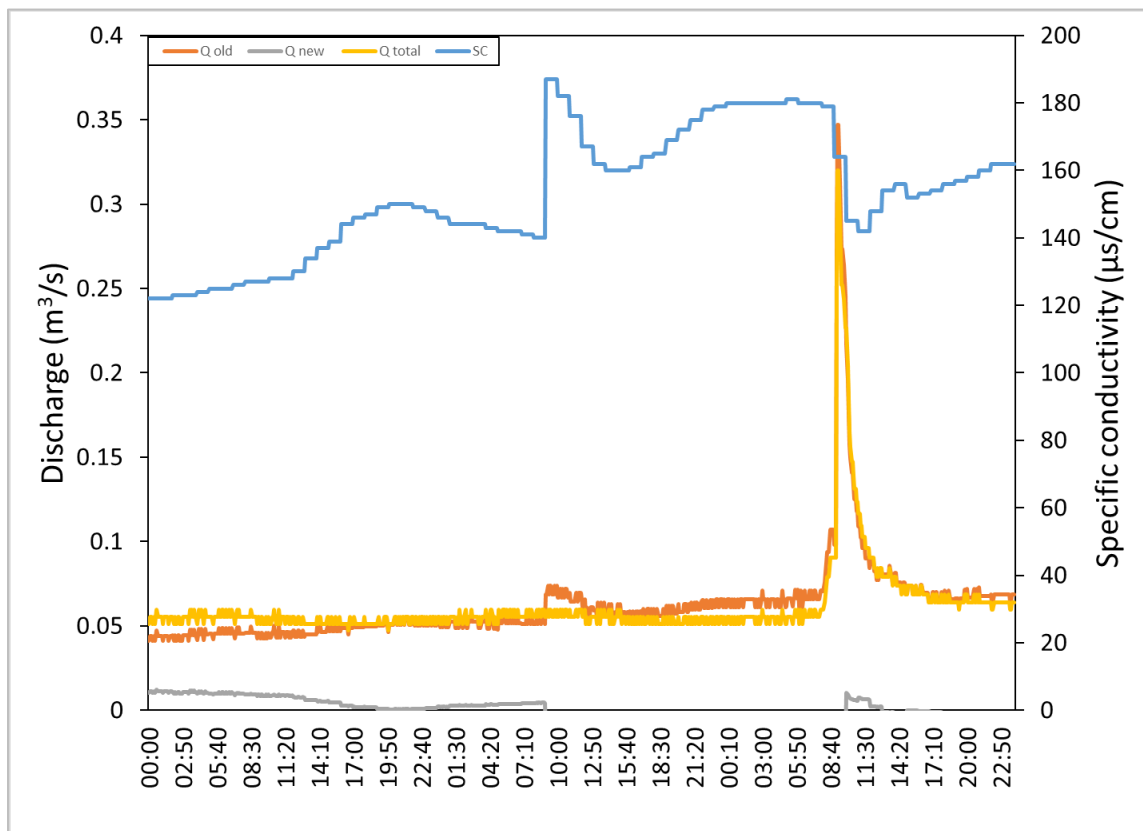


Figure 2.24 Hydrograph separation for a pre-restoration storm on 4/22/2016, showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which most runoff is inferred to be from the old water component.

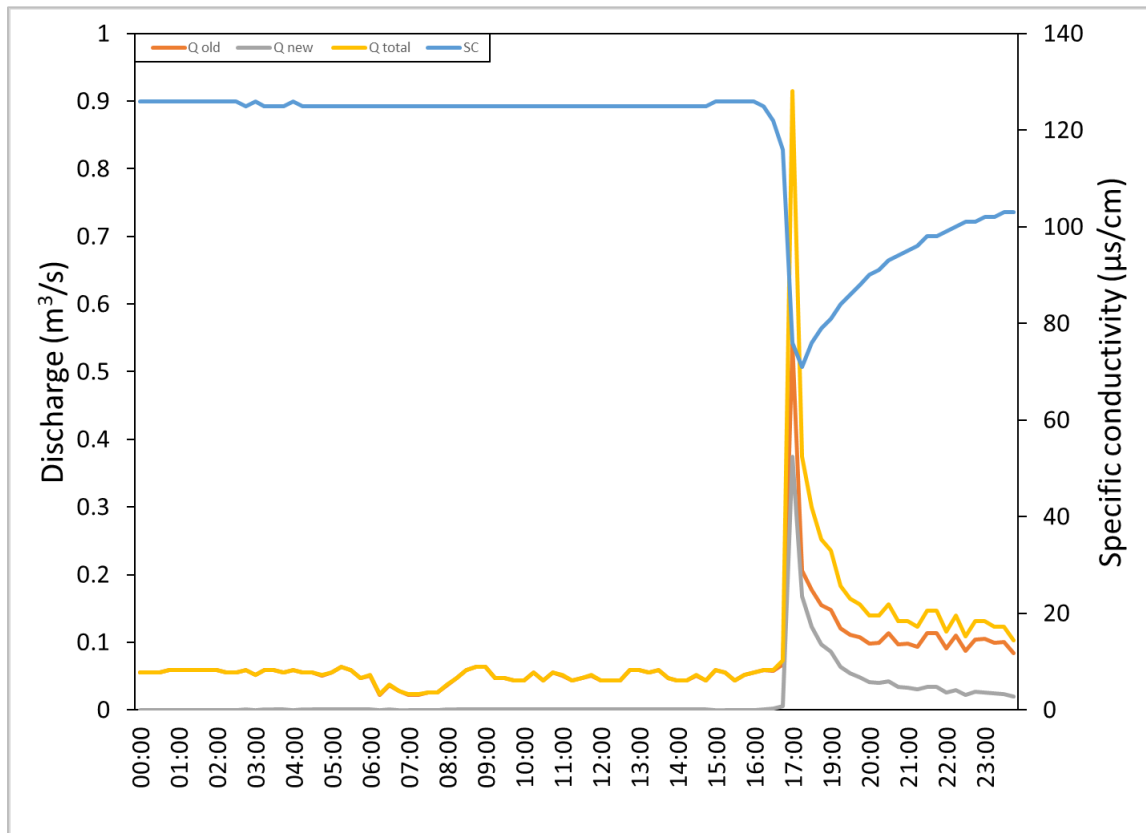


Figure 2.25 Pre-restoration hydrograph separation for 6/03/2016 showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which a major contributing factor to runoff is inferred to be from the old water component.

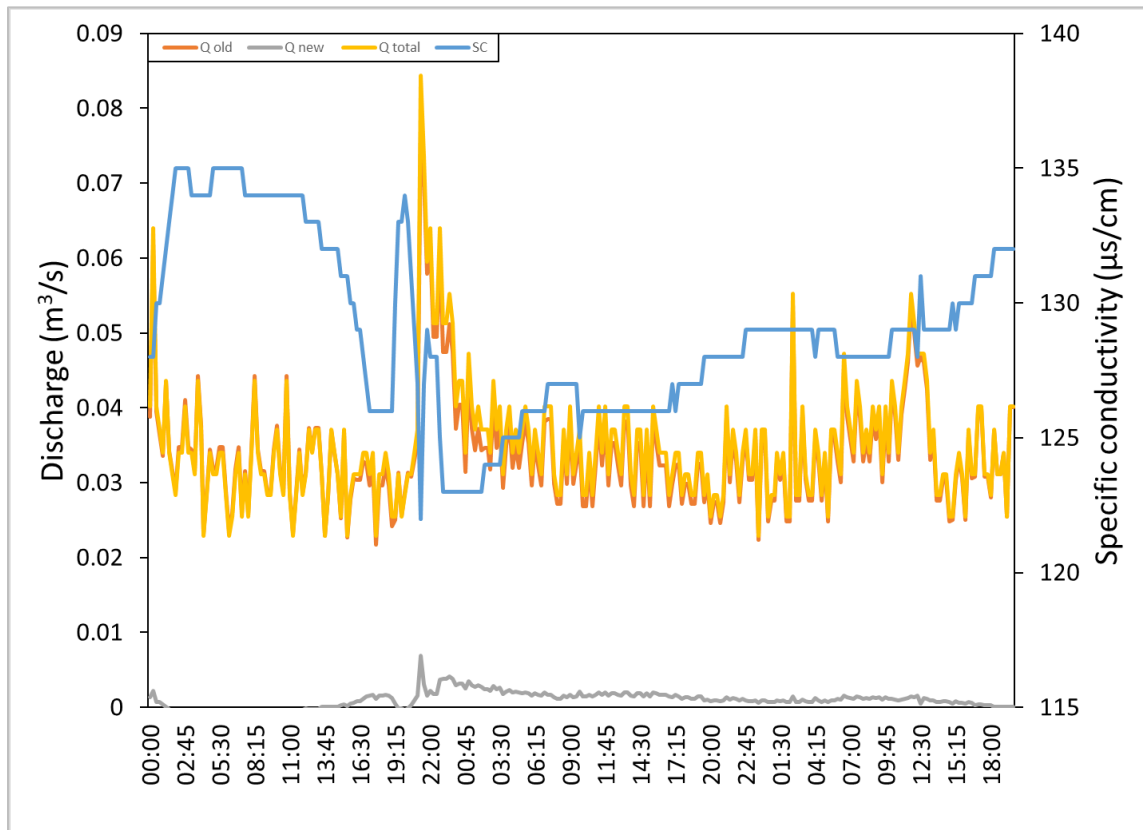


Figure 2.26 Pre-restoration hydrograph separation for 6/30/2016 showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which most runoff is inferred to be from the old water component.

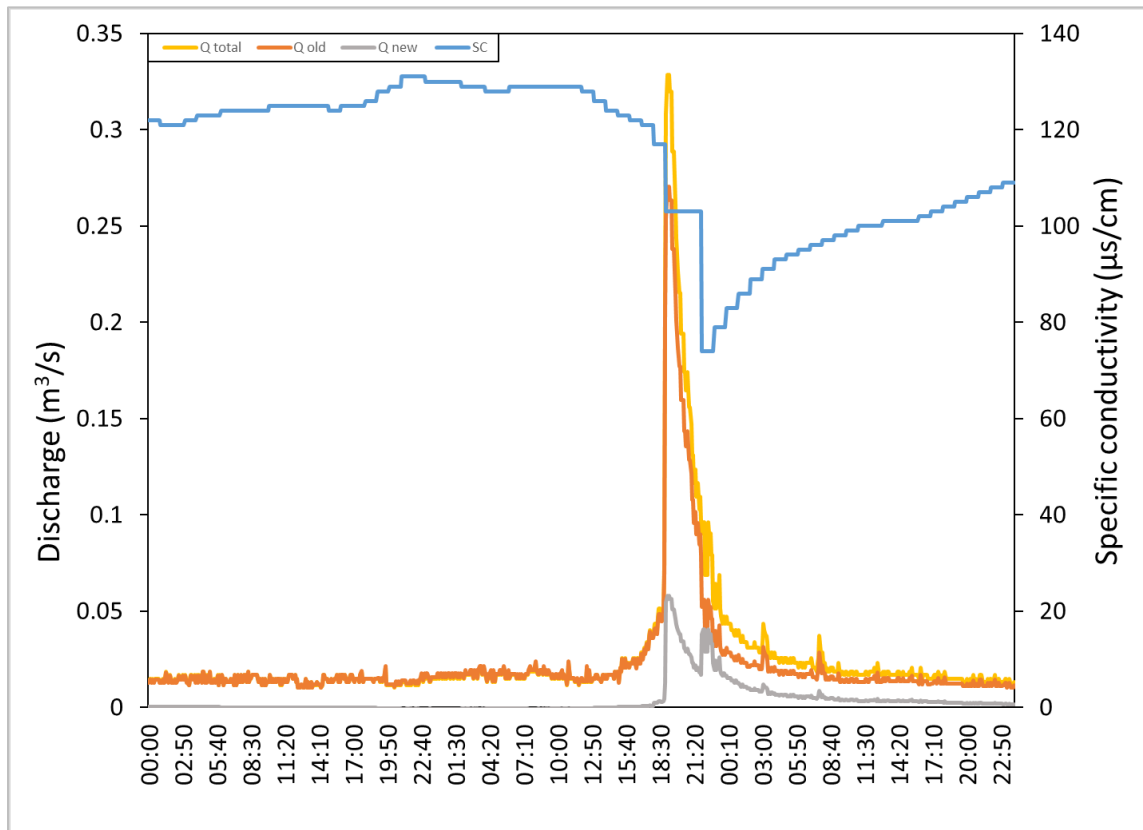


Figure 2.27 Pre-restoration hydrograph separation for storm on 1/13/2018 showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which a major contributing factor to runoff is inferred to be from the old water component.

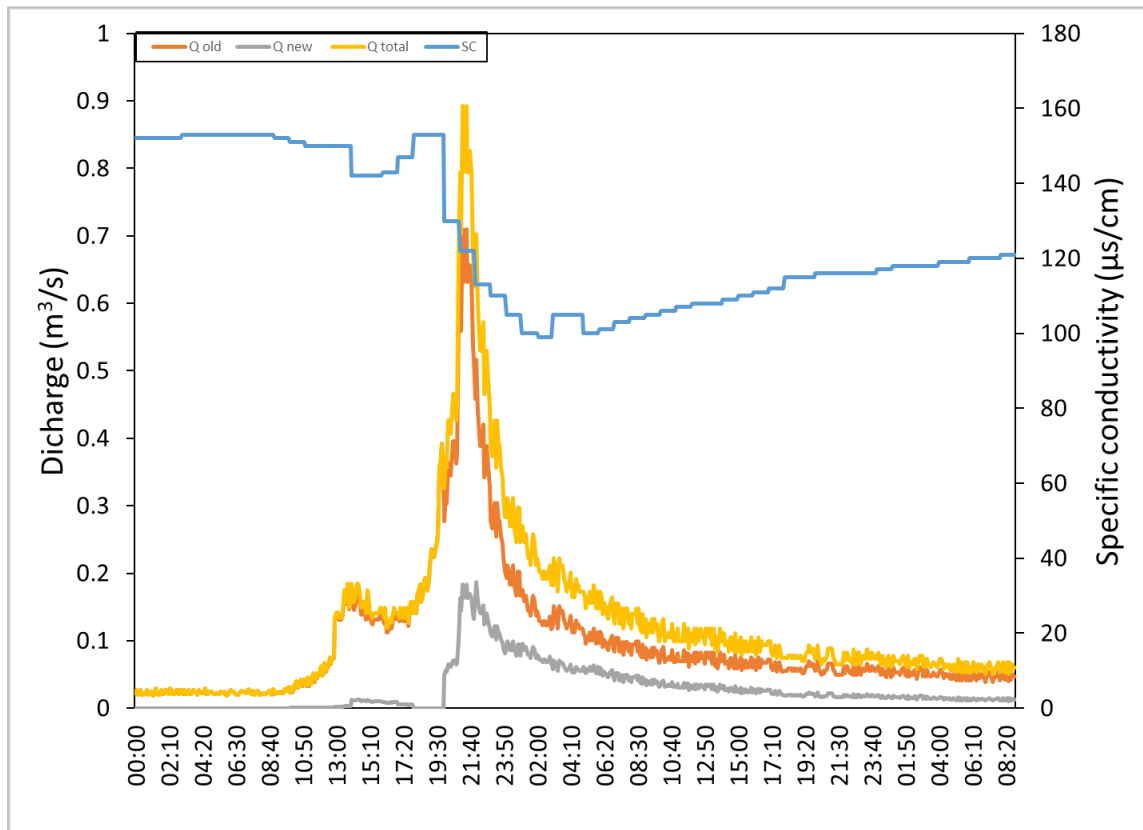


Figure 2.28 Post-restoration hydrograph separation for 11/25/2019 showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which a major contributing factor to runoff is inferred to be from the old water component.

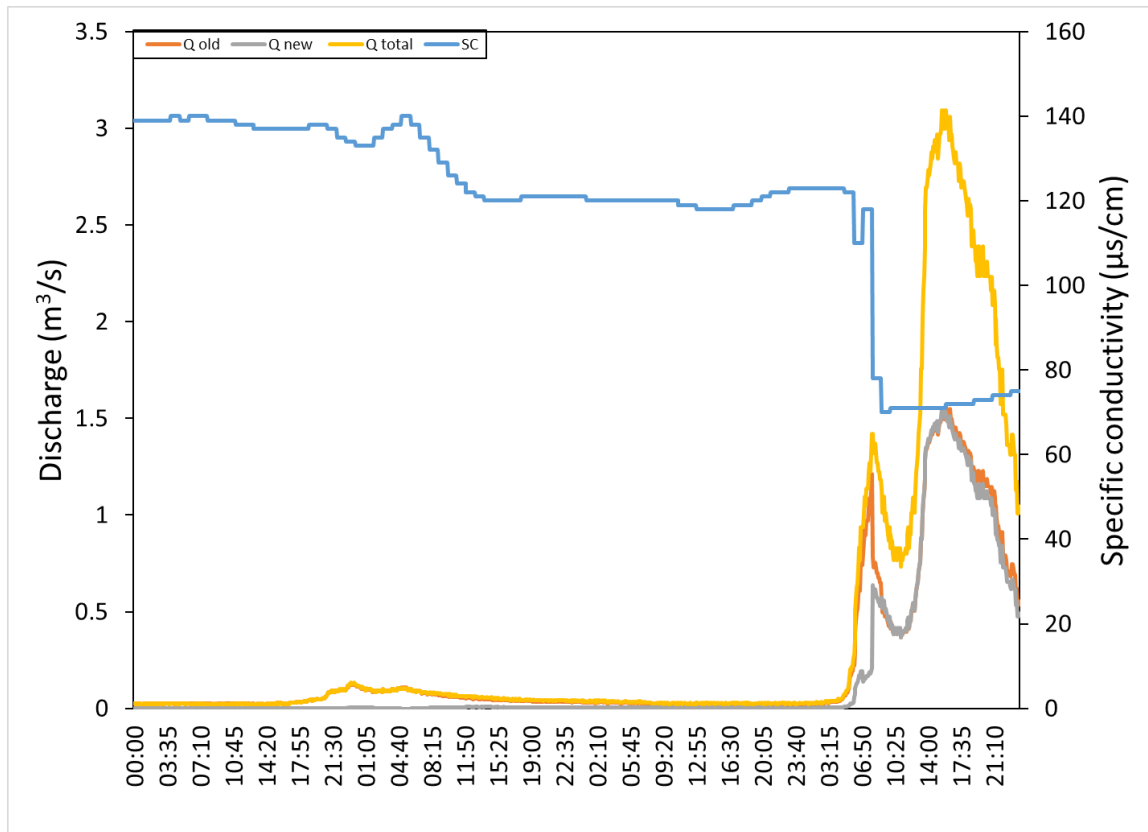


Figure 2.29 Post-restoration hydrograph separation for 12/11/2019 showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which a major contributing factor to runoff is inferred to be from the old water component.

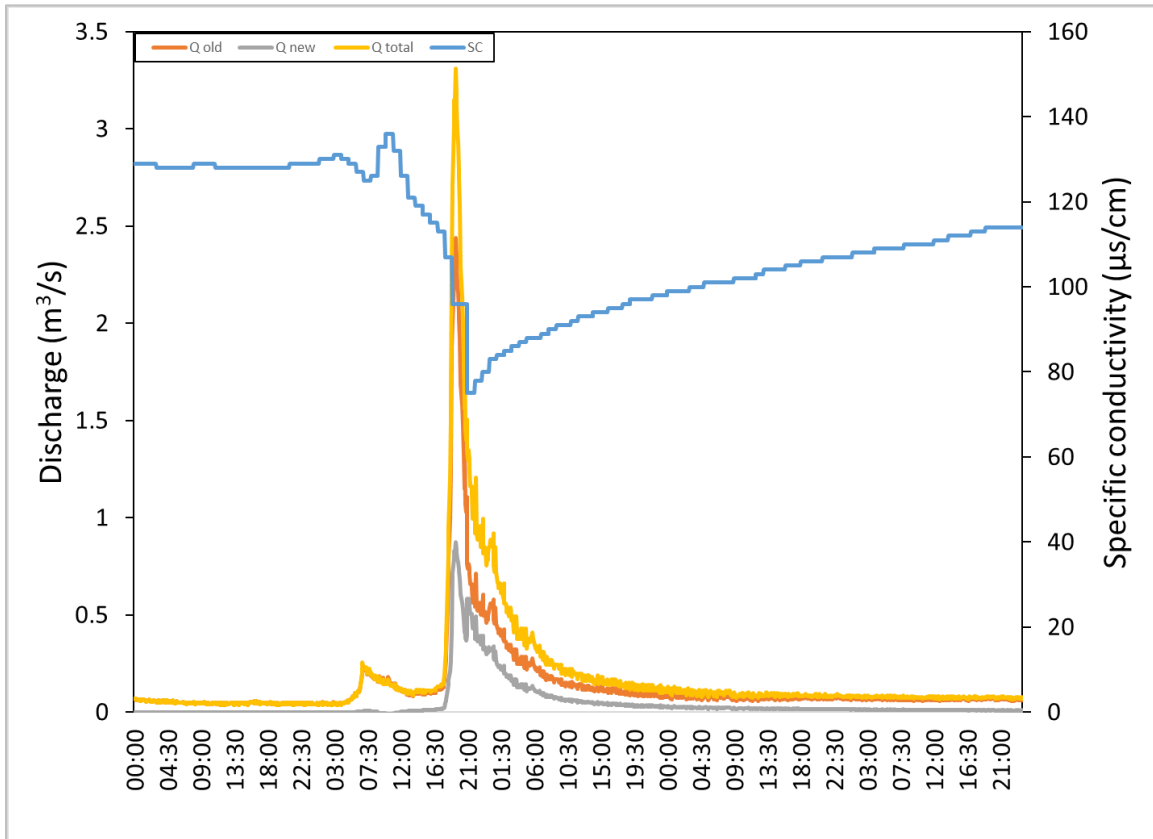


Figure 2.30 Post-restoration hydrograph separation for 1/25/2020 showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which a major contributing factor to runoff is inferred to be from the old water component.

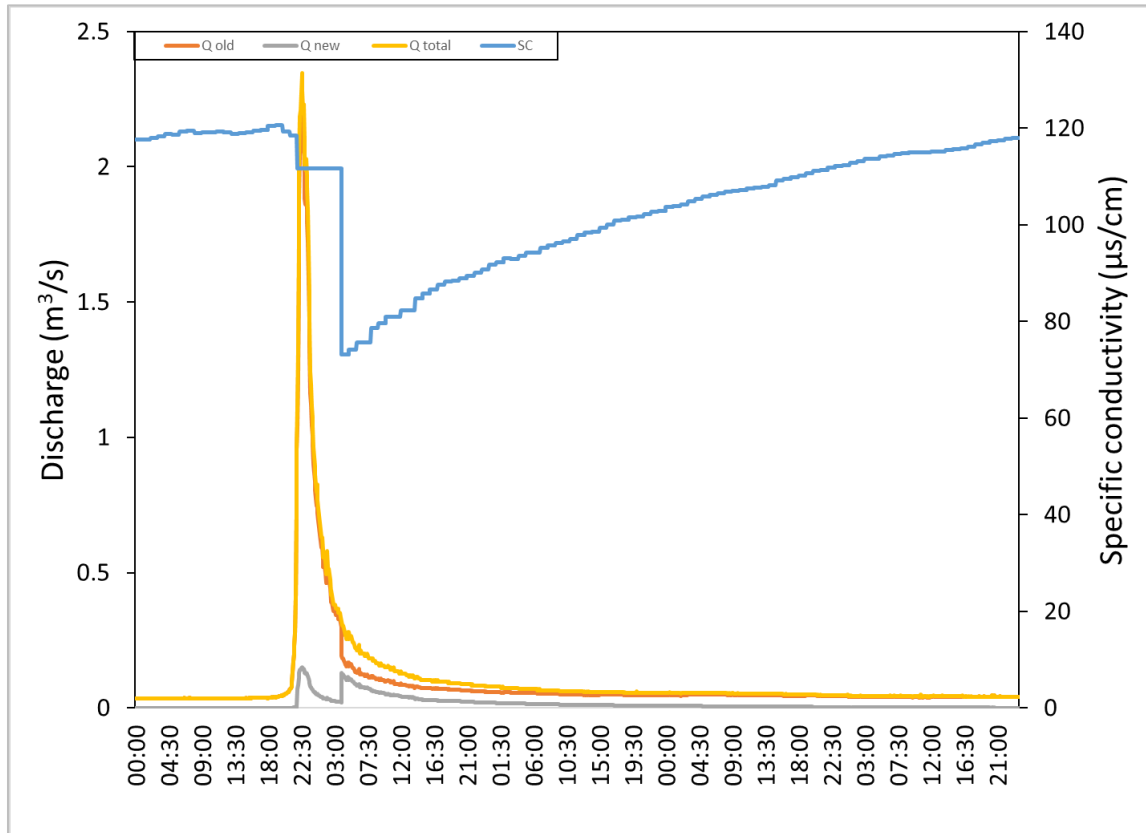


Figure 2.31 Post-restoration hydrograph separation for 12/06/2020 showing specific conductance ($\mu\text{s}/\text{cm}$) and discharge (m^3/s) components. This figure provides an example of a stream in which a major contributing factor to runoff is inferred to be from the old water component.

2.4.4 (RQ2.4) Does the seasonality of near-stream shallow groundwater change between the pre- and post-restoration stream?

2.4.4.1 D1 – Developed sub-watershed

Data from time series decomposition of D1R well water levels can be seen in Figure 2.32. Observed data for the D1R well indicate an increase in water level over the 2018 restoration period. Consistent with the visually evident increase in D1R groundwater levels, trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period. That observed increase in water level continued through the end of the monitoring period (Section 2.4.1). After detrending for

examination of the seasonal cycle, a Mann-Whitney test showed that there was not a statistically significant difference ($W = 288, p = 1$) between the D1R pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data, meaning that the seasonal cycling did not change significantly pre- vs. post-restoration. The median for the pre-restoration period was 0.003 compared to 0.003 for the post-restoration period. Random decomposed data showed a slight increase in randomness in the post-restoration period (Figure 2.32) indicating that the groundwater within the watershed may be more connected to the surrounding ecosystem leading to more sudden changes and may indicate that the D1R transect has not yet stabilized from restoration impacts.

Results of decomposition of D1U well water levels can be seen in Figure 2.33. Observed data for the D1U well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time in the during-restoration period. That observed increase in water level continued through the during-restoration period with the post-restoration periods exhibiting higher water levels than the pre-restoration period. A Mann-Whitney test showed that there was not a statistically significant difference ($W = 288, p = 1$) between the D1U pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The median for the pre-restoration period was 0.13 compared to 0.13 for the post-restoration period. Random decomposed data showed a decrease in randomness in the during-restoration period followed by a return to nearly pre-restoration randomness in the post-restoration period (Figure 2.33) indicating that the groundwater level within the D1U well may be stabilized in the post-restoration period.

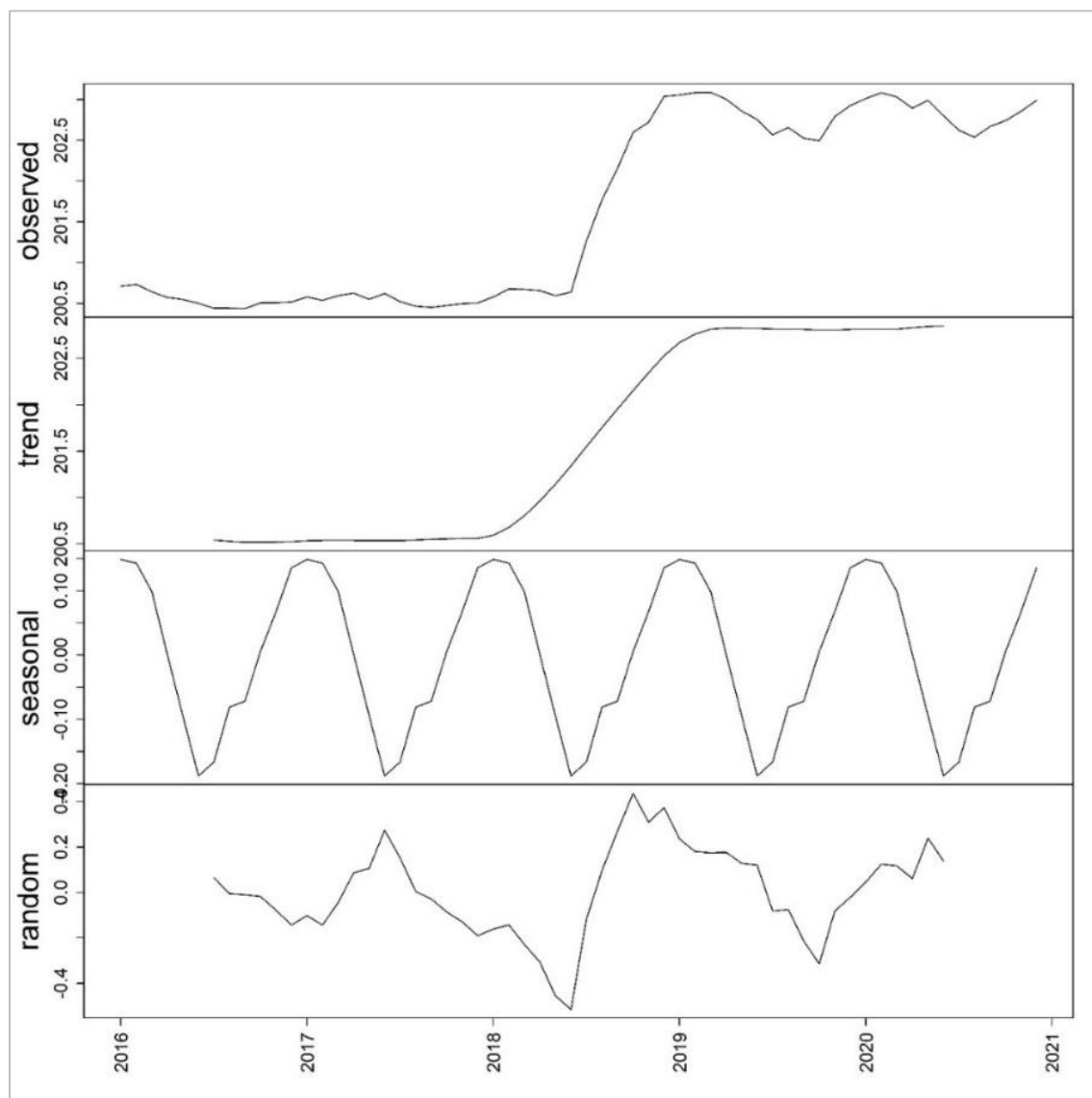


Figure 2.32 Groundwater level (m) decomposition from time series analysis for the riparian well (D1R). Data represent the sampling period from January 2016 – March 2021.

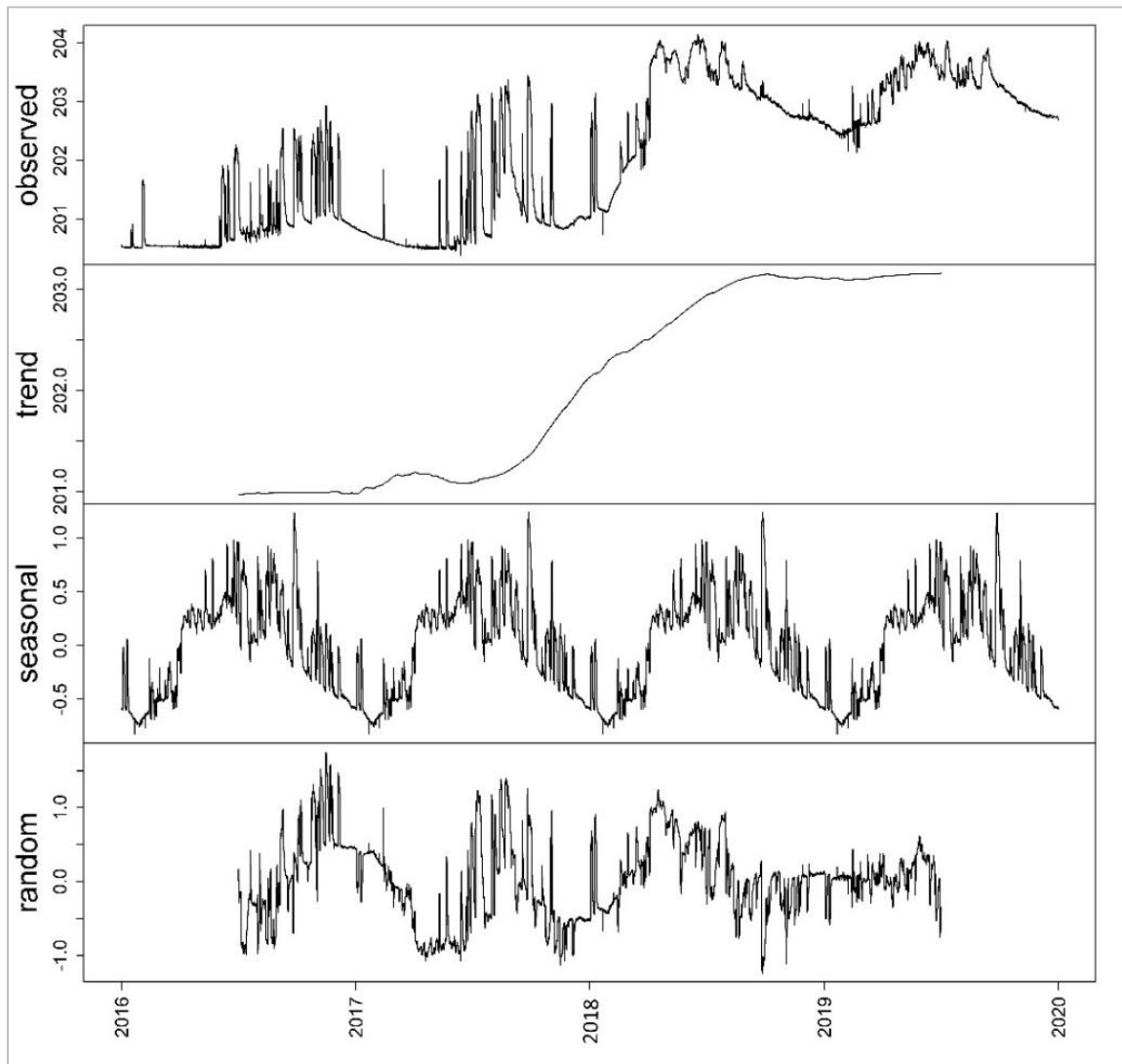


Figure 2.33 Groundwater level (m) decomposition from time series analysis for the upland well (D1U). Data represent the sampling period from September 2016 – March 2021.

2.4.4.2 P1 – Pond controlled/ park developed sub-watershed

Data from decomposition of P1R well water levels can be seen in Figure 2.34.

Observed data for the P1R well indicate an increase in water level after the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period. That observed increase in water level continued through to the end of the monitoring period. A Mann-Whitney test showed that there was not a statistically significant difference ($W = 288, p = 1$) between the P1R pre-restoration

decomposed seasonal data compared to the post-restoration decomposed seasonal data.

The median for the pre-restoration period was -0.07 compared to -0.07 for the post-restoration period. Random decomposed data showed a slight decrease in randomness in the post-restoration period (Figure 2.34) indicating that the groundwater level within the P1R well may have reached a new equilibrium in the post-restoration period.

Data from decomposition of P1U well water levels can be seen in Figure 2.35. Observed data for the P1U well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period with a return to near pre-restoration levels. A Mann-Whitney test showed that there was not a statistically significant difference ($W = 290$, $p = 0.98$) between the P1U pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The seasonal median for the pre-restoration period was 8.57×10^{-5} compared to 8.57×10^{-5} for the post-restoration period. Random decomposed data showed a slight decrease in randomness in the post-restoration period (Figure 2.35) indicating that the groundwater level within the P1U well may have reached a new equilibrium in the post-restoration period.

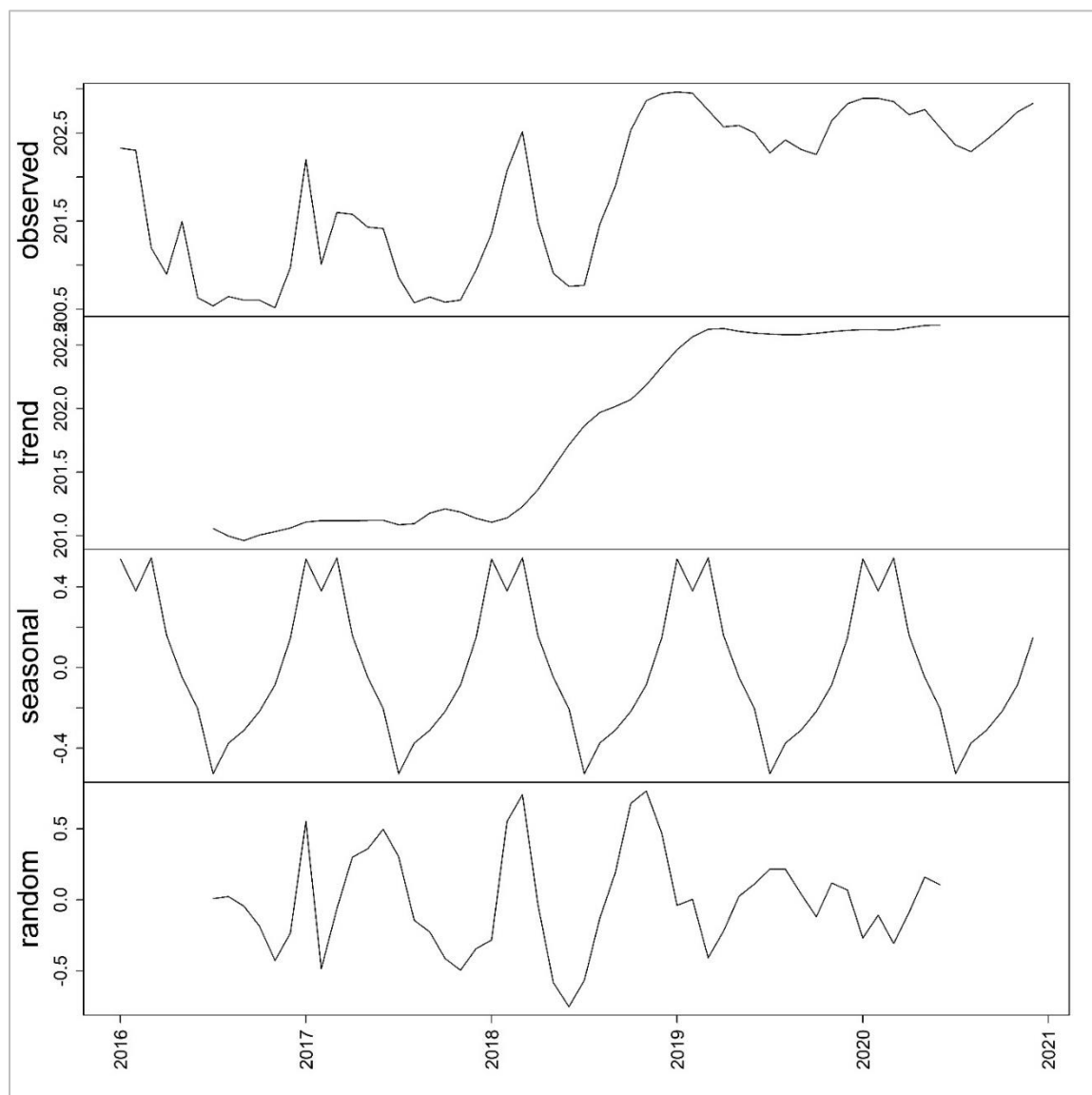


Figure 2.34 Groundwater level (m) decomposition from time series analysis for the riparian well (P1R). Data represent the sampling period from January 2016 – March 2021.

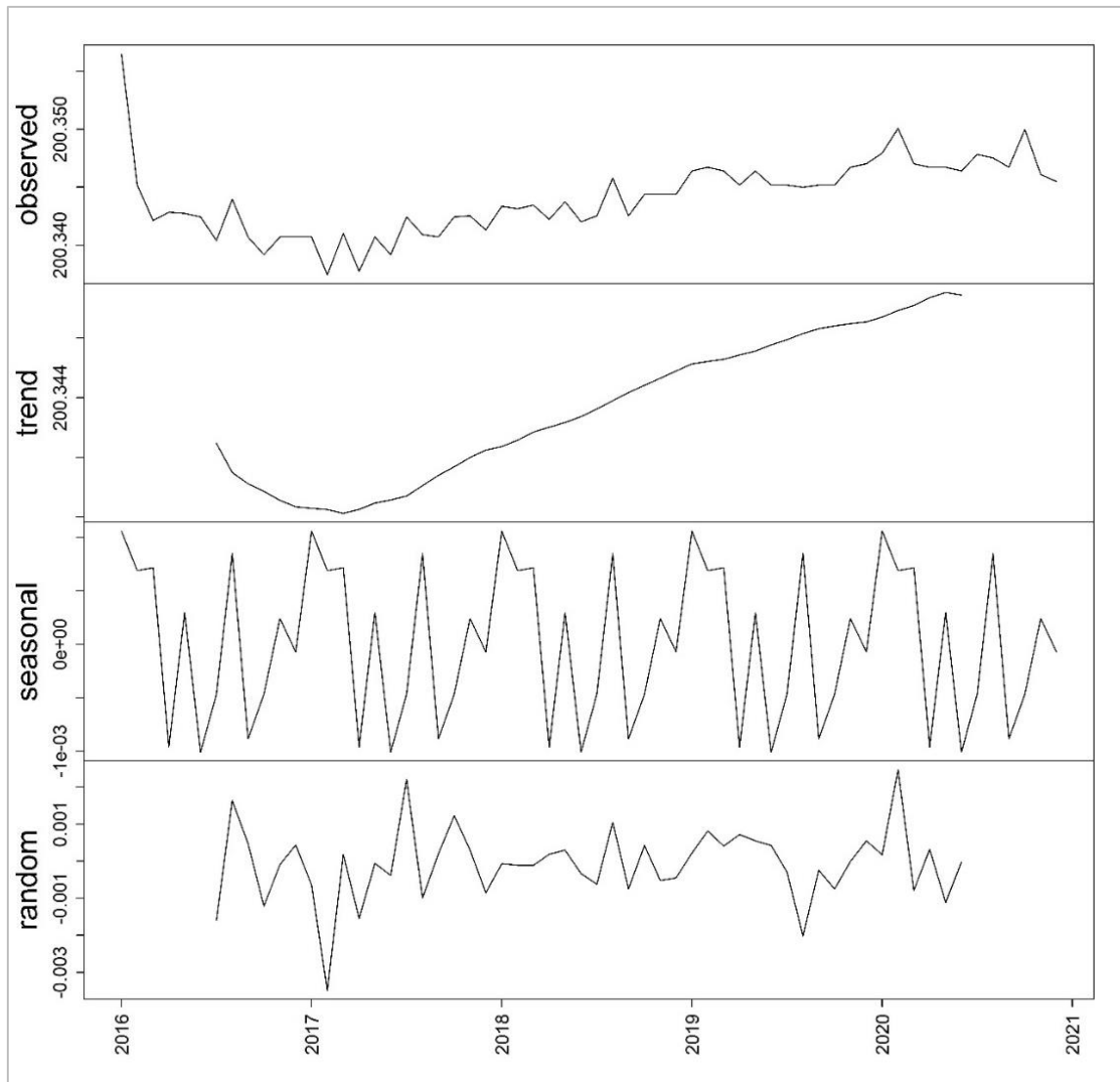


Figure 2.35 Groundwater level (m) decomposition from time series analysis for the upland well (P1U). Data represent the sampling period from September 2016 – March 2021.

2.4.4.3 C1 – Control sub-watershed

Data from decomposition of C1R well water levels can be seen in Figure 2.36.

Observed data for the C1R well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-

restoration to the post-restoration period. That observed increase in water level continued through to the end of the monitoring period. A Mann-Whitney test showed that there was not a statistically significant difference ($W = 288, p = 1$) between the C1R pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The median for the pre-restoration period was 0.02 compared to 0.02 for the post-restoration period. Random decomposed data showed a slight increase in randomness in the beginning of the post-restoration period with an overall decreasing trend in randomness (Figure 2.37) indicating that the groundwater level within the C1R well may be stabilized in the post-restoration period.

Data from decomposition of C1U well water levels can be seen in Figure 2.37. Observed data for the C1U well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period. That observed increase in water level continued through to the post-restoration monitoring period. A Mann-Whitney test showed that there was not a statistically significant difference ($W = 288, p = 1$) between the C1U pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The median for the pre-restoration period was 0.0002 compared to 0.0002 for the post-restoration period. Random decomposed data showed a slight decrease in randomness in the during-restoration period with a return to near pre-restoration randomness in the post-restoration period (Figure 2.37) indicating that the groundwater level within the C1U well may be stabilized in the post-restoration period.

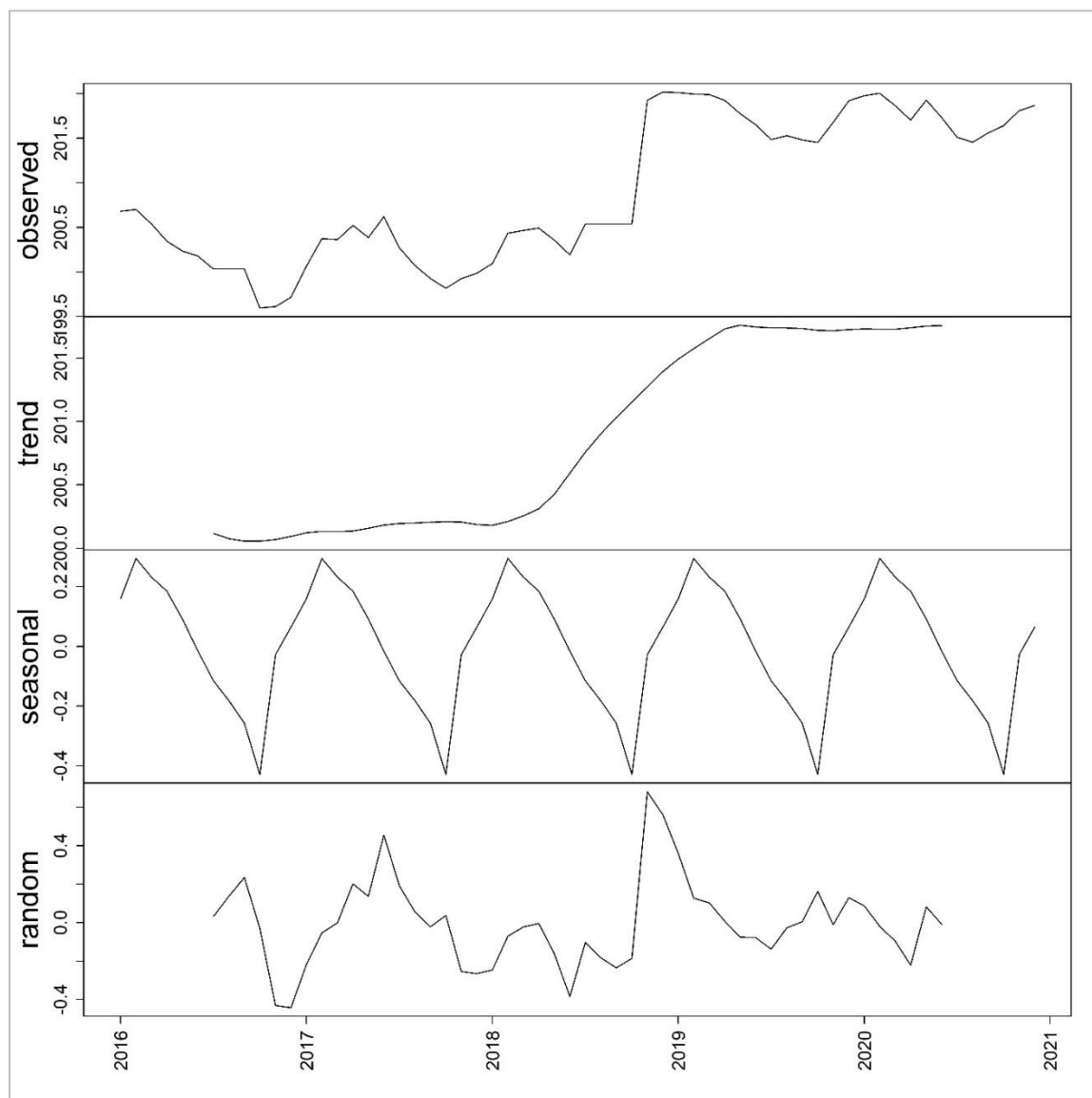


Figure 2.36 Groundwater level (m) decomposition from time series analysis for the riparian well (C1R). Data represent the sampling period from January 2016 – March 2021.

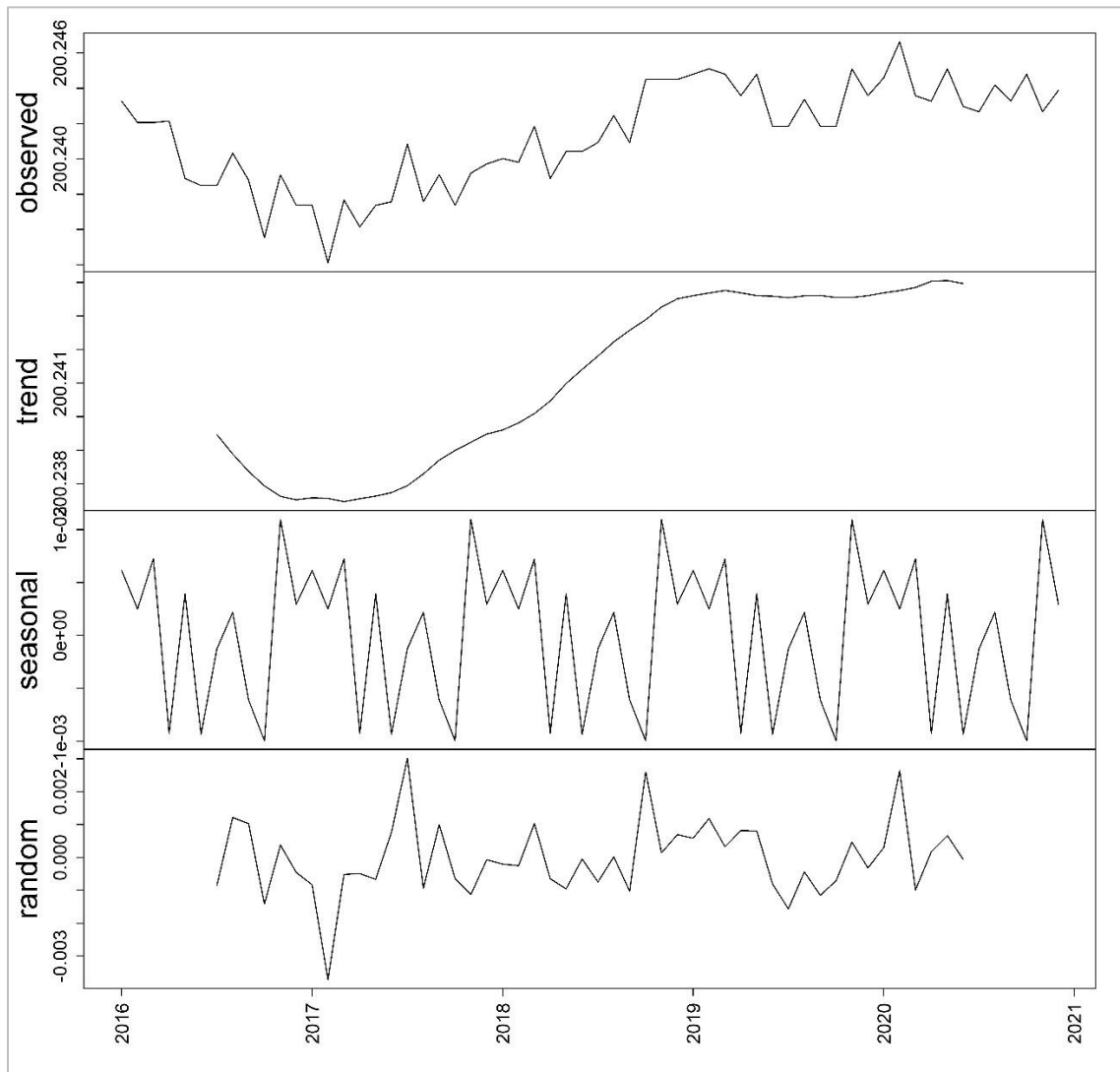


Figure 2.37 Groundwater level (m) decomposition from time series analysis for the upland well (C1U). Data represent the sampling period from September 2016 – March 2021.

2.4.4.4 A1 – Agricultural sub-watershed

Data from decomposition of A1R well water levels can be seen in Figure 2.38.

Observed data for the A1R well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period. That observed increase in water level continued through to the end of the monitoring period. A Mann-Whitney test showed that there was

not a statistically significant difference ($W = 288, p = 1$) between the A1R pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The median for the pre-restoration period was 0.03 compared to 0.03 for the post-restoration period. Random decomposed data showed a slight decrease in randomness in the post-restoration period (Figure 2.38) indicating that the groundwater level within the A1R well may have reached a new equilibrium in the post-restoration period.

Data from decomposition of A1U well water levels can be seen in Figure 2.39. Observed data for the A1U well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period with a dip in water levels in the during-construction period. That observed increase in water level continued through to the end of the monitoring period. A Mann-Whitney test showed that there was not a statistically significant difference ($W = 288, p = 1$) between the A1U pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The median for the pre-restoration period was 0.14 compared to 0.14 for the post-restoration period. Random decomposed data showed a slight increase in randomness in the post-restoration period (Figure 2.39) indicating that the groundwater within the watershed may be more connected to the surrounding ecosystem leading to more sudden changes and may indicated that the A1U transect has not yet stabilized from restoration impacts.

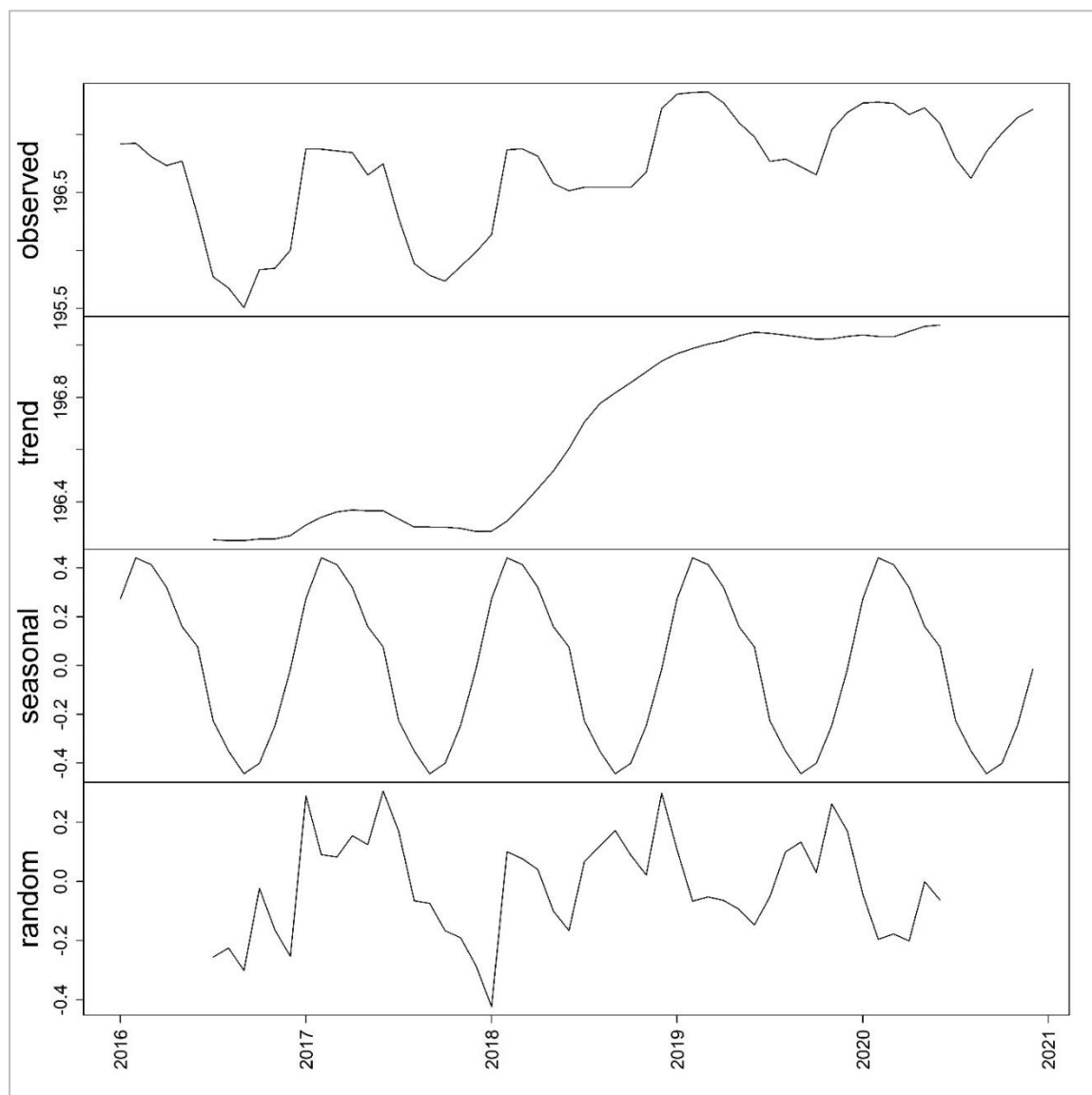


Figure 2.38 Groundwater level (m) decomposition from time series analysis for the riparian well (A1R). Data represent the sampling period from January 2016 – March 2021.

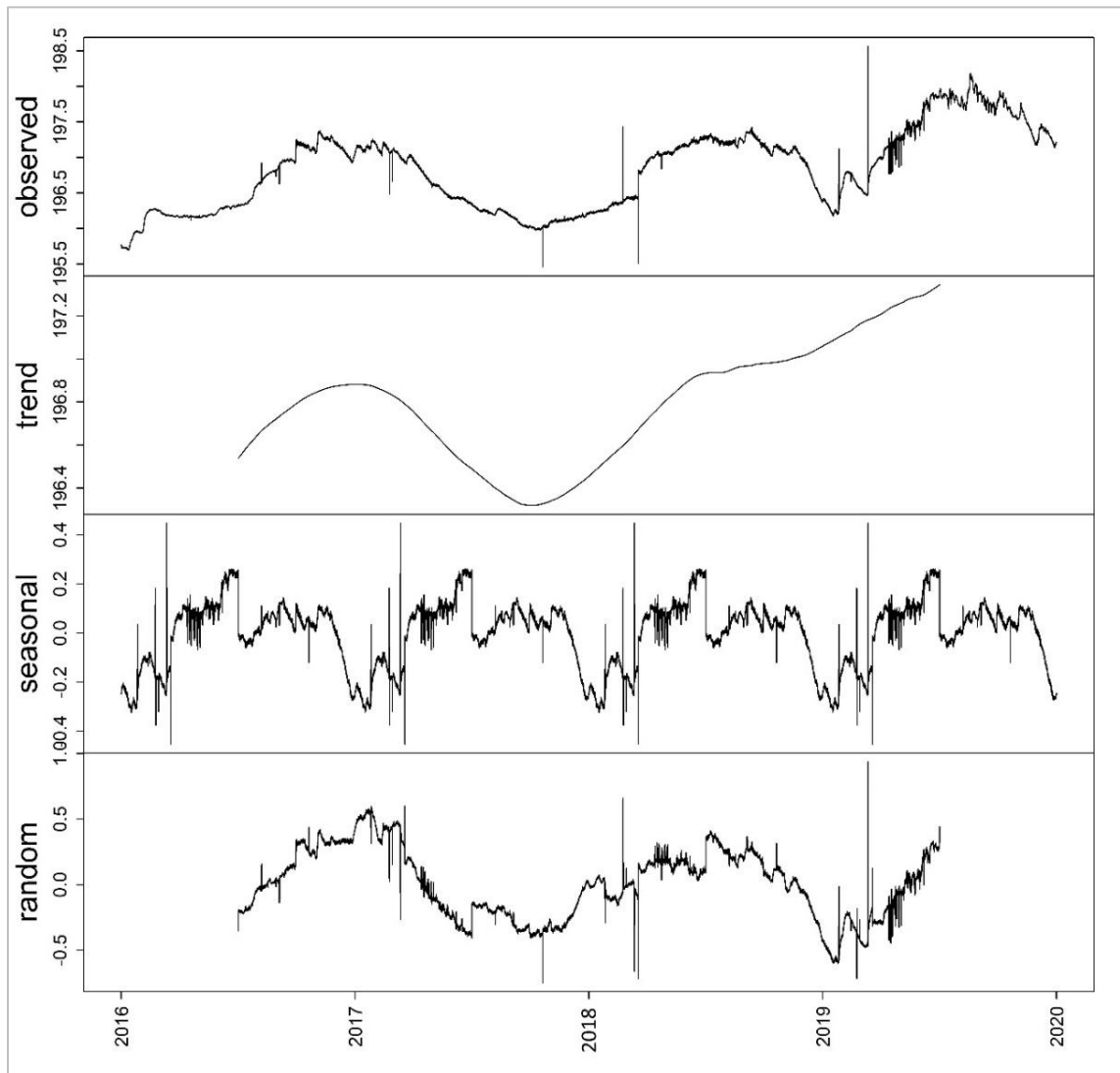


Figure 2.39 Groundwater level (m) decomposition from time series analysis for the upland well (A1U). Data represent the sampling period from September 2016 – March 2021.

2.4.4.5 R2 – Mainstem watershed

Data from decomposition of R2R well water levels can be seen in Figure 1.39.

Observed data for the R2R well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period. That observed increase in water level continued through to the end of the monitoring period. A Mann-Whitney test showed that there was

not a statistically significant difference ($W = 288, p = 1$) between the R2R pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The median for the pre-restoration period was -0.02 compared to -0.02 for the post-restoration period. Random decomposed data showed a slight increase in randomness at the beginning of the post-restoration period with an overall decrease in randomness towards the end of the monitoring period (Figure 2.40) indicating that the R2R well may have reached a new equilibrium in the post-restoration period.

Data from decomposition of R2U well water levels can be seen in Figure 2.41. Observed data for the R2U well indicate an increase in water level over the 2018 restoration period. Trend data indicate an increasing trend over time from the pre-restoration to the post-restoration period. That observed increase in water level continued through to the end of the monitoring period. A Mann-Whitney test showed that there was not a statistically significant difference ($W = 288, p = 1$) between the R2U pre-restoration decomposed seasonal data compared to the post-restoration decomposed seasonal data. The median for the pre-restoration period was 0.02 compared to 0.02 for the post-restoration period. Random decomposed data showed a slight decrease in randomness in the post-restoration period (Figure 1.39) indicating that the R2U well may have reached a new equilibrium in the post-restoration period.

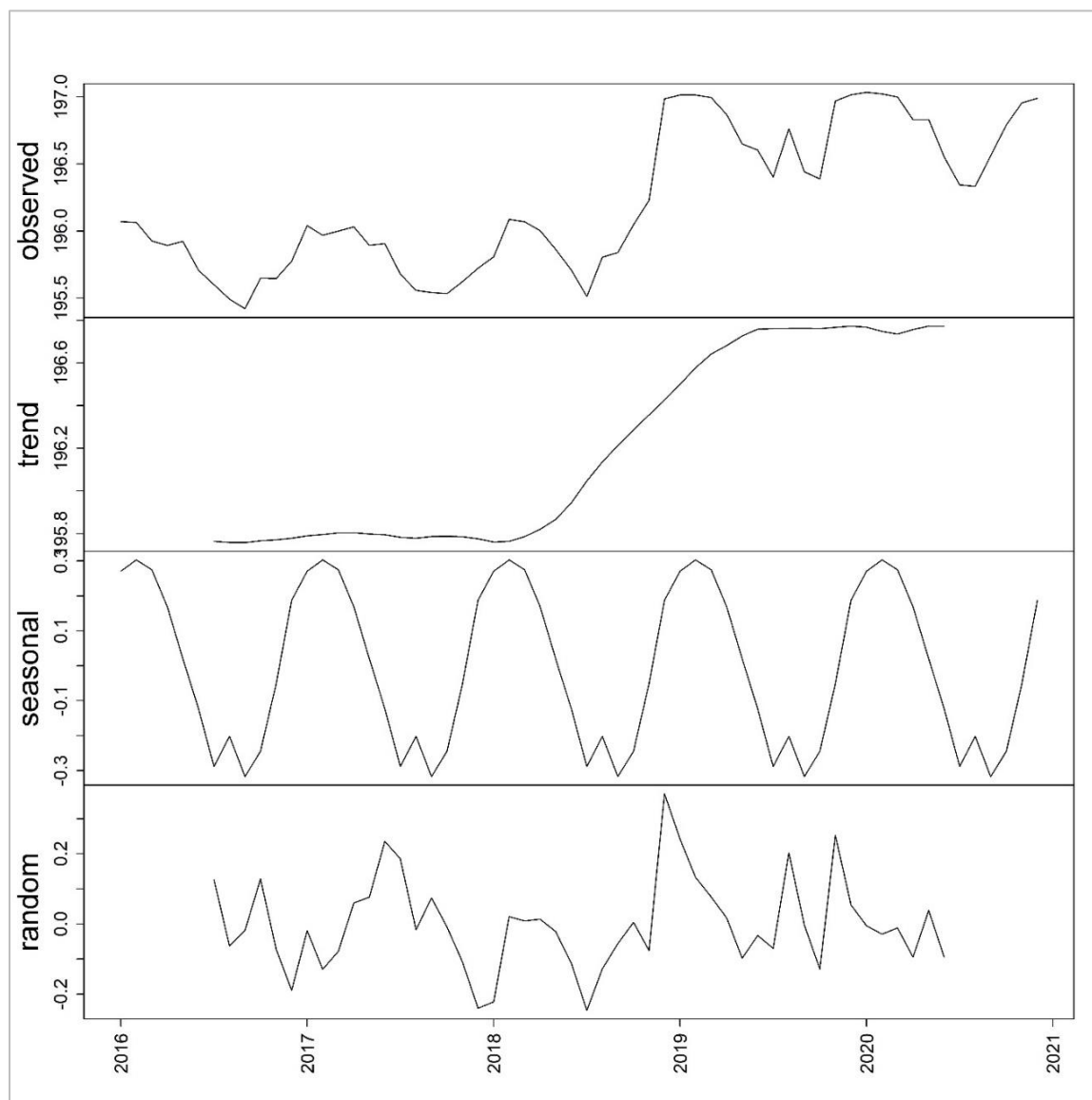


Figure 2.40 Groundwater level (m) decomposition from time series analysis for the riparian well (R2R). Data represent the sampling period from January 2016 – March 2021.

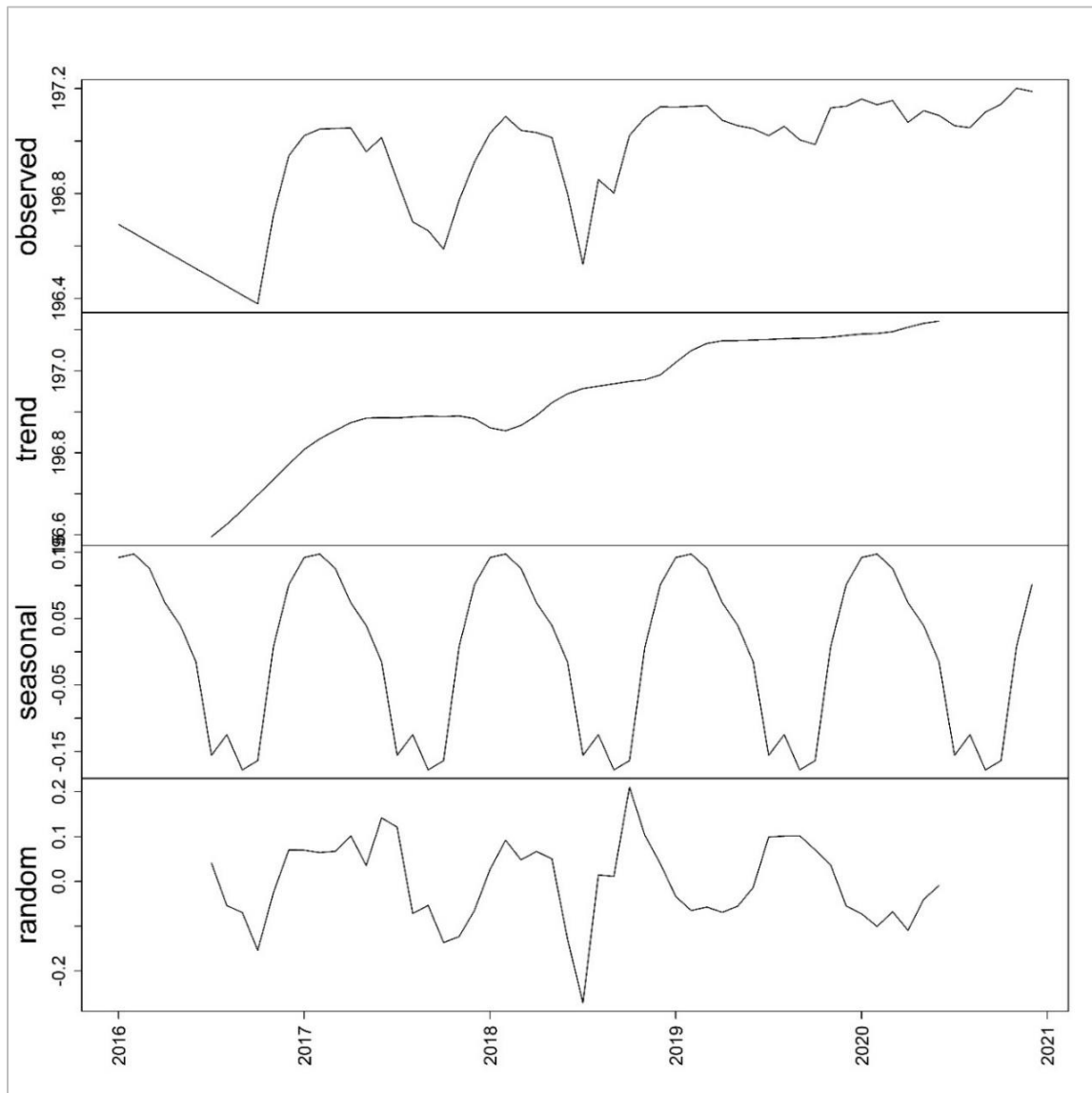


Figure 2.41 Groundwater level (m) decomposition from time series analysis for the upland well (R2U). Data represent the sampling period from September 2016 – March 2021.

2.4.4.6 Watershed changes in groundwater seasonality

Overall, changes in water levels throughout the watershed were not consistent across all transects. Trended change in water level showed increases in the post-restoration period in seven of 10 wells (Table 2.26). Three wells saw a neutral change in water level (Table 2.26). In all wells, the seasonality changes were non-significant (Table

2.26) indicating that the season cycling of groundwater within the watershed was not impacted by restoration processes. Findings are consistent with the hypotheses that seasonality would not change (RQ2.5). However, D1U, P1U, and C1U findings were not consistent with the hypothesis that seasonal cycling would continue at a higher elevation (RQ2.5).

*Table 2.26 Trended changes water level (m) and seasonality from the pre-restoration to the post-restoration period. Significance is indicated by *.*

Site	Trended change in water level from pre to post	Seasonality Statistical Significance
D1U	Neutral	/
D1R	Increased	/
P1U	Neutral	/
P1R	Increased	/
C1U	Neutral	/
C1R	Increased	/
A1U	Increased	/
A1R	Increased	/
R2U	Increased	/
R2R	Increased	/

2.5 Discussion

2.5.1 RQ2.1

Restoration was successful at changing the water levels throughout the watershed. Here, raising the stream channels and stream levels to a higher elevation effectively increased groundwater storage at both the riparian and upland wells allowing for a more robust saturated zone. Restoration increased the water levels at riparian and upland wells indicating that the watershed stores more shallow groundwater year round than in the pre-restoration period (Rose & Peters, 2001; Hardison et al., 2009). This increased groundwater throughout the watershed has the potential to increase hyporheic exchange,

increase water retention, increase baseflow consistency and constancy reducing the chances of stream drying and maintaining water quality, and ultimately lead to a watershed in which biogeochemical transformations can occur and ecological needs are met (Winter et al., 1998; Rose & Peters, 2001; Hardison et al., 2009; Tonina & Buffington, 2009; Healy & Scanlon, 2010; Hester & Gooseff, 2010; Hiscock & Bense, 2014; Hester et al., 2017; Zimmer & McGlynn, 2017; Hunt et al., 2018).

Mann-Whitney test analyses of groundwater levels were found to be significantly different, indicating that groundwater storage changed significantly between the pre- and post-restoration periods and indicating a transient increase in water stored in the watershed due to restoration. Within the Reedy Creek watershed, restoration is considered successful as it was able to increase GW-SW interactions, as groundwater and surface water sources are more connected, within the watershed which may provide increased quality of water. Furthermore, restoration was able to increase groundwater storage within the watershed which may lead to beneficial wetland-like functions as the water table is maintained closer to the land surface.

2.5.2 RQ2.2

The current study found that hydraulic gradients responded differently at transects throughout the watershed. Restoration led to a steeper hydraulic gradient in three of five overall study transects (D1, P1, and A1) and led to a decrease at two overall study transects (C1 and R2). Here, restoration at D1, P1, and A1 transects led to increased hydraulic gradients which may be due to changes in well locations (A1), stream location post-restoration (P1 affected by D1), or study design with transects not being completely perpendicular (D1, P1, and A1) leading to results indicating that one location along the

transect was higher than in actuality. Restoration of the stream at the R2 and C1 transects led to decreased hydraulic gradients which may be attributed to the location of the study sites and the relocation of both wells at the C1 transect. Here, the R2 upland well was placed in a wetland where groundwater elevation was above the ground surface for the majority of the year and once the groundwater increased at the R2 riparian well, the gradient decreased. Restoration increased groundwater levels within the riparian and upland well in conjunction with raising streambeds and surface water elevations leading to gentler hydraulic gradients within the pond influenced watershed and increased potential for long-term water storage. This paradox, where hydraulic gradients increased in upper stream reaches instead of decreasing may lead to more groundwater discharge in the stream – increasing baseflow – instead of enhancing hyporheic exchange.

Maintaining sufficient baseflow into summer and fall should benefit local flora and fauna by ensuring the availability of water during the drought-prone season (Hunt et al., 2018; Jansson et al., 2007; Kath et al., 2018). Additionally, gentler hydraulic gradients could lead to slower movement of water through the soil which can enhance nutrient processing (Wallis et al., 1981; Grimm et al., 2003; Jansson et al., 2007; Stein et al., 2010; Durand et al., 2011; Zarnetske et al., 2011b; Marmonier et al., 2012; Griebler & Avramov, 2015; Johnson et al., 2016; Yabusaki et al., 2017). Overall, Reedy streams exhibited both increased and decreased gradients with some stream reaches aiding baseflow and others enhancing hyporheic exchange.

Here, stream and groundwater levels rose throughout the watershed and, therefore, the change in hydraulic gradients was directly related to which (e.g.: stream or groundwater levels) rose more. In certain transects (D1, P1, and A1), data indicate that

the groundwater levels rose more than the corresponding stream levels, leading to an overall increase in gradients from the upland to the stream. In other transects (C1 and R2), data indicate a larger increase in stream level than in corresponding groundwater levels leading to an overall decrease in gradients from the upland to stream. However, at some sites (A1) large gaps in data pre-restoration led to uncertainties in changes in gradients.

Mann-Whitney test results for hydraulic gradients suggest that gradients did change from their pre-restoration levels at all sites except R2. Increases in hydraulic gradients occurred in three of five study transects (D1, C1, and A1) and these changes were statistically significant indicating that groundwater levels and hydraulic gradients differ between the pre- and post-restoration periods. Here, restoration was not successful at reducing gradients throughout the watershed, indicating that there were steeper hydraulic gradients. However, at these transects within the watershed, increased gradients could lead to increases in groundwater-driven baseflow – maintaining stream ecological function through drier months. Overall, restoration altered hydraulic gradients leading to increased groundwater-driven baseflow and increasing the potential for hydraulic gradients at various locations within the watershed.

2.5.3 RQ2.3

Hydrograph separation results indicate that old water endmember percentages decreased from pre-restoration to post-restoration. Post-restoration, there was an increase in new water endmember percentages indicating that precipitation contributed more to stormflows in the post-restoration period. Here, restoration increased groundwater levels

throughout the watershed, which, in turn, may have increased the potential of saturation-excess overland flow.

Hydrograph separation results indicate that baseflow amounts increased from pre-restoration levels to post-restoration levels. Here, increase in baseflow amounts may be attributed to restoration induced changes in the watershed. For instance, raising stream channel elevations allowed for an increase in groundwater storage which, in turn, increased the potential for additional baseflow discharge throughout the year. Moreover, changes in hydraulic gradients driven by the changes in stream and groundwater levels subsequently added to stream baseflow.

In both the pre- and post-restoration periods, the hydrograph separation was most successful when separating the end members of low intensity short-duration storms and least successful with long intensity long-duration storms (Table 2.27). Moreover, the hydrograph separation method was more successful in the post-restoration period than in the pre-restoration period with low percentages of negative cases for the new water endmember post-restoration (Table 2.27). Results from all hydrograph separation attempts implied that once separated, the individual hydrograph component new water endmembers were less than zero, which is non-meaningful according to a two-component mixing model. Here, results indicated there was a negative fraction for the new water endmember in 7-50 percent of comparisons (Table 2.27) indicating that the old water endmember was contributing more than 100% of flow. A possible explanation of negative endmembers includes having specific conductivity of water from each endmember that may not be different enough to cause a significant change in the specific

conductivity of stream flow. Without having this change, more water may be attributed to the old water endmember than the new water endmember.

Results show that during the 4/22/2016 storm, the old water endmember amounts was higher than actual recorded total flow (Figure 2.24). One possible explanation of why the hydrograph separation may have been unsuccessful within the Reedy Creek watershed may be due to the SC not being sensitive enough to distinguish between the different endmembers. In other chemical hydrograph separation studies, SC of precipitation was significantly different than the SC of the baseflow stream and stormflows (Pelletier & Andréassian, 2020), such as was seen in research by Pellerin et al. (2008) where the addition of road salts led to significantly different SC in stormflows.

Table 2.27 Percentage of hydrograph separation results that were negative (i.e.: indicating that one endmember was responsible for more than 100% of total flow) for each endmember

	Date	Old Water	New Water	Storm category
Pre-restoration	4/22/16	0	50	low intensity long-duration
	6/03/16	0	24	high intensity short-duration
	6/30/16	0	19	low intensity short-duration
	1/13/18	0	46	high intensity long-duration
Post-restoration	11/25/19	0	7	low intensity short-duration
	12/11/19	0	18	low intensity long-duration
	1/25/20	0	15	high intensity long-duration
	12/06/20	0	11	high intensity short-duration

Due to uncertainties surrounding the results (i.e.: negative modeled flow rates for new water endmembers) it cannot be said for certain that restoration led to an increase in baseflow amounts during the post-restoration period, if baseflow changes were due solely to increases in precipitation via increased throughfall, or if baseflow changes were due to reduced evapotranspiration caused by clear-cutting. However, if SC had shown a significant difference, results would indicate that baseflow was different between the pre- and post-restoration periods. An increase in baseflow amounts would have provided evidence that restoration was successful at increasing groundwater storage and decreasing the flashiness characteristic of urban streams. If correlation analysis was significant, it would be expected that restoration had maintained the GW-SW connections necessary for baseflow. Restoration would have been considered successful if it was able to increase groundwater storage leading to increased baseflow amounts at the watershed outlet. However, it may be assumed that baseflow amounts were increased in regions where hydraulic gradients increased in research question 2.2. This information would be valuable to restoration practitioners as baseflow is necessary for the continuation of both watershed instream ecological processes and the maintenance of downstream ecological functioning and processes.

Result from the flashiness analysis indicated that there was no significant change in baseflow contributions over time. It is important to note that this study used daily average Q while Nathan & McMahon (1990) used instantaneous Q . Using daily average Q may have led to results being skewed by any high-flow events or low-flow periods. Here, baseflow remained the same over time indicating that the stream did not become significantly more baseflow-dominated pre- vs. post-restoration. This may be attributed

to an increase in precipitation in the post-restoration period and is indicative that restoration was successful at reducing flashiness as the streams moved more stormflows without increasing the flashiness index.

2.5.4 RQ2.4

Seasonal cycling of groundwater was not changed (i.e.: post-restoration groundwater exhibited the same seasonal cycling pattern as pre-restoration). Restoration led to an increase in seasonal groundwater levels at all 10 wells indicating that the watershed has the capacity to store more groundwater than in the pre-restoration period and that cycling has continued seasonally but at a higher elevation. Increases and a return to neutral levels for watershed groundwater did not have an impact on seasonal cycling and may have provided the water needed to support robust riparian vegetation (Zimmer & McGlynn, 2017).

Mann-Whitney test results for seasonality indicate that seasonal water table cycling did not change from pre-restoration patterns. In all cases, these patterns were not statistically different between seasons in the pre- and post-restoration periods. Here, restoration was successful at increasing water levels while not significantly impacting the seasonal cycling of the water table. Overall, changes in groundwater indicate a more stable water table in the post-restoration period.

2.6 Conclusion

Groundwater is a vitally important component of the hydrologic cycle and makes up nearly 30% of all freshwater on earth where it works to meet hydrologic and ecologic needs (Hornberger et al., 1998; Winter et al., 1998; Healy & Scanlon, 2010; Famiglietti, 2014; Holden, 2014; Griebler & Avramov, 2015). Currently groundwater is threatened

from unsustainable use, climate change, stream channel incision, altered hydraulic regimes, and increasing demands from population growth which has strained groundwater supplies and led to increasing conflicts over water resources (Healy & Scanlon, 2010; Famiglietti, 2014; Holden, 2014). This combined alteration of groundwater level and hydrologic regimes happens globally and calls for effective management plans and practices (Healy & Scanlon, 2010).

Post-restoration, trends in water levels throughout the watershed indicate that there was an increase in groundwater levels, an increase in hydraulic gradients at three of five transects, baseflow increased, and seasonal water table cycling remained the same although at a higher elevation at most transects. These results indicate that the whole-watershed restoration approach used within the Reedy Creek watershed was successful at altering the hydrological regimes of the watershed in a way that provides support for ecological communities and enhances the biophysical processes needed for the ecosystem to thrive.

Here, planning for future restoration projects around these findings could have significant implications for water availability. For instance, if target restoration sites have deep groundwater due to heavily incised streams, restoration practitioners could use a similar restoration approach to increase groundwater levels and ultimately enhance the potential for ecosystem functioning. Having the water table closer to the land surface also works to increase the chance of flooding of the surrounding floodplain and acts as a type of hydrologic retention. Additionally, the increase in water levels throughout the watershed, while beneficial in the study watershed, could have detrimental impacts on floodplain trees and man-made structures – especially those with basements – as an

increase in groundwater could suffocate the roots of trees and flood structures. Here, restoration practitioners and watershed managers should understand that restoration may have effects not thought of or planned for but nonetheless could have far-reaching effects.

Understanding how hydraulic gradients change after restoration may help restoration practitioners and watershed managers better prepare for future climate shifts as hydraulic gradients are related to groundwater storage and retention. Increasing groundwater levels and having gentler hydraulic gradients has important implications – especially in the drier seasons with low gradients allowing for a steady discharge of groundwater. In areas where restoration increases the groundwater levels and hydraulic gradients become steeper within the watershed, groundwater discharge may be increased (e.g.: D1, P1, and A1). Here, in two of five transects, restoration was considered successful as it was able to increase groundwater storage and stream water level elevation thus reducing hydraulic gradients (i.e.: gradients became more gentle) within the watershed over time. It is important to note that while gradients became more steep in three of five transects, these changes may be temporary as the watershed seeks to find an equilibrium with newly transformed streams. It is also important to note that the morphological differences in streams and potential changes in soil material (e.g.: material brought in to backfill channels) post-restoration may have altered surface water and groundwater flow paths leading to toward-stream gradients in some areas and away-from-stream gradients in other areas as hyporheic exchange happens at multiple timescales, spatial scales, and directions (Hester & Gooseff, 2010). Overall, alterations in gradient direction and steepness may be evidence of streams switching from gaining to losing and vice-versa – but generally tend to be gaining both pre- and post-restoration.

Future research within the watershed should focus on investigating the mechanisms for gradient alterations which will allow for more insight into restoration-induced watershed-scale hydrologic retention.

Understanding the connection between precipitation and groundwater is essential for watershed management. Here, groundwater can act as a reservoir buffering intense stormflows, leading to overall less flashy streams, and increasing groundwater storage to prolong baseflow into dry season months. Moreover, recharge rates can indicate the infiltration capacity of the watershed. As this study focused on precipitation and groundwater level changes within the same months, future research could investigate if there is a time lag between precipitation and changes in groundwater levels.

Understanding the individual contributions of old and new water to streams is important for watershed planning and management. To do this, future hydrograph separations are needed to parse out the individual contributions. Within Piedmont streams, methods that use SC may not be sensitive enough to differentiate between the endmember waters. For instance, for SC hydrograph separation to work, groundwater, surface water, and precipitation need to have significantly different concentrations or results will be skewed. A limitation of this study is that the SC data for each rain event was collected from a location outside the study watershed and, therefore, future research should plan to collect precipitation within the watershed of study to ensure accurate SC concentrations. Moreover, future research should focus on determining which chemical method is most appropriate for urban forested streams within the Piedmont of North Carolina where SC concentrations may not be effective.

For restoration practitioners, knowledge of how whole-watershed stream restoration impacts water table dynamics is important. For instance, these findings can help restoration practitioners and watershed managers understand where, when, and how to utilize the land to ensure groundwater is used in a sustainable manner. In areas with well-defined wet and dry periods, managing water sources and sinks throughout the wet period may provide the prolonged baseflow to streams within the dry season. As climate change is set to alter the hydrological cycle in both quantity (e.g.: precipitation) and timing (e.g.: seasons), it is imperative that watershed managers understand seasonal groundwater cycling to provide management plans that take into account seasonal highs and seasonal lows ensuring that ecosystem and human needs are met – both within the restored watershed itself and downstream.

3 INFLUENCE OF STREAM RESTORATION ON NITROGEN IN AN URBAN FORESTED WATERSHED

3.1 Introduction

Nitrogen (N) pollution is a substantial component of water quality degradation worldwide (Grimm et al., 2003; Drever, 2005; Kaushal et al., 2014; Zhou et al., 2017; Journey et al., 2018; Lavelle et al., 2019) and is the most human-modified of all global biogeochemical cycles (Schlesinger & Bernhardt, 2013). Anthropogenic changes to ecosystems have resulted in increased atmospheric, where most nitrogen is derived, and terrestrial N which have led to harmful algal blooms, degradation of habitat, and altered aquatic food webs (Kaushal et al., 2014; Journey et al., 2018). Nitrogen loads in streams have been artificially elevated due to human caused environmental stressors such as the introduction of wastewater, increased impervious cover, land-use changes (e.g.: agriculture), and the burning of fossil fuels (Driscoll et al., 2003; Kaushal et al., 2014).

Nitrogen is needed to maintain ecosystem functioning as it is a key nutrient to support aquatic primary production. As such, N can be a key limiting factor, along with phosphorous, controlling the rate of primary production in both aquatic and terrestrial ecosystems (Vitousek et al., 2010; Schlesinger & Bernhardt, 2013). According to Schlesinger & Bernhardt (2013), N is a fundamental element of the enzymes that are responsible for biochemical reactions that reduce and oxidize carbon which is needed for nutrient processing. Roberts & Mulholland (2007) found that increases in primary production (i.e.: the creation of organic matter/biomass) within forested streams in eastern Tennessee led to a decrease in nitrate concentrations. The creation of biomass instream is temporary as aquatic flora and fauna complete their lifecycles at different

timescales and once lifecycles are complete and biomass starts to degrade, nitrogen may be released back into the stream. Moreover, this temporary storage of N that is assimilated into biomass is responsible for 84% of nitrate removal from streams with denitrification accounting for the other 16% (Cardinale, 2011). However, when in excess, N can be detrimental to freshwater ecosystems. For instance, when there is an overabundance of N, there may be an excess of aquatic plants and algae biomass causing reduced dissolved oxygen levels (i.e.: plant uptake via respiration) and shading of the water column. Excess N in water can pose health risks for both aquatic species (e.g.: eutrophication) and humans (e.g.: blue baby disease) (Drever, 2005). Moreover, in forested ecosystems, N concentrations may be regulated by the growing versus dormant season such as a decrease in inorganic N during the growing season as seen by Arheimer et al. (1996). Here, decreases in inorganic N are driven by biotic uptake in the form of flora primary productivity. Similarly, Bernhardt & Likens (2004) found that during the dormant season, forested streams within the Hubbard Brook Experimental Forest had higher concentrations of N than compared to the growing season.

The effects of anthropogenic stressors are most evident in urban environments where streams exhibit signs of the urban stream syndrome (USS), which include flashy hydrographs, excess nutrient loading, and altered stream channel morphology and hydrology (Walsh et al., 2005). These characteristics effectively decrease the potential of streams to naturally transform N. Within streams exhibiting the USS, human intervention may be needed to promote N processing.

In stream ecosystems exhibiting the USS, nutrient processing occurs in a similar manner as natural streams. Here, as dissolved nutrients are accumulated by organisms,

converted to organic forms, and returned to the water as inorganic forms after organisms and materials degrade (Schlesinger & Bernhardt, 2013). These inorganic forms are transported downstream where they have two fates: 1) to be accumulated once again by organisms which begin the process again (e.g.: nutrient spiraling) or, 2) to have the organisms convert them to a form that may be released to the atmosphere (Schlesinger & Bernhardt, 2013). However, one or more components of this cycle may be altered by anthropogenic induced ecosystem degradation which may be alleviated through physical restoration. Here, physical changes (e.g.: the creation of riffle-pool sequences) within the watershed induced by restoration may lead, inadvertently, to ecological restoration (e.g.: creation of habitats for aquatic species to colonize). An essential challenge for sustainable ecosystems is restoring stream ecosystems in a way such that all stages of the N cycle are functioning.

To begin to address nutrient cycling within streams exhibiting characteristics of the USS, restoration projects should incorporate strategies (e.g.: reconnect floodplains with channels, reduce channelization, and alter channel features (e.g.: addition of rock or log cross vanes, back eddies) which create hotspots (e.g.: areas of hyporheic exchange and instream pools) for retention, uptake, and removal (e.g.: denitrification and biological uptake) which reduce aquatic N concentrations. In general, as N additions to streams continue to increase worldwide, researchers must examine the benefits of various restoration approaches to determine the best strategies for remedying nutrient pollution. This study aims to determine whole-watershed stream restoration's ability to alter N concentration and flux in the 6 km² Reedy Creek watershed.

3.1.1 Nitrogen Cycling in Small Forest Streams

Within forested stream ecosystems, N is controlled by a variety of processes, mechanisms, and sources. Nitrogen within forested streams can be derived from a/biotic fixation and the recycling and decomposition of dead materials (Schlesinger & Bernhardt, 2013). Nitrogen within these ecosystems is dominated by organic (e.g.: proteins & urea) and inorganic forms (e.g.: NO_3^- & NH_4^+) and in an unimpacted system, N should be efficiently transformed from one form to another. The transformation of nitrogen within stream ecosystems is dominated by microorganisms through oxidation and reduction (Reisinger et al., 2016; Xia et al., 2018). Nitrogen transformation is a cyclical process that is controlled by the availability of carbon, nitrogen, temperature, oxygen concentration, and residence times (Mitchell, 2001; Xia et al., 2018). Oxygen concentrations are important as the amount of oxygen present dictates which processes occurs (Trimmer et al., 2012) with denitrification occurring in the absence of oxygen and nitrification occurring in the presence of oxygen. The nitrogen cycle includes the processes of denitrification (i.e.: the transformation of nitrate to gaseous N) and nitrification (i.e.: the transformation of ammonium to nitrate) (Figure 3.1)

Residence time plays an important role in nitrogen retention and transformation as longer residence time also provides more opportunities for nitrogen removal (Grimm et al., 2003; Durand et al., 2011; Zarnetske et al., 2011b; Xia et al., 2018). Moreover, residence time is intimately connected to sediment type and size and the presence/absence of various other stream features (e.g.: weirs, log vanes, pools, surface water – floodplain connections). Therefore, the presence or absence of these features is important in understanding net de/nitrification. With stream restoration, the addition of

stream features such as log cross vanes may allow for hyporheic exchange (Crispell & Endreny, 2009) which can increase residence times leading to an increase in the potential of oxygen removal. This increase in residence time enhances the conditions needed to decrease oxygen levels and, ultimately, increase denitrification rates. Moreover, the presence of carbon, from essential riparian and floodplain vegetation, is essential for microorganisms as it fuels denitrification (Xia et al., 2018). Furthermore, temperature has been shown to regulate denitrification/nitrification with evidence suggesting that warmer temperatures create favorable conditions to increase biological activity of denitrifying bacteria leading to increasing denitrification (Zheng et al., 2016; Xia et al., 2018).

Throughout the N cycle, N takes on many forms from gaseous N (created via denitrification or deposited from the atmosphere) to ammonium (created via ammonification) to nitrate (created via nitrification or enters the stream via overland flow from agricultural/urban areas). The presence of each form is important for the growth and energy usage of various organisms. To begin, N fixation, is the process of biological accumulation by which N_2 gas is converted into ammonium by microorganisms (Trimmer et al., 2012). This process is important as it converts biologically unavailable gaseous N to biologically available N in the form of ammonium which is more easily taken up by organisms. In the Piedmont of North Carolina, cyanobacteria and several riparian plant species are nitrogen fixers including the common cattail (*Typha latifolia*) (Biesboer, 1984) and black locust (*Robinia pseudoacacia*) (Funk & Roach, 1961).

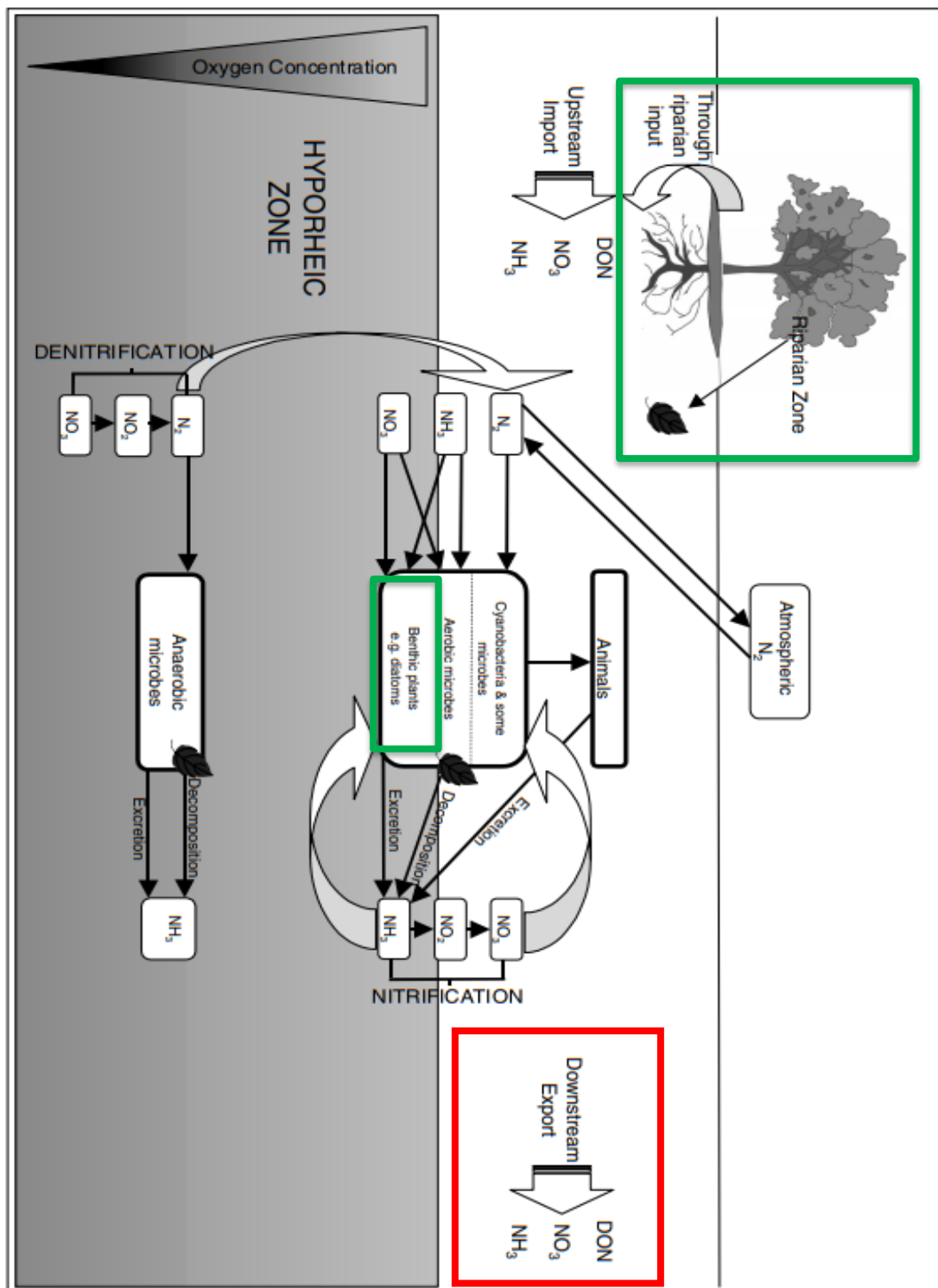


Figure 3.1 The freshwater nitrogen cycle (Adapted from Connolly & Pearson, 2005). Red box indicates focus of study which is downstream exports from the restored watershed. Green boxes represent potential explanatory variables that will be explored.

Nitrification is a stepwise process by which ammonium is oxidized and transformed to nitrite (NO_2^-) and finally nitrate (NO_3^-) (Durand et al., 2011; Trimmer et al., 2012; Xia et al., 2018) via ammonium-oxidizing microorganisms and nitrite-oxidizing chemoautotrophs (Trimmer et al., 2012). For this process to occur, microorganisms need consistent temperatures and elevated oxygen levels. According to Durand et al. (2011), nitrification is essential for increasing N availability for primary producers.

Denitrification is the process by which microorganisms reduce and transform NO_3^- into gaseous N where it is returned to the atmosphere (Trimmer et al., 2012). Denitrification is influenced by a variety of environmental parameters such as oxygen concentration, the presence of plant life (i.e.: increasing flow paths for water), presence of nitrate, availability of electron donors (i.e.: organic carbon), abundance and diversity of denitrifying bacteria, and water residence times (Mitchell, 2001; Grimm et al., 2003; Durand et al., 2011; Zarnetske et al., 2011b; Xia et al., 2018; Dlamini et al., 2020). For this process to occur, low-oxygen or anoxic conditions must be met (Xia et al., 2018), therefore, denitrification tends to occur in soils, sediments, and anoxic bodies of water (Trimmer et al., 2012). Microorganisms that perform this transformation are heterotrophs and must have a readily available supply of organic carbon (Trimmer et al., 2012; Dlamini et al., 2020).

Finally, ammonification is the process by which organic N is degraded and transformed to inorganic N in the form of ammonium (Xia et al., 2018). This process begins when organisms release N, either through excretion or death, allowing microorganisms to colonize, decompose, and transform organic N into ammonium which is released into the ecosystem and becomes readily available for uptake by other

organisms (Trimmer et al., 2012). The process of ammonification can occur under both oxic and anoxic conditions (Xia et al., 2018) and frequently occurs in hyporheic zones (Zarnetske et al., 2011b).

Globally, general trends of N deposition have switched from N limitation to N saturation. Nitrogen cycling within streams tends to be efficient with streams converting ~50% of all N species present into N_2 . However, with excess N present, streams act as pipelines exporting the other half directly to the ocean (Xia et al., 2018). Within streams, solute concentrations are controlled by a variety of factors including geology, topography, groundwater-surface water (GW-SW) mixing, transit time of water through the watershed, and hydrology (Jacobs et al., 2017; Mitchell, 2001). According to Zarnetske et al. (2011a, 2011b), once solutes enter the stream, whether that stream exhibits net nitrification or denitrifications depends upon residence times of water in the hyporheic zone and uptake rates of oxygen.

The source, transport, and transformation of N varies within and among land-use types. Forested and low-order streams are hotspots of N processing. Within this land-use type, nutrients entering stream water are controlled by soil characteristics such as the content of organic/inorganic matter, soil temperature, pH, and microorganism communities (Jacobs et al., 2017) with the dominant form of N coming from litterfall via riparian vegetation (Pinay et al., 2002). Additionally, atmospheric N deposition is a growing source of concern, especially in Piedmont streams (Journey et al., 2018). Within unimpacted streams, nitrification is regulated by the availability of allochthonous inputs of carbon and nitrogen with elevated levels of carbon slowing, and elevated N increasing, the rates of nitrification (Starry et al., 2005). According to Starry et al. (2005), oxic

conditions are needed for the nitrification process to occur and there is a positive correlation between oxygen levels and nitrification rates. The presence of riparian and instream vegetation can enhance nitrification directly through increasing soil oxygen levels (Piña-Ochoa & Álvarez-Cobelas, 2006). Temperature of water regulates nutrient uptake with warmer temperatures increasing uptake via riparian vegetation. Moreover, nitrification tends to be highest in summer when the amount of carbon present is at its lowest and water temperatures are highest, leading to increased bacterial activity.

Having readily available sources of organic carbon increases the potential for denitrification (Xia et al., 2018). Instream woody debris creates areas of low flow and increased hyporheic exchange which leads to nutrient retention and transformation via denitrification. Increased residence times provide the anoxic conditions needed for denitrification (Grimm et al., 2003). Floodplain connections also lead to increased nutrient retention and transformation as water inundation brings nutrients which, if flood conditions are sustained, infiltrate and proceed through the denitrification process (Pinay et al., 2000). Lenat & Crawford (1994) found that forested systems (Yang et al., 2007) exhibited less nitrogen than urban land-uses and farmland. In general, these streams tend to be nutrient transformers as assimilation and deposition rates are proportional (Messina & Conner, 1998).

3.1.2 Urban/Impacted and Restored Stream Nitrogen Cycling

Similarly to their forested counterparts, nitrogen dynamics within urbanized/impacted streams are regulated by sources of N and N processing within the watershed (Reisinger et al., 2016). Urban streams exhibit elevated nutrient loads (Lavelle et al., 2019) and, when compared to natural streams, urban streams exhibit differing N

dynamics due to various anthropogenic impacts such as the addition of stormwater, sewage, reduced carbon inputs, and decreased hyporheic exchange (Schlesinger & Bernhardt, 2013; Ledford et al., 2017; Mayer et al., 2022). Further alteration of the landscape by humans increases impervious cover which acts as a funnel guiding urban overland flow into streams (Reisinger et al., 2016), limits the ability of nutrients to infiltrate and be transformed, and leads to increased nutrient loads in urban streams. Moreover, urban streams exhibit shorter residence times which leads to decreased N removal (Hall, 2007). Reisinger et al. (2016) found that nitrification and ammonification were abundant in urban streams while denitrification was highly variable but not significantly different than reference sites resulting in urban streams being net exporters of N.

Once a stream has been impaired, similar to the streams within the Reedy Creek watershed, human intervention may be necessary to restore unimpacted function. Restoration is an approach used to offset and remedy nutrient issues representative of degraded streams and has been used in the mitigation of nitrogen (Mayer et al., 2022). Typically, restoration takes place at the reach scale rather than the whole-watershed scale (i.e.: less than 1 km of stream length restored) (Bernhardt et al., 2005). However, reach-scale restoration has been shown to have little effect on N flux as these projects tend to be minimal in spatial scale, have limited long-term data (Lavelle et al., 2019; Mayer et al., 2022), and can be overwhelmed by upstream land-uses in the watershed beyond the restored channel. Additionally, Thompson et al. (2018) found that restoration at the reach-scale increased nutrient retention but the effects were not noticed downstream of the restoration site. According to Lewis et al. (1996), restoration projects need to take

place on large scales (i.e.: greater than 1 km of stream length), include headwaters, and address upstream land-use practices away from the immediate riparian corridor to ensure long-term success. Restoration of headwater streams can provide a buffer for downstream habitats as headwaters are extremely susceptible to nutrient pollution (Lavelle et al., 2019). In forested systems where large machinery is used for restoration, riparian vegetation is often removed, causing a shift from light-limited production to light-excess production in which instream algal production and biomass increase (Mallory & Richardson, 2005). This shift from light-limited to light-excess may cause an increase in nutrient transient storage instream nutrient retention as nutrients are absorbed by instream vegetation (Mallory & Richardson, 2005). This increase in light availability is due to the newly planted riparian vegetation needing decades to grow and provide shading to the stream.

Successional changes of riparian and instream vegetation will lead to altered nutrient dynamics; this is another reason that stressors should be addressed on larger spatial scales. Moreover, Filoso & Palmer (2011) hypothesized that whole-watershed restoration should lead to a decrease in N. Therefore, effective N management in urban watersheds should occur at the landscape scale creating conditions needed for capturing, retaining, and transforming N (Filoso & Palmer, 2011). Restoration practices, such as creating shallower stream channels, can lead to changes in water table depth which can also aid denitrification (Lammers & Bledsoe, 2017) through the rewetting of soil which provides conditions needed for denitrification (Hasselquist et al., 2017). To further expand on this point, Groffman & Crawford (2003) found that urban riparian soils had the same denitrifying potential as rural riparian soils and suggested that denitrification

rates could be increased if surface runoff in urban watersheds could be retained longer, via the use of stormwater detention ponds with vegetated wetlands, and allowed to slowly infiltrate into soils. Furthermore, reconnecting streams to the surrounding floodplain may increase C additions to the stream that have built up over the years which could further fuel denitrification (Hasselquist et al., 2017). Nutrient cycling is directly related to stream length, bank surface area, and hydrologic retention within the restored site. In other words, when restoration takes place at the reach scale, the conditions needed for nutrient cycling may not be met and, therefore, may not provide significant changes to nutrient export.

3.2 Study Objectives and Research Questions

The objective of this study was to determine if the whole-watershed stream restoration approach used in the Reedy Creek watershed in which stream channel morphology and characteristics were altered has changed N retention and seasonal concentration and flux. I acknowledge the role of stormflow for moving the majority of materials. However, for this study, I will focus on baseflow nitrogen dynamics as studies within this domain are limited (Hamel et al., 2015), baseflow is the condition where nutrient processing occurs, and baseflow is responsible for mediating water quality (Rolls et al., 2012). Moreover, according to Duncan et al. (2017) and Janke et al. (2014), nutrient concentrations have been shown to have little variation between various discharge levels. During their study, Duncan et al. (2017) determined that concentrations of nitrate decreased with increasing discharge or behaved chemostatically (i.e.: little change occurs). For this study I asked three questions:

(RQ3.1) *To what extent does restoration (i.e.: changes in AFDM and canopy cover) affect baseflow organic and inorganic nitrogen concentration and flux within the watershed?* I hypothesized that:

Species	Hypothesized change from pre- to post-restoration	Hypothesized mechanism
Total Dissolved Nitrogen (TDN)	Decrease	This is due to the characteristics of the watershed before restoration such as limited connection to floodplains, reduced stream complexity, and reduced hyporheic and GW-SW interactions which limits inorganic nitrogen transformation and, therefore, post-restoration total dissolved nitrogen content should be decreased due to a hypothesized increase in denitrification and transient storage of inorganic nitrogen. Furthermore, watershed characteristics will change post-restoration (e.g.: light availability, algae biomass, reconnected floodplains) leading to decreased N. Here, increased in light available to the stream via changes in canopy cover may increase algae biomass (AFDM) leading to an increase in transients inorganic N storage (e.g.: uptake by algae) and an increase in organic N sources (e.g.: decay of algae biomass)
Dissolved Organic Nitrogen (DON)	Increase	This is due to the streams being more connected to their floodplains allowing for the in-washing of particulate and dissolved organic N from the floodplain into the channel and an increase in algae biomass. Moreover, this is also due to a potential increase in microbial activity breaking down organic matter.
Nitrate	Decrease	This is due to changes in watershed characteristics which are hypothesized to lead to increases in light availability, algae biomass allowing for an increase in transient storage of nitrate (i.e.: storage within the biomass of instream algae). Moreover, changes in watershed hydrological functions such as reconnected floodplains, increased GW-SW interactions, and higher water tables are hypothesized to lead to increased water residence times and provide the conditions needed for nitrate transformation to occur.
Ammonium	Decrease	This is due to changes in watershed characteristics which are hypothesized to lead to increases in light availability, algae biomass allowing for an increase in transient storage of ammonium (i.e.: storage within the biomass of instream algae).

(RQ3.2) *Is there a dormant season component to organic and inorganic nitrogen*

flux in the pre- or post-restoration system? I hypothesized that:

Species	Change from pre- to post-restoration	Hypothesized mechanism
Total Dissolved Nitrogen (TDN)	Decrease	This is due to an increase in discharge during the dormant season overall coupled with changing characteristics within the watershed. For instance, before restoration the watershed exhibited limited connection to floodplains, reduced stream complexity, reduced hyporheic and GW-SW exchange, and decreased transient storage (i.e.: instream and riparian vegetation due to it being the dormant season) potential leading to increased inorganic nitrogen. Furthermore, watershed characteristics will change post-restoration (e.g.: light availability, algae biomass, reconnected floodplains) leading to decreased inorganic N as inorganic N is transformed and stored within the watershed.
Dissolved Organic Nitrogen (DON)	Increase	This is due to the streams being more connected to their floodplains allowing for the in-washing of particulate and dissolved organic N from the floodplain. These DON additions may be increased due to increases in biological breakdown during the fall as decomposers work to breakdown newly fallen organic matter. Also, end of lifecycle additions from increased instream biomass.
Nitrate	Decrease	This is due to a decrease in biological demand and changes in the watershed leading to more stable instream biomass (i.e.: limited shading as leaf off occurs) which uptake readily available nitrate.
Ammonium	Decrease	This is due to a decrease in biological demand and changes in the watershed leading to more stable instream biomass (i.e.: limited shading as leaf off occurs) which uptake readily available ammonium.

(RQ3.3) *Is there a growing season component to organic and inorganic*

nitrogen flux in the pre- or post-restoration system? I hypothesized that:

Species	Change from pre- to post-restoration	Hypothesized mechanism
Total Dissolved Nitrogen (TDN)	Decrease	This is due to a hypothesized decrease in discharge overall during the growing season. Furthermore, watershed characteristics will change post-restoration (e.g.: light availability, algae biomass, reconnected floodplains) leading to decreased overall inorganic N as readily available inorganic N will be taken up.
Dissolved Organic Nitrogen (DON)	Increase	This is due to the streams being more connected to their floodplains allowing for the in-washing of particulate and dissolved organic N from the floodplain. Also, additions from increased instream biomass at the end of their life cycles. Furthermore, increases in temperatures are hypothesized to lead to an increase in microorganism activity and reproduction further increasing biomass.
Nitrate	Decrease	This is due to changes in watershed characteristics where a reduction in canopy cover is hypothesized to increased light availability leading to an overall increase in algae biomass. Here, this biomass is hypothesized to increase uptake of readily available inorganic nitrogen species and lead to a decrease over the growing season. Moreover, increased GW-SW interactions and higher water tables may lead to increased nitrate uptake by riparian vegetation.
Ammonium	Decrease	This is due to changes in watershed characteristics where a reduction in canopy cover is hypothesized to increased light availability leading to an overall increase in algae biomass. Here, this biomass is hypothesized to increase uptake of readily available ammonium species and lead to a decrease over the growing season.

To test these hypotheses, baseflow surface water samples were collected and analyzed for N.

3.3 Methods

3.3.1 Field Methods

Water samples were taken weekly (Table 3.1) from May 2016 through March 2021 (two years before to two years after restoration) at the outlet of the Reedy Creek watershed during baseflow (site R2). Stream water samples were collected in acid

washed 1 L polyethylene bottles. Bottles and caps were rinsed three times in surface water directly downstream from the sampling site before sample was taken. The samples were taken by submerging the bottle upstream and allowing water to flow in, leaving no headspace. Bottles were capped and transported back to the lab for immediate filtering within approximately 4 hours of collection. Filtered samples for nutrient analysis were frozen in 50 mL polypropylene centrifuge tubes or refrigerated in 20mL glass vials until analyzed. Standard water quality parameters (e.g.: DO, pH, SC, & temperature) were also recorded at the collection site.

Table 3.1 Frequency of nitrogen, canopy cover, and AFDM sample collection

Sample Type	Approximate Frequency of Collection (except during construction period)	Location of Sampling
N water samples	Weekly	R1
Canopy cover	Seasonally (approximately 4 times a year)	All Reaches
Algal biomass	Seasonally (approximately 4 times a year)	All Reaches

Canopy closure was measured seasonally (Table 3.1) using a spherical densiometer at all restored stream reaches. Pre-restoration measurements were recorded at 0, 10, 20, 40, and 50 meters downstream along the 50-meter study reach (Spring 2016 - Winter 2018). Post-restoration measurements were taken every 20 (i.e.: 0, 20, 40, 60, 80, and 100) meters downstream along the 100-meter stream reach (Summer 2018 - Spring 2021). The change to sampled reach length was due to limited field assistants and safety of access during the before restoration period. At each measuring point, densiometer readings were taken in the north, south, east, and west directions. Measurements were taken from the middle of the wetted channel at each cross-section. Percent canopy cover

was calculated using standard methods (Strickler, 1959). The four points recorded at each reach were averaged for a total watershed canopy cover per seasonal sampling period.

To quantify seasonal algal biomass, benthic algae ash-free dry mass (AFDM) was assessed from sediment cores taken seasonally at all restored stream reaches (Table 3.1). For this study it was assumed that AFDM consisted of only algal biomass. Sediment samples were taken using a five cm diameter plastic corer inserted two cm into the sediments. Pre-restoration measurements used the same location scheme as canopy cover for both the Spring 2016 - Winter 2018 and Summer 2018 - Summer 2020. Changes to sampled reach length was due to limited field assistants and safety of access before restoration. Each sample was taken at a random point along the transect cross-section. Samples were stored in individually labeled whirl-packs and taken back to the lab where they were frozen until analyzed. All replicates were averaged for one data point per season for the entire watershed.

3.3.2 Lab Methods

In the lab, N water samples collected in were divided into $\text{NO}_3\text{-N}$, $\text{NH}_4^+\text{-N}$, and total dissolved N (TDN). To prepare samples for analysis, TDN was filtered through 0.7- μm glass fiber filters (GF/F) and $\text{NO}_3\text{-N}$ and $\text{NH}_4^+\text{-N}$ samples were vacuum filtered through 0.2-micron polyethersulfone filters using a borosilicate glass filtration funnel and tower which were rinsed between samples. Twenty mL of filtered TDN samples were transferred to 20 mL glass screw-top vials and refrigerated until analyzed. Forty mL of filtered samples for $\text{NO}_3\text{-N}$ and $\text{NH}_4^+\text{-N}$ were transferred to 50 mL centrifuge tubes and frozen at $\sim 0^\circ\text{C}$ until analyzed. Chemical preservatives were not added. Samples were thawed at room temperature for 24 hours before analysis. A Shimadzu TOC-TN Analyzer

was used to analyze TDN. Using this procedure, two standards of known concentration (10 mg/L and 20 mg/L C/N) were used to calibrate the instrument. Lachat flow injection analysis was used to measure concentrations of $\text{NO}_3\text{-N}$. Standard operating procedure QuikChem Method 10-107-04-1-A was followed for $\text{NO}_3\text{-N}$. Using this procedure, five standards of known concentrations (0.04, 0.16, 0.40, 1.60, and 4.00 mg N/L as NO_3) were used to calibrate the instrument. Lachat flow injection analysis was used to measure concentrations of $\text{NH}_4^+\text{-N}$. Standard operating procedure QuikChem Method 10-107-06-1-C was followed for $\text{NH}_4^+\text{-N}$. Using this procedure, seven standards of known concentrations (0.02, 0.04, 0.10, 0.40, 1.00, 1.60, and 4.00 mg N/L as $\text{NH}_4^+\text{-N}$) were used to calibrate the instrument.

Table 3.2 Summary of the N species and analysis methods used in this study.

Species	Form	Measured via
$\text{NO}_3\text{-N}$	Inorganic	Flow Injection Analyzer
$\text{NH}_4^+\text{-N}$	Inorganic	Flow Injection Analyzer
Total Dissolved Nitrogen	Inorganic and Organic	TOC/TN Analyzer
Dissolved organic Nitrogen	Organic	Calculated (Equation 3.4)

To quantify instream algal biomass, algae sediment cores were removed from the freezer and left to thaw for 24 hours in a dark refrigerator. After thawing, 3 mL of sediment was subsampled into in a 15 mL dark tube. 10 mL of 90% acetone was added to each tube and tubes were shaken vigorously on a vortex mixer. Samples were placed in the dark refrigerator for 24 hours before analysis. After sitting for 24 hours, samples were analyzed for AFDM following the EPA standard method 445.0 (1997). Samples were placed in pre-labeled and pre-weighed tins which were placed in a drying oven for 24-48

hours at 50 °C after which they were removed and weighed. Finally, tins were placed in a furnace at 500 °C for 4 hours, removed, and weighed for AFDM.

3.3.3 Statistical Analysis

Concentration and flux of N were divided into pre- and post-restoration periods and into respective seasons (winter/spring dormant season & summer/fall growing season) to identify potential patterns between before and after restoration periods and seasons. Flux was calculated using Equation 3.1 for the watershed outlet (Figure 1.4, R1). Discharge measurements at R1 were taken from a USGS gauging station located at the watershed outlet, approximately 150m downstream of the R1 surface water sampling location (USGS site 0212427947 (US Geological Survey, 2022a)). This monitoring station records data continuously at five-minute increments and provides values for daily mean discharge which were used for all flux calculations.

$$\text{Flux (mg s}^{-1}\text{)} = \text{concentration (mg/L)} * \text{discharge (ft}^3 \text{ s}^{-1}\text{)} * (28.3 \text{ L/ 1 ft}^3\text{)} \quad \text{Equation 3.1}$$

Percent canopy closure within the watershed was calculated using Equation 3.2. An average canopy closure percentage was calculated for each sampling date for the entire watershed.

$$\% \text{ canopy closure} = (\# \text{ of hits/total observations}) * 100 \quad \text{Equation 3.2}$$

Percent AFDM was calculated using Equation 3.3 to determine the percentage of instream algal biomass within the watershed.

$$\% \text{ AFDM} = (\text{ashed weight (g)} - \text{dry weight (g)}) / \text{dry weight (g)} * 100 \quad \text{Equation 3.3}$$

Dissolved organic N was calculated using Equation 3.4.

$$\text{Dissolved organic N} = \text{TDN (as N)} - (\text{nitrate (as N)} + \text{ammonium (as N)}) \quad \text{Equation 3.4}$$

Basic descriptive statistics (e.g.: mean, median, mode, standard error, standard deviation, range, minimum, and maximum) were calculated for nutrient concentration, flux, canopy cover, and AFDM. A non-parametric Mann-Whitney test was used to assess differences in nutrient concentrations, flux, canopy cover, and AFDM before and after treatment (i.e.: restoration) (Whitlock & Schluter, 2009). All statistical analyses were conducted in R (R Core Team, 2019). R packages used include base R (R Core Team, 2019) and jtools (Long, 2020). Jtools was used for summarizing regression results (Long, 2020).

To answer RQ3.1, *to what extent does restoration (i.e.: changes in AFDM and canopy cover) affect baseflow organic and inorganic nitrogen concentration and flux within the watershed*, box and whiskers plots were constructed for nitrogen concentration, nitrogen flux, canopy cover, and AFDM. Mann-Whitney tests were used to assess differences in organic and inorganic N concentration and flux before and after restoration (Whitlock & Schluter, 2009). Multiple regression analyses were used to determine if relationships exist between N concentration and flux, percent canopy closure, and algal biomass for seasonally collected data. In the event of missing data, such as a missing N analysis when all other analysis parameters were present, data

interpolation was used to create a complete dataset for regression analysis. All analyses took place on the pre- and post-restoration period datasets.

To answer RQ3.2, *is there a dormant season component to organic and inorganic nitrogen flux in the pre- or post-restoration system*, data were divided into dormant and growing seasons. Line graphs were created for the dormant seasons organic and inorganic N flux before and after restoration and visually examined. Seasonal plots were constructed with data from each year overlapping (i.e.: plots will run from October through March). Box and whiskers plots were created to visualize differences between the pre- and post-restoration dormant season organic and inorganic nitrogen flux. Mann-Whitney tests were used to assess differences in organic and inorganic N flux during the dormant season (Whitlock & Schluter, 2009). Data were compared using a seasonal index to assess the impact of seasonal months on nitrogen flux before and after restoration during the dormant season (Equation 3.5). Here, seasonal indices were used to determine which months had the largest and smallest impact on dormant season flux.

$$\text{Monthly Seasonal Indices} = \frac{\text{Average flux of the month}}{\text{Average flux of the period}} \quad \text{Equation 3.5.}$$

To answer RQ3.3, *Is there a growing season component to organic and inorganic nitrogen flux in the pre- or post-restoration system*, data were divided into dormant and growing seasons. Data were graphed for the growing seasons organic and inorganic N flux before and after restoration and visually examined. Seasonal plots were constructed with data from each year overlapping (i.e.: plots will run from April through September).

Box and whiskers plots were created to visualize differences between the pre- and post-restoration growing season organic and inorganic nitrogen flux. Mann-Whitney tests were used to assess differences in organic and inorganic N flux during the growing season (Whitlock & Schluter, 2009). Data were compared using a seasonal index to assess the impact of seasonal months on nitrogen flux before and after restoration during the growing season (Equation 3.5). Here, seasonal indices were used to determine which months had the largest and smallest impact on growing season flux.

3.4 Results

3.4.1 (RQ3.1) To what extent does restoration (i.e.: changes in AFDM and canopy cover) affect baseflow organic and inorganic nitrogen concentration and flux within the watershed?

3.4.1.1 Discharge

Average discharge at the watershed outlet increased from a pre-restoration average of 0.04 m³/s to a post-restoration average of 0.05 m³/s (Table 3.3). Here, pre-restoration discharge variation was lower (0.13 m³/s) than the post-restoration period (0.21 m³/s). Results of a Mann-Whitney test indicate that the increase in discharge was not statistically significant ($W = 788, p > 0.63$).

Table 3.3 Average discharge (Q) (m³/s) at the watershed outlet (R1) in the pre- and post-restoration periods

	<i>Pre-Q (m³/s)</i>	<i>Post-Q (m³/s)</i>
Mean	0.04	0.05
Standard Error	0.00	0.01
Median	0.04	0.03
Mode	0.06	0.02
Standard Deviation	0.03	0.05
Range	0.13	0.21
Minimum	0.01	0.00
Maximum	0.13	0.21

3.4.1.2 AFDM

Percent AFDM measurements were compared before and after restoration (Figure 3.2 & Figure 3.3). Here, the majority of percent AFDM data are found between 0-10 percent with few outliers (Figure 3.2). Data exhibit seasonal cycles with percent AFDM being highest in the summer and lowest during the winter (Figure 3.3). Data indicate that percent AFDM increased from a pre-restoration average of 1.6% to a post-restoration average of 2.8% (Table 3.4). Here, post-restoration streams were less shaded due to clear cutting of riparian vegetation allowing for an increase in sunlight to reach the stream and stimulate algae growth. Percent AFDM was higher (2.83%) and more variable (87.58%) in the post-restoration period when compared to the pre-restoration period (Figure 3.3). Average percent AFDM in the pre-restoration period was lowest (1.60%) and least variable (32.75%) than the post-restoration period (Figure 3.3). Results of a Mann-Whitney test indicate that this increase in percent AFDM, 1.2%, was statistically significant, $W = 10$, $p < 0.003$. Overall, restoration led to an increase in percent AFDM over time (Figure 3.4).

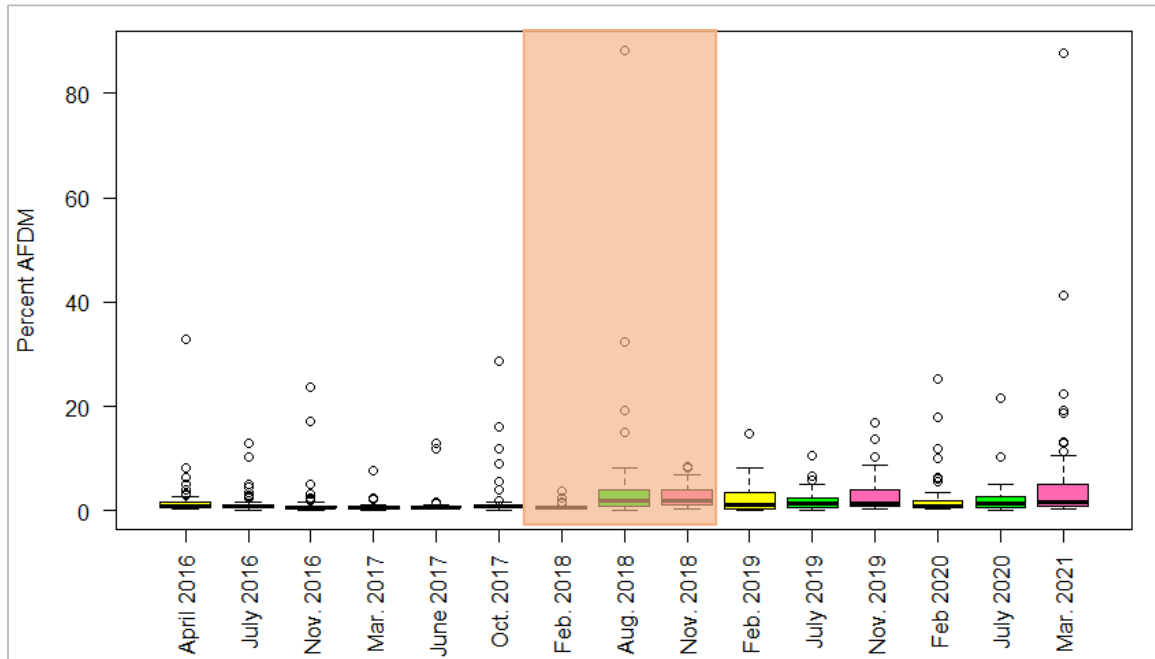


Figure 3.2 Box and whiskers plots for watershed average percent AFDM in the pre-, during, and post-restoration periods ($N = 11$ sites). Orange bar represents the during-construction period. Data represent entire sampling period from April 2016 – March 2021.

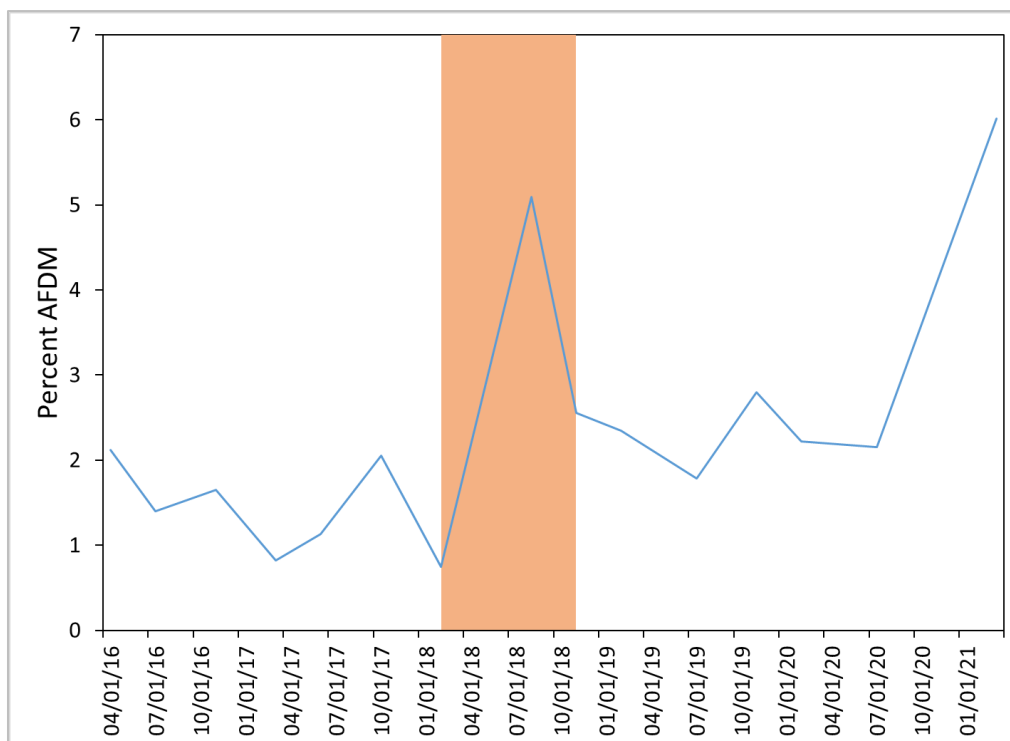


Figure 3.3 Watershed average percent AFDM in the pre-, during, and post-restoration periods. The during-restoration period is indicated by the orange bar. Data represent the sampling period from April 2016 – March 2021.

Table 3.4 Descriptive statistics for watershed average percent AFDM in the pre- and post-restoration periods within the watershed. Data represent the sampling period from April 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
Sample Size	7	9
Mean	1.60	2.83
Standard Error	0.18	0.31
Median	0.71	1.27
Mode	0.66	0.45
Standard Deviation	3.52	5.96
Range	32.75	87.58
Minimum	0.05	0.15
Maximum	32.80	87.73

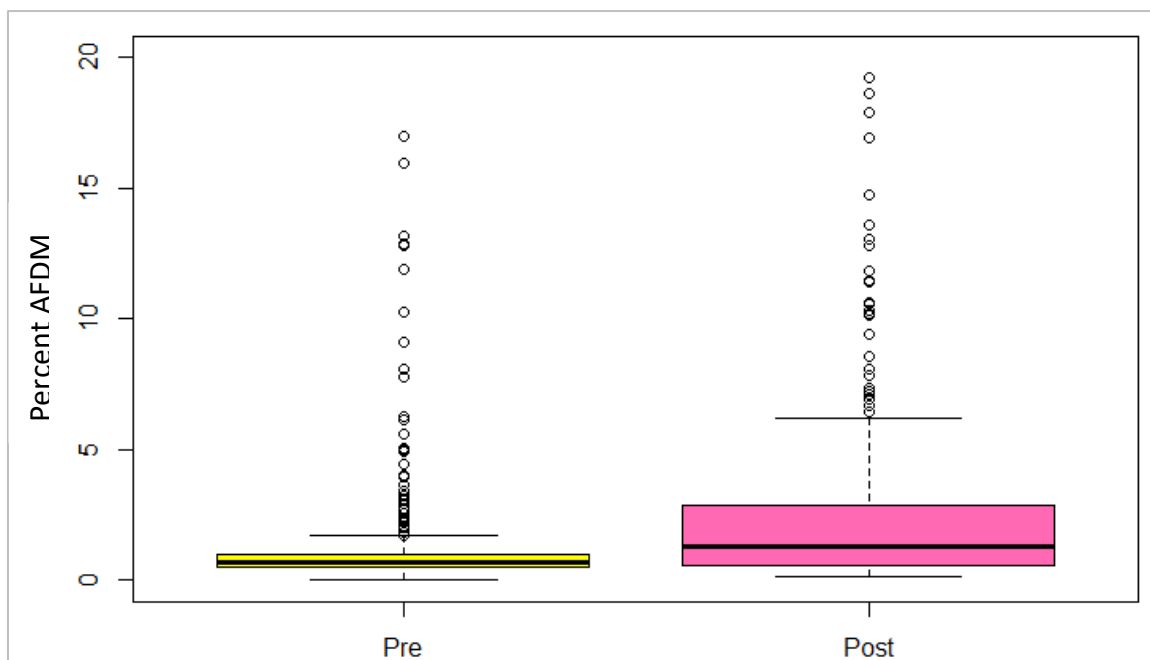


Figure 3.4 Boxplots of watershed average percent AFDM in the pre- and post-restoration periods (outliers above 20 % were removed from graph). Data represent the sampling period from April 2016 – March 2021.

3.4.1.3 Canopy Cover

Canopy cover results show seasonal cycling throughout the study period with summer highs and winter lows as foliage regrow and is lost in the riparian zone (Figure

3.5). Seasonal cycling was much reduced in the post-restoration period as trees were clear cut, greatly reducing the potential for shading of the stream. Average canopy cover was highest (56.26 %) and most variable (80.24 %) in the pre-restoration period (Figure 3.6). Average canopy cover in the post-restoration period was lower (49.62 %) and less variable (18.56 %) than the pre-restoration period (Figure 3.6). Overall, restoration led to a decrease in % canopy cover over time (Figure 3.5). Canopy cover measurements were compared before and after restoration. Data indicate that canopy cover decreased from a pre-restoration average of 56 % to a post-restoration average of 50 % (Table 3.5). This decrease in canopy cover, 6 %, was not statistically significant, $W = 52$, $p > 0.62$.

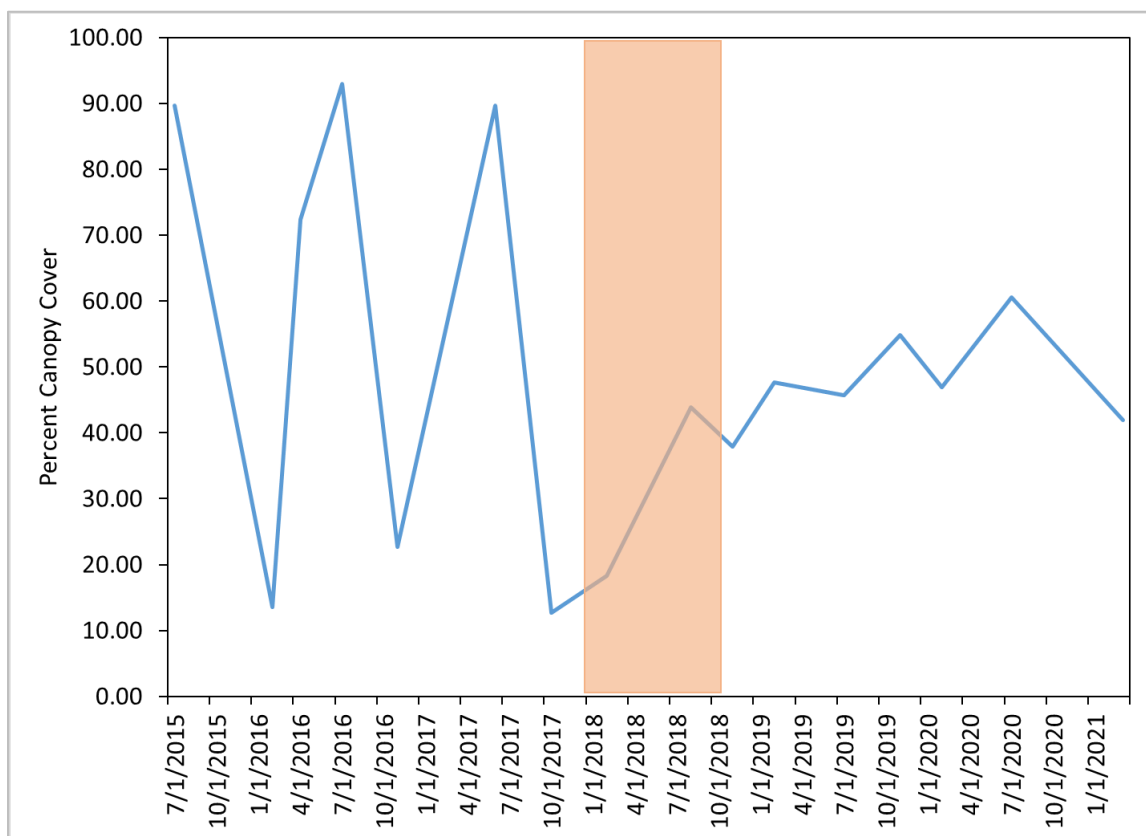


Figure 3.5 Watershed average percent canopy cover in the pre-, during, and post-restoration periods. The during-restoration period is indicated by the orange bar. Data represent the sampling period from July 2015 – March 2021.

Table 3.5 Descriptive statistics for watershed average percent canopy cover in the pre- and post-restoration periods. Data represent the sampling period from July 2015 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
	10	9
Mean	56.26	49.62
Standard Error	14.38	2.77
Median	72.42	47.35
Mode	/	/
Standard Deviation	38.05	6.79
Range	80.24	18.56
Minimum	12.75	41.97
Maximum	92.99	60.53

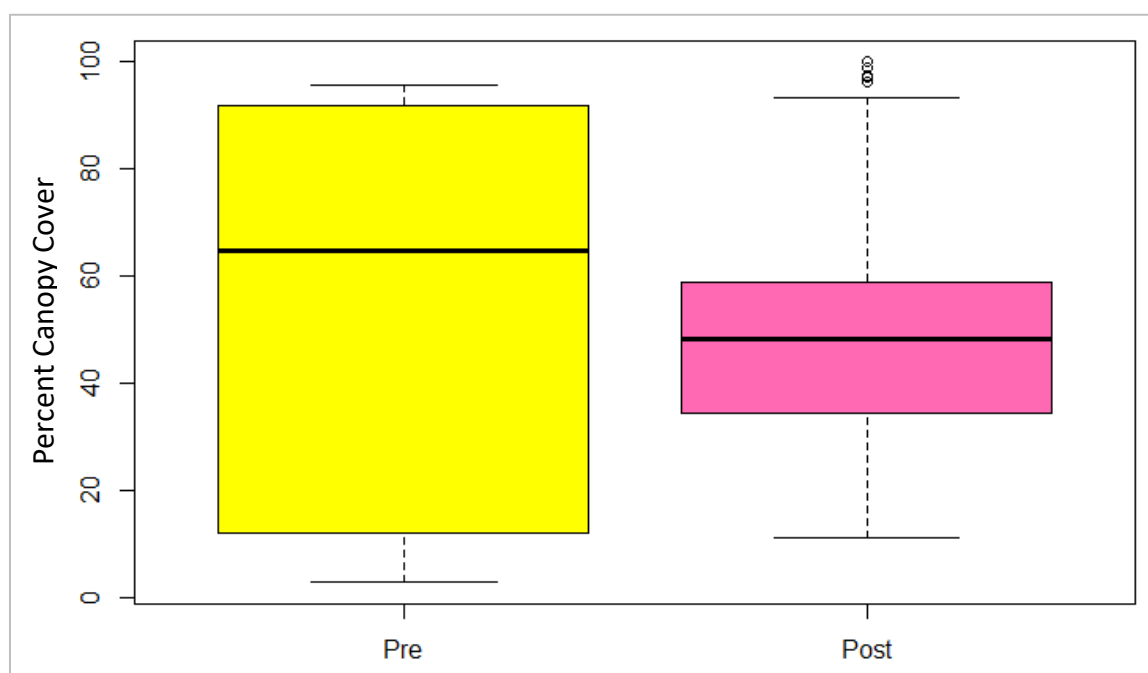


Figure 3.6 Boxplot of watershed average percent canopy cover in the pre- and post-restoration period. Data represent the sampling period from July 2015 – March 2021.

3.4.1.4 Nitrate

3.4.1.4.1 Nitrate Concentration

Nitrate (i.e.: nitrate + nitrite, hereafter referred to as nitrate, as nitrite is considered negligible) concentrations were measured before and after restoration. Nitrate concentration results show seasonal cycling throughout the study period with summer

lows and winter highs with seasonal cycling being much reduced in the post-restoration period. Data indicate that nitrate concentrations decreased from a pre-restoration average of 0.29 mg N/L to a post-restoration average of 0.08 mg N/L. Average nitrate concentrations were higher (0.39 mg N/L) and more variable (0.62 mg N/L) in the pre-restoration period when compared to the post-restoration period (Figure 3.7). Post-restoration, average nitrate concentrations were lower (0.08 mg N/L) and less variable (0.27 mg N/L) than pre-restoration (Figure 3.7). This decrease in nitrate, 0.21 mg N/L, was statistically significant, $W = 1335$, $p < 1.39 \times 10^{-09}$. Overall, restoration led to a decrease in nitrate concentrations over time (Figure 3.8).

Table 3.6 Descriptive statistics for nitrate concentrations (mg N/L) in the pre- and post-restoration periods at the R1 (watershed outlet) site. Data represent the sampling period from May 2016 – March 2021.

Summary Statistic	Pre-Restoration mg N/L	Post-Restoration mg N/L
Mean	0.39	0.08
Standard Error	0.07	0.02
Median	0.29	0.06
Mode	/	0.00
Standard Deviation	0.41	0.09
Range	2.36	0.27
Minimum	0.06	0.00
Maximum	2.43	0.27

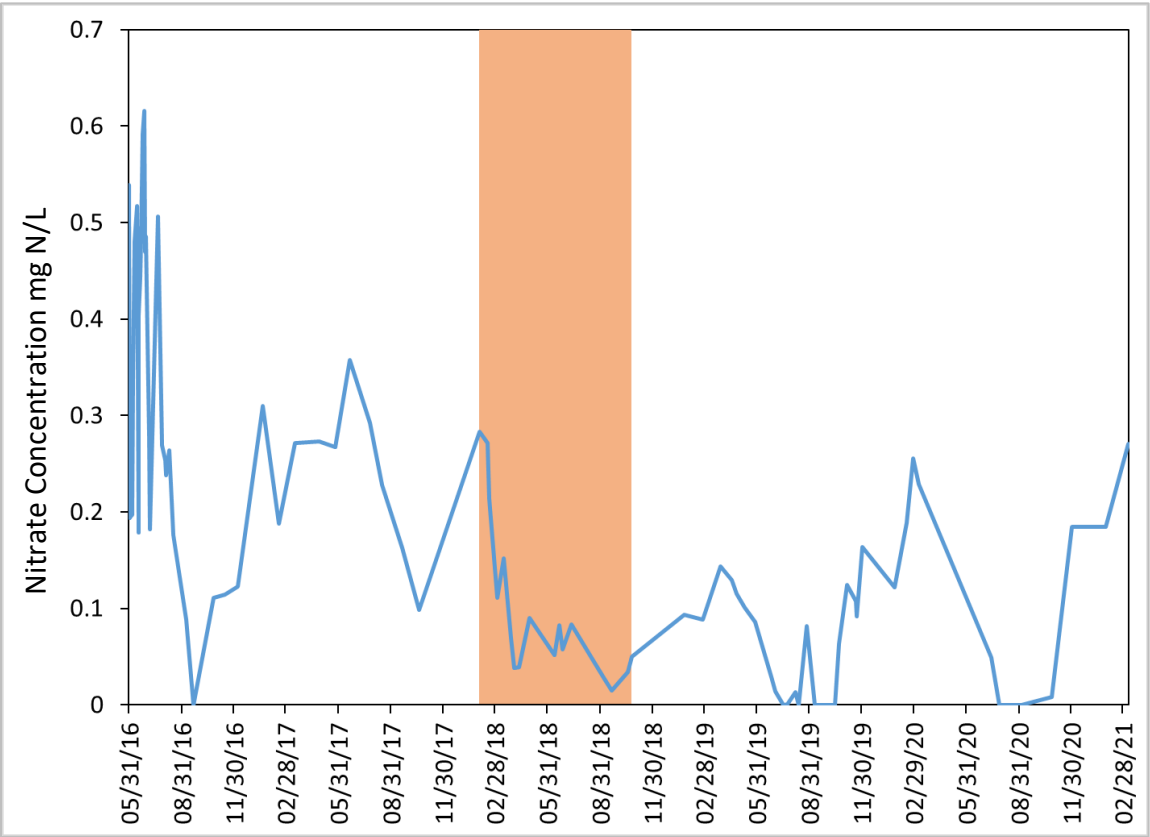


Figure 3.7 Nitrate concentrations (mg N/L) at site R1 in the pre-, during, and post-restoration periods at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from May 2016 – March 2021.

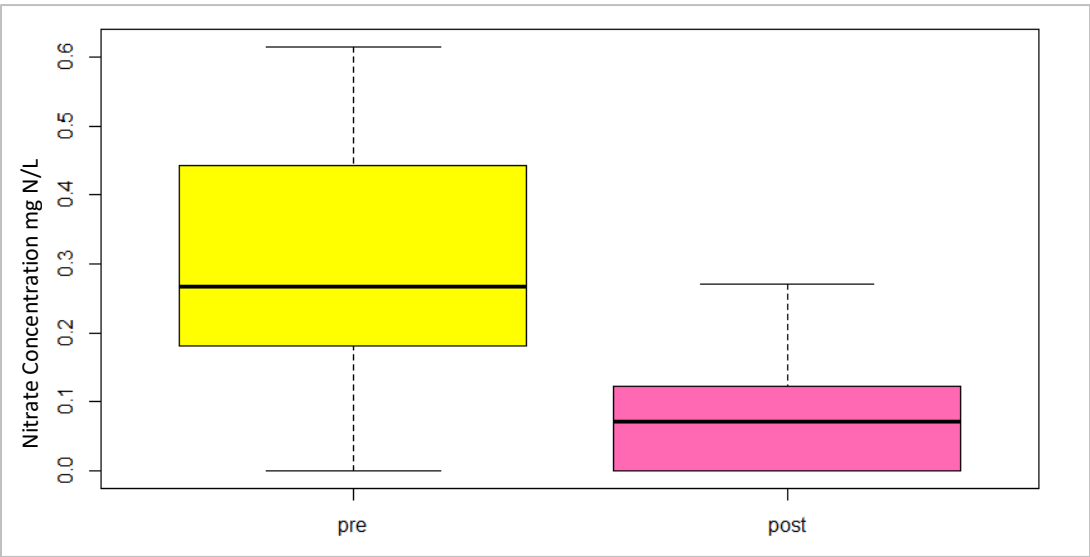


Figure 3.8 Box and whiskers plot for nitrate concentration (mg N/L) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from May 2016 – March 2021.

3.4.1.4.1.1 Nitrate Concentration Regression Results

Nitrate concentrations were measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the nitrate concentration, canopy cover, and AFDM, ($F(2, 12) = 0.11$, $p > 0.9$, $R^2 = 0.02$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was not a significant effect between the nitrate concentration, canopy cover, and AFDM, ($F(2, 3) = 0.50$, $p > 0.65$, $R^2 = 0.25$).

3.4.1.4.2 Baseflow Nitrate Flux

Baseflow nitrate flux measurements were compared before and after restoration. Nitrate flux results show seasonal cycling throughout the study period with summer highs and winter lows pre-restoration and winter/summer highs and fall lows post-restoration. Data indicate that nitrate flux decreased from a pre-restoration average of 12.86 mg N/s to a post-restoration average of 5.13 mg N/s (Table 3.7). Average nitrate flux was higher (12.86 mg N/s) and more variable (45.75 mg N/s) in the pre-restoration period when compared to post-restoration (Figure 3.9). Post-restoration, average nitrate flux was lower (5.13 mg N/s) and less variable (23.75 mg N/s) than pre-restoration (Figure 3.9). Results of a Mann-Whitney test indicate that this decrease in nitrate flux, 7.73 mg N/s, was statistically significant, $W = 1134$, $p < 6.20 \times 10^{-05}$. Overall, restoration led to a decrease in nitrate flux over time (Figure 3.10).

Table 3.7 Descriptive statistics for nitrate flux (mg N/s) at the R1 site. Data represent the sampling period from May 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
Mean	12.86	5.13
Standard Error	1.64	1.08
Median	11.35	1.44
Mode	/	0.00
Standard Deviation	10.13	6.77
Range	46.75	23.75
Minimum	0.00	0.00
Maximum	46.75	23.75

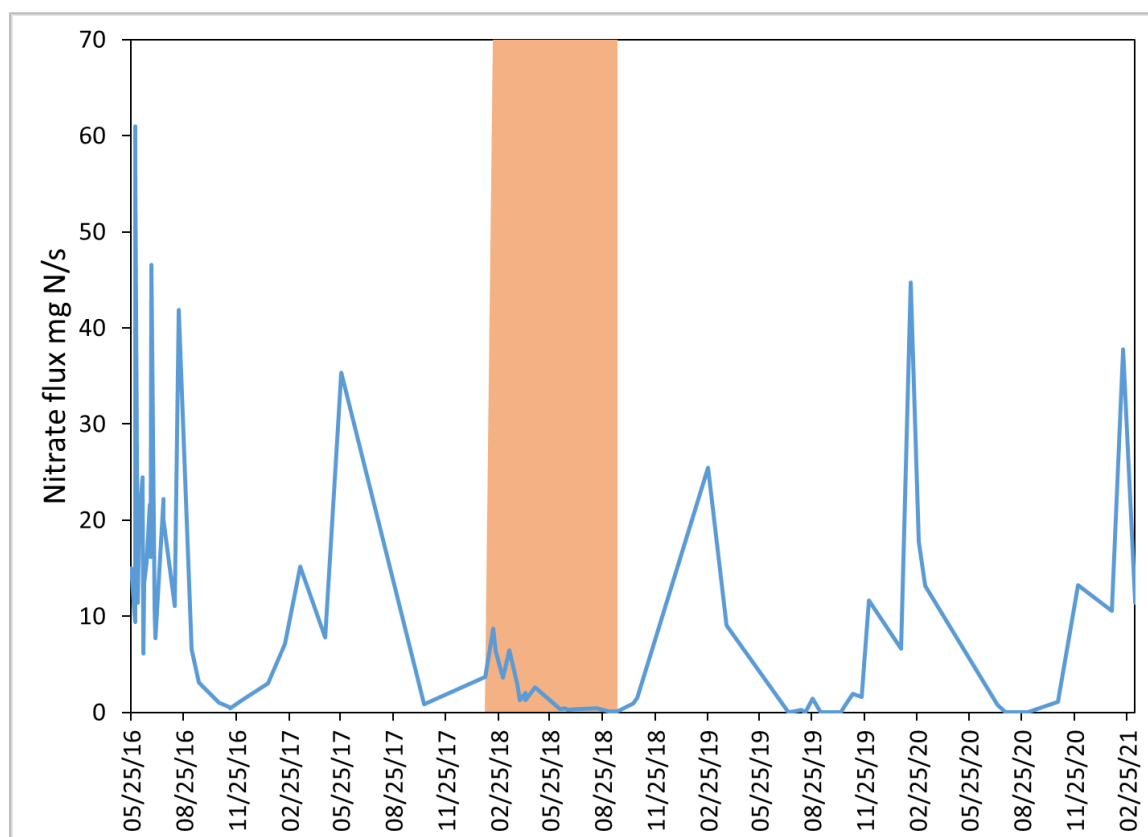


Figure 3.9 Nitrate flux (mg N/s) dynamics over time at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from May 2016 – March 2021.

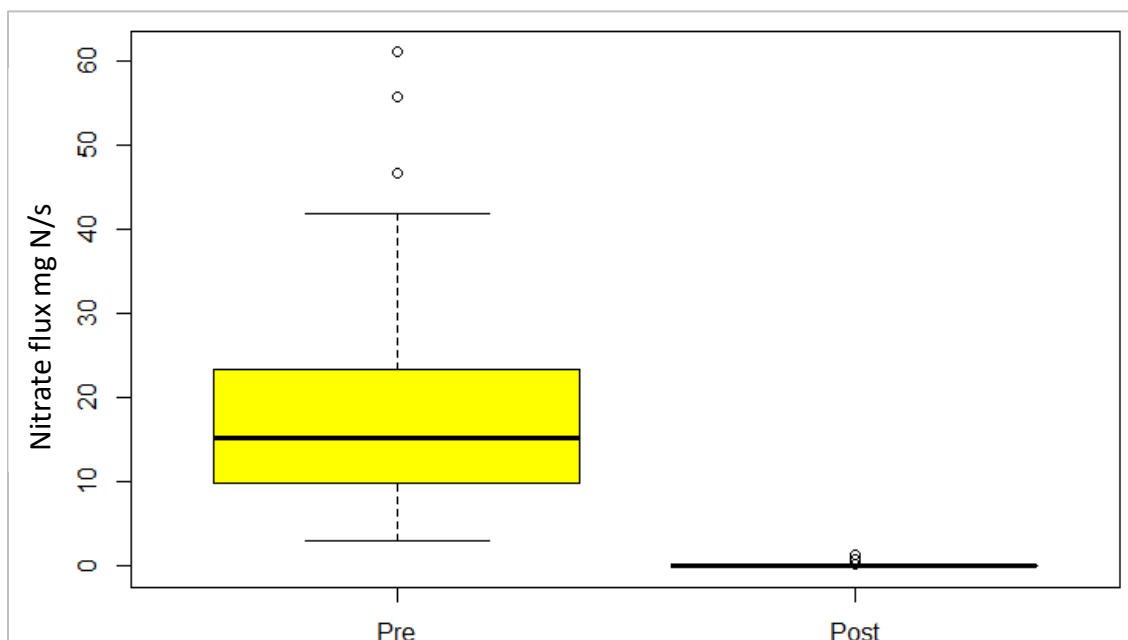


Figure 3.10 Box and whiskers plot for nitrate flux (mg N/s) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from May 2016 – March 2021.

3.4.1.4.2.1 Nitrate Flux Regression Results

Nitrate flux were measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the nitrate flux, canopy cover, and AFDM, ($F(2, 2) = 12.00$, $p > 0.08$, $R^2 = 0.92$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was not a significant effect between the nitrate flux, canopy cover, and AFDM, ($F(2, 3) = 0.52$, $p > 0.64$, $R^2 = 0.26$).

3.4.1.5 Ammonium

3.4.1.5.1 Ammonium Concentration

Ammonium concentration measurements were compared before and after restoration. Ammonium concentration results show seasonal cycling throughout the study period with summer highs, winter lows, and seasonal cycling being much reduced in the pre-restoration period. Data indicate that ammonium increased from a pre-restoration

average of 0.02 mg N/L to a post-restoration average of 0.03 mg N/L (Table 3.8).

Ammonium was lower (0.02 mg N/L) and less variable (0.08 mg N/L) in the pre-restoration period when compared to the post-restoration period (Figure 3.11). Average ammonium in the post-restoration period was highest (0.03 mg N/L) and more variable (0.13 mg N/L) than the pre-restoration period (Figure 3.11). Results of a Mann-Whitney test indicate this increase in ammonium, 0.01 mg N/L, was not statistically significant, $W = 750.5$, $p < 0.92$. Overall, restoration led to an increase in average ammonium concentrations over time (Figure 3.12). However, the median ammonium concentration decreased in the post-restoration period indicating that data may be skewed.

Table 3.8 Summary statistic for ammonium (mg N/L) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from June 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
	mg N/L	mg N/L
Mean	0.02	0.03
Standard Error	0.00	0.01
Median	0.02	0.01
Mode	0.00	0.00
Standard Deviation	0.02	0.04
Range	0.08	0.13
Minimum	0.00	0.00
Maximum	0.08	0.13

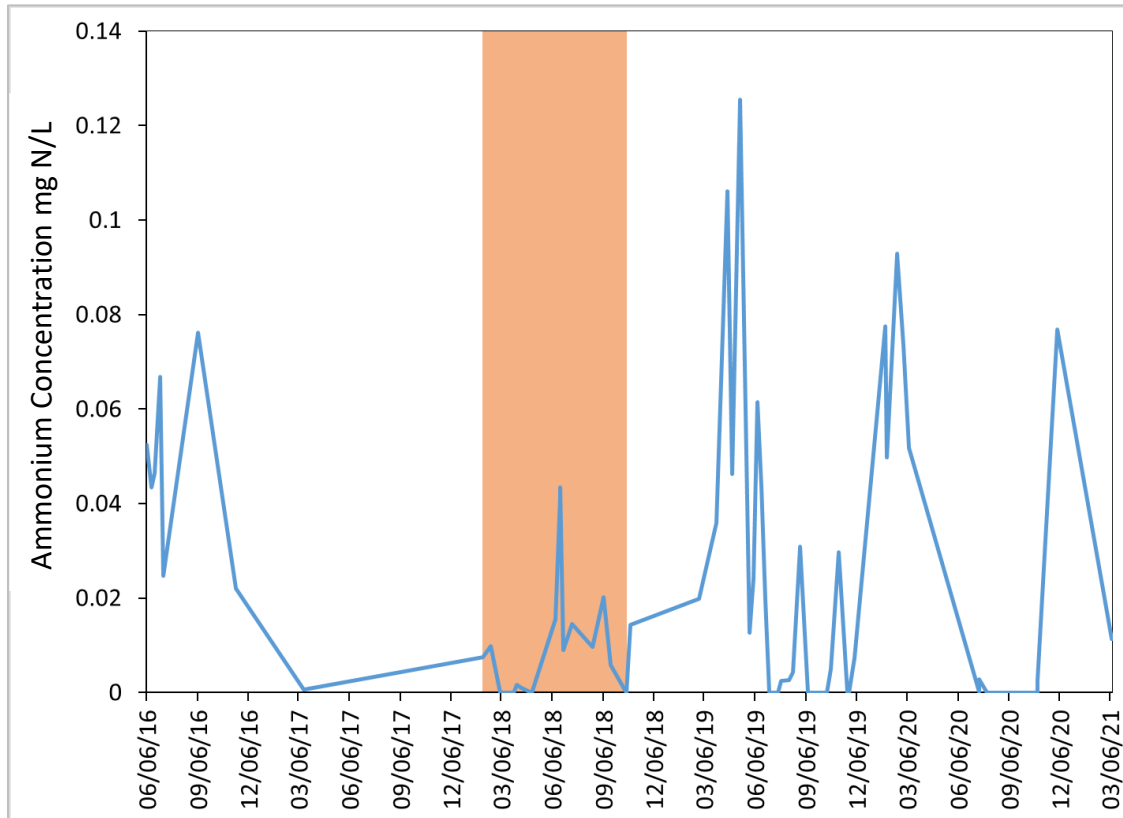


Figure 3.11 Ammonium concentration (mg N/L) in the pre-, during, and post-restoration periods at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from June 2016 – March 2021.

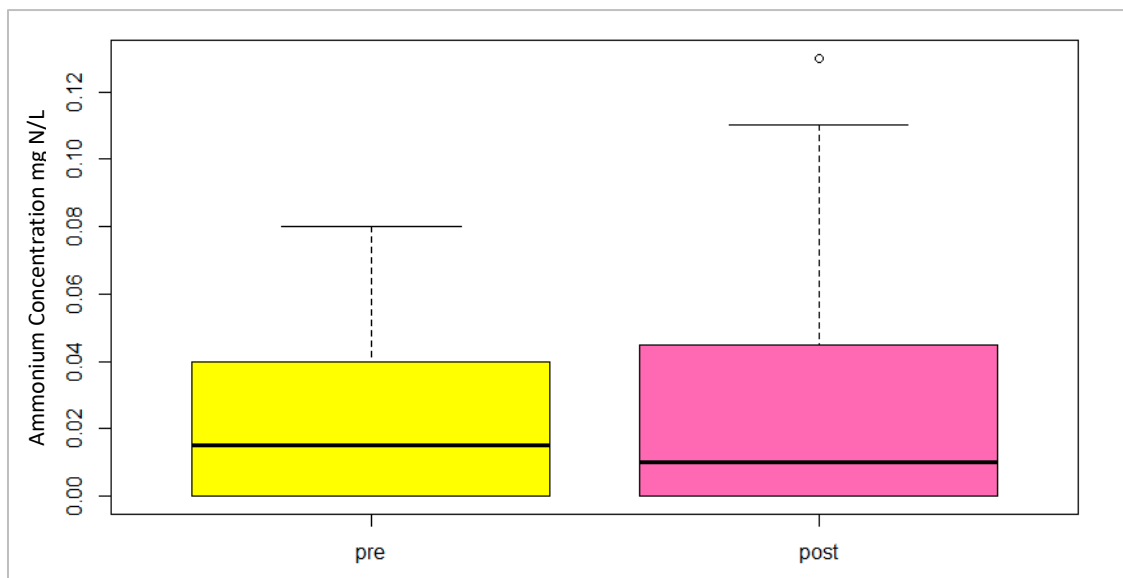


Figure 3.12 Box and whiskers plot for ammonium concentration (mg N/L) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from June 2016 – March 2021.

3.4.1.5.1.1 Ammonium Concentration Regression Results

Ammonium concentrations were measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the ammonium concentration, canopy cover, and AFDM, ($F(2, 12) = 1.81$, $p > 0.2$, $R^2 = 0.23$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was not a significant effect between the total ammonium concentration, canopy cover, and AFDM, ($F(2, 3) = 1.71$, $p > 0.32$, $R^2 = 0.53$).

3.4.1.5.2 Baseflow Ammonium Flux

Baseflow ammonium flux were compared before and after restoration. Ammonium flux results show seasonal cycling throughout the study period with summer highs, winter lows, and seasonal cycling being much reduced in the pre-restoration period. Data indicate that ammonium flux increased from a pre-restoration average of 0.87 mg N/s to a post-restoration average of 2.15 mg N/s (Table 3.9). Average ammonium flux was lower (1.95 mg N/s) and less variable (6.44 mg N/s) in the pre-restoration period (Figure 3.13). Average ammonium flux in the post-restoration period was higher (2.15 mg N/s) and more variable (21.16 mg N/s) than the pre-restoration period (Figure 3.13). Results of a Mann-Whitney test indicate that this increase in ammonium, 1.28 mg N/s, was not statistically significant, $W = 769$, $p > 0.77$. Overall, restoration led to an increase in average ammonium flux over time (Figure 3.14).

Table 3.9 Descriptive statistics for ammonium flux (mg N/s) at the R1 site. Data represent the sampling period from June 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
	mg N/s	mg N/s
Mean	0.87	2.15
Standard Error	0.21	0.66
Median	0.41	0.45
Mode	0.00	0.00
Standard Deviation	1.28	4.13
Range	6.44	21.16
Minimum	0.00	0.00
Maximum	6.44	21.16

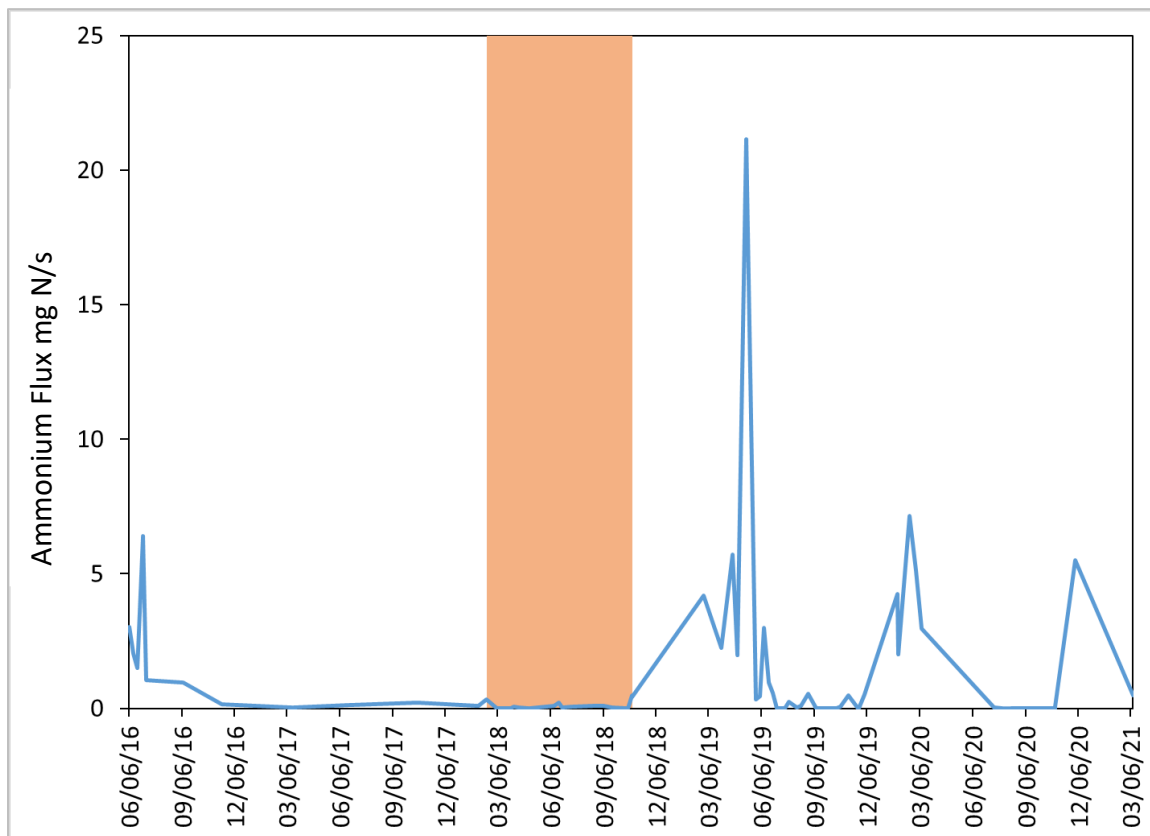


Figure 3.13 Ammonium flux (mg/s) dynamics over time at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from June 2016 – March 2021.

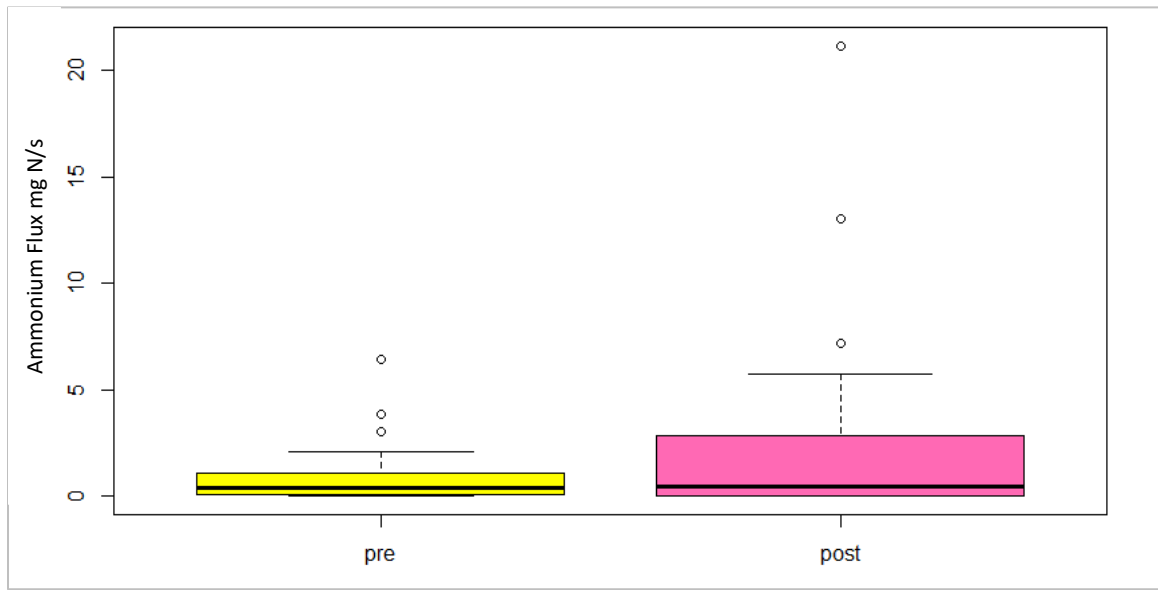


Figure 3.14 Box and whiskers plot for ammonium flux (mg/s) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from June 2016 – March 2021.

3.4.1.5.2.1 Ammonium Flux Regression Results

Ammonium flux were measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the ammonium flux, canopy cover, and AFDM, ($F(2, 12) = 0.30$, $p > 0.75$, $R^2 = 0.05$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was a significant effect between the ammonium flux, canopy cover, and AFDM, ($F(2, 3) = 75.82$, $p < 0.001$, $R^2 = 0.98$).

3.4.1.6 Total Dissolved Nitrogen

3.4.1.6.1 Total Dissolved Nitrogen (TDN) Concentration

Total dissolved nitrogen concentrations were measured before and after restoration. Data indicate that TDN concentrations decreased from a pre-restoration average of 0.43 mg N/L to a post-restoration average of 0.32 mg N/L (Table 3.10).

Average TDN concentration in the pre-restoration period was higher (0.43 mg N/L) and more variable (0.77 mg N/L) than post-restoration (Figure 3.15). Average TDN concentration was lower (0.32 mg N/L) and less variable (0.31 mg N/L) in the post-restoration period when compared to pre-restoration (Figure 3.15). This decrease in TDN, 0.11 mg N/L, was statistically significant, $W = 1011$, $p < 0.006$. Overall, restoration led to a decrease in TDN concentration over time (Figure 3.16).

Table 3.10 Total dissolved nitrogen concentration (mg N/L) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from May 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
Mean	0.43	0.32
Standard Error	0.03	0.02
Median	0.44	0.32
Mode	0.00	0.48
Standard Deviation	0.21	0.10
Range	0.77	0.31
Minimum	0.00	0.17
Maximum	0.77	0.48

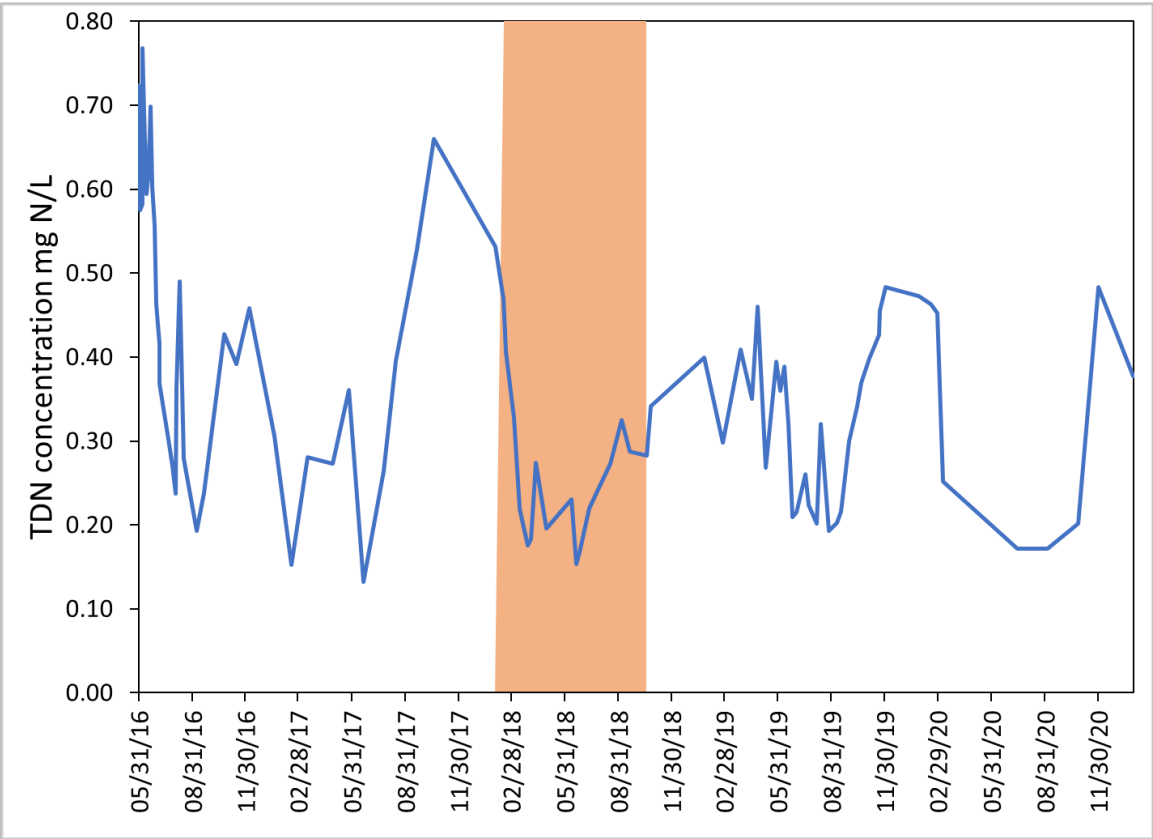


Figure 3.15 Total dissolved nitrogen concentrations (mg N/L) in the pre-, during, and post-restoration periods at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from May 2016 – March 2021.

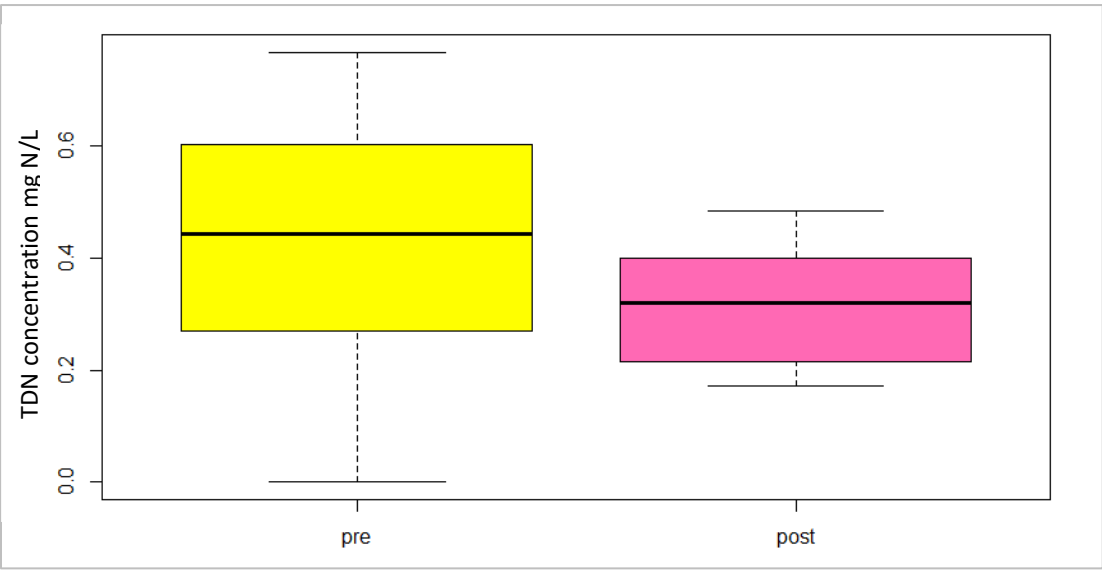


Figure 3.16 Box and whiskers plot for total dissolved nitrogen concentration (mg N/L) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from May 2016 – March 2021.

3.4.1.6.1.1 *Total Dissolved Nitrogen Concentration Regression Results*

Total dissolved nitrogen concentrations were measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the total dissolved nitrogen concentration, canopy cover, and AFDM, ($F(2, 12) = 0.42$, $p > 0.67$, $R^2 = 0.07$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was not a significant effect between the total nitrogen concentration, canopy cover, and AFDM, ($F(2, 3) = 3.65$, $p > 0.16$, $R^2 = 0.71$).

3.4.1.6.2 Baseflow Total Dissolved Nitrogen (TDN) Flux

Baseflow total dissolved nitrogen flux measurements were compared before and after restoration. Data indicate that TDN flux decreased from a pre-restoration average of 16.96 (mg N/s) to a post-restoration average of 16.94 (mg N/s) (Table 3.11). Average TDN flux was higher (16.96 mg N/s) and less variable (74.68 mg N/s) in the pre-restoration period (Figure 3.17). Average TDN flux in the post-restoration period was lower (16.94 mg N/s) and more variable (82.91 mg N/s) than the pre-restoration period (Figure 3.17). Results of a Mann-Whitney test indicate that this decrease in TDN flux, 1.91 mg N/s, was not statistically significant, $W = 821$, $p < 0.41$. Overall, restoration led to a decrease in total nitrogen flux over time (Table 3.11).

Table 3.11 Descriptive statistics for total dissolved nitrogen flux (mg N/s) at the R1 site. Data represent the sampling period from May 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
Mean	16.96	16.94
Standard Error	2.48	3.32
Median	14.72	7.38
Mode	0.00	/
Standard Deviation	15.30	20.73
Range	74.68	82.91
Minimum	0.00	1.17
Maximum	74.68	84.07

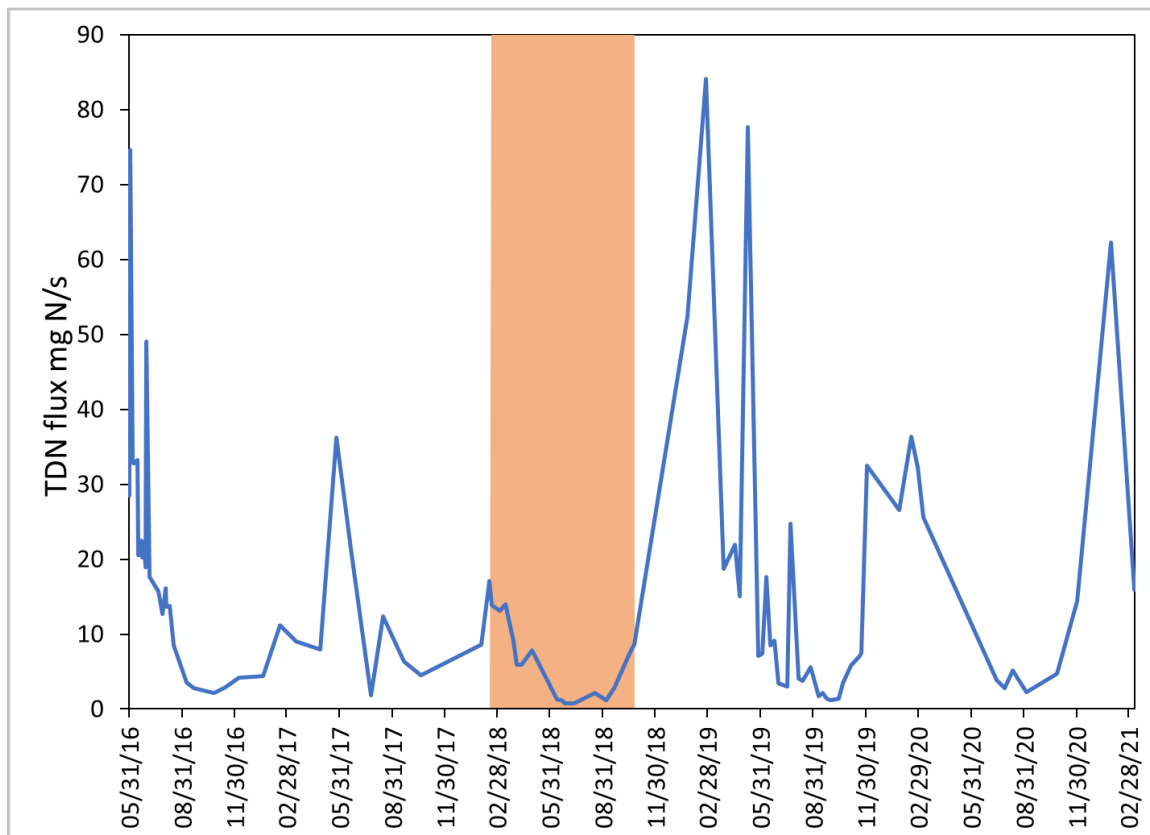


Figure 3.17 Total dissolved nitrogen (mg N/s) flux dynamics at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from May 2016 – March 2021.

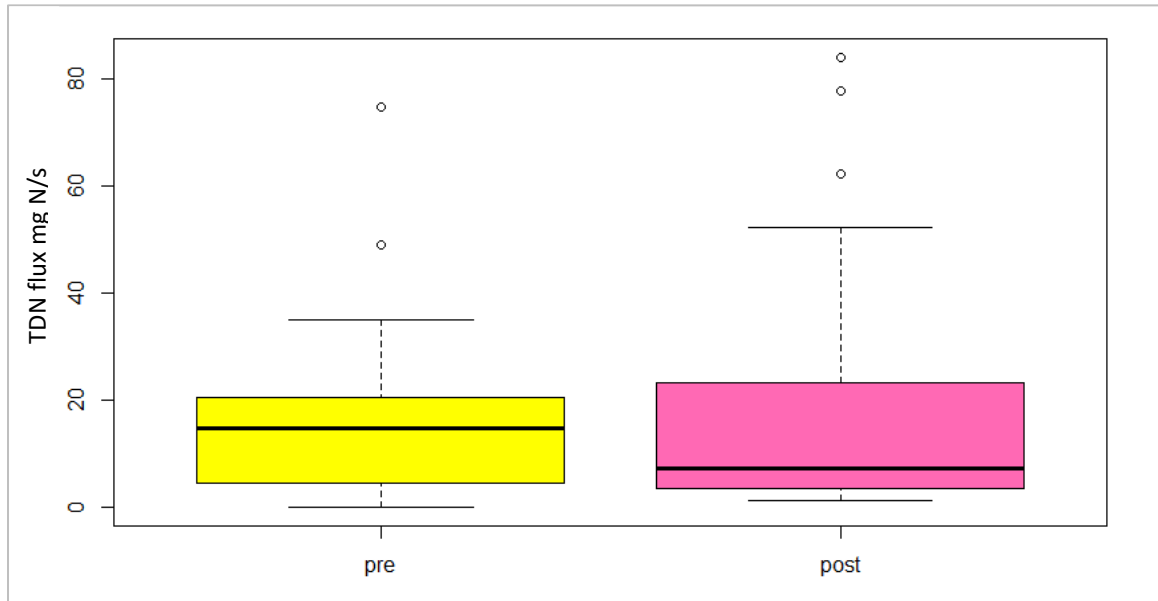


Figure 3.18 Box and whiskers plot for total dissolved nitrogen flux (mg N/s) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from May 2016 – March 2021.

3.4.1.6.2.1 Total Dissolved Nitrogen Flux Regression Results

Total dissolved nitrogen was measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the TDN flux, canopy cover, and AFDM, ($F(2, 12) = 0.42$, $p > 0.67$, $R^2 = 0.07$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was not a significant effect between the total nitrogen flux, canopy cover, and AFDM, ($F(2, 3) = 2.47$, $p > 0.23$, $R^2 = 0.62$).

3.4.1.7 Dissolved Organic Nitrogen (DON)

3.4.1.7.1 Dissolved Organic Nitrogen (DON) Concentration

Dissolved organic nitrogen concentration measurements were compared before and after restoration. DON concentration increased from a pre-restoration average of 0.15 mg N/L to a post-restoration average of 0.21 mg N/L (Table 3.12). Average DON was

lowest (0.15 mg N/L) and most variable (0.51 mg N/L) in the pre-restoration period (Figure 3.19). Average DON in the post-restoration period was highest (0.21 mg N/L) and most variable (0.34 mg N/L) than the pre-restoration period (Figure 3.19). Results of a Mann-Whitney test indicate that this increase in DON, 0.02 mg N/L, was statistically significant, $W = 465$, $p < 0.005$. Overall, restoration led to an increase in dissolved organic nitrogen over time (Figure 3.20).

Table 3.12 Descriptive statistics for total organic nitrogen concentration (mg N/L) at the R1 site. Data represent the sampling period from May 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
Mean	0.15	0.21
Standard Error	0.02	0.01
Median	0.12	0.21
Mode	0.00	/
Standard Deviation	0.14	0.07
Range	0.51	0.34
Minimum	0.00	0.00
Maximum	0.51	0.34

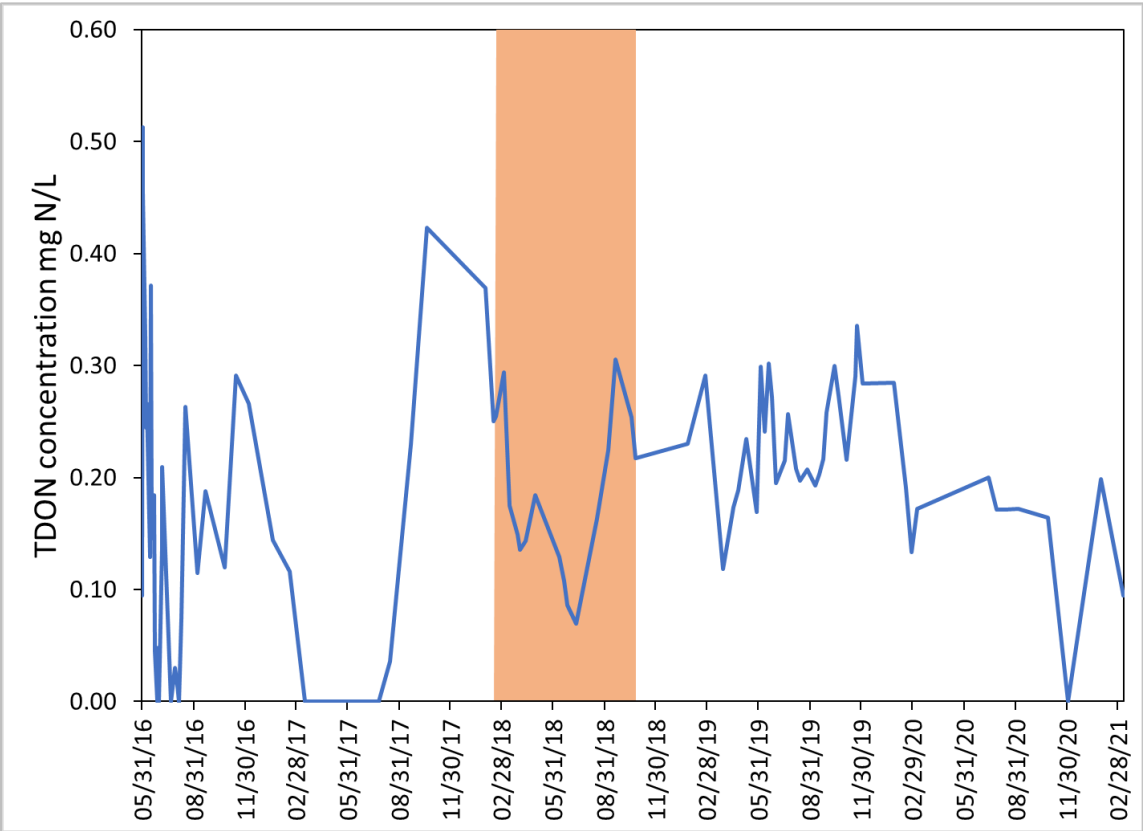


Figure 3.19 Dissolved organic nitrogen (mg N/L) concentrations over time at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from May 2016 – March 2021. Zeros are non-detects and were left as zeros in the dataset.

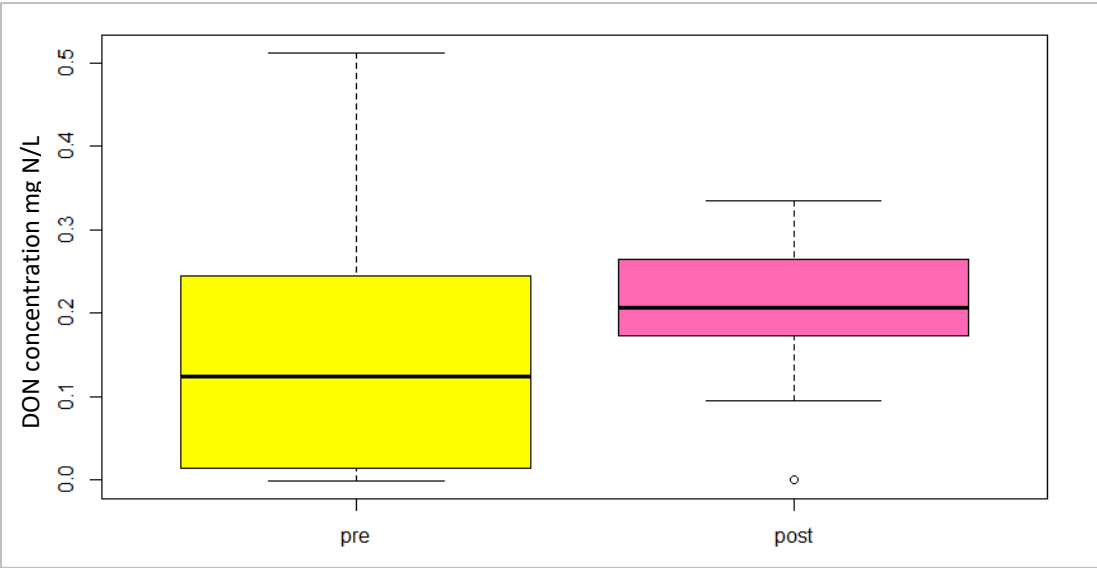


Figure 3.20 Box and whiskers plot for DON concentration (mg N/L) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from May 2016 – March 2021.

3.4.1.7.1.1 Dissolved Organic Nitrogen Concentration Regression Results

Dissolved organic nitrogen was measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the dissolved organic nitrogen concentration, canopy cover, and AFDM, ($F(2, 12) = 0.62$, $p > 0.56$, $R^2 = 0.09$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was not a significant effect between the dissolved organic nitrogen concentration, canopy cover, and AFDM, ($F(2, 3) = 1.33$, $p > 0.39$, $R^2 = 0.47$).

3.4.1.7.2 Baseflow Dissolved organic Nitrogen Flux

Baseflow dissolved organic nitrogen flux measurements were compared before and after restoration. Data indicate that DON flux increased from a pre-restoration average of 5.50 mg N/s to a post-restoration average of 9.77 mg N/s (Table 3.13). Average DON flux was lower (5.50 mg N/s) and less variable (49.57 mg N/s) in the pre-restoration period (Figure 3.21). Average canopy cover in the post-restoration period was higher (9.77 mg N/s) and more variable (61.32 mg N/s) than the pre-restoration period (Figure 3.21). Results of a Mann-Whitney test indicate that this increase in total organic nitrogen, 4.27 mg N/s, was statistically significant, $W = 447$, $p < 0.003$. Overall, restoration led to an increase in average DON flux over time (Figure 3.22).

Table 3.13 Descriptive statistics of dissolved organic nitrogen flux (mg N/s) at the R1 site. Data represent the sampling period from May 2016 – March 2021.

Summary Statistic	Pre-Restoration	Post-Restoration
Mean	5.50	9.77
Standard Error	1.49	1.99
Median	2.76	5.15
Mode	0.00	/
Standard Deviation	9.17	12.44
Range	49.50	61.32
Minimum	0.00	0.00
Maximum	49.51	61.32

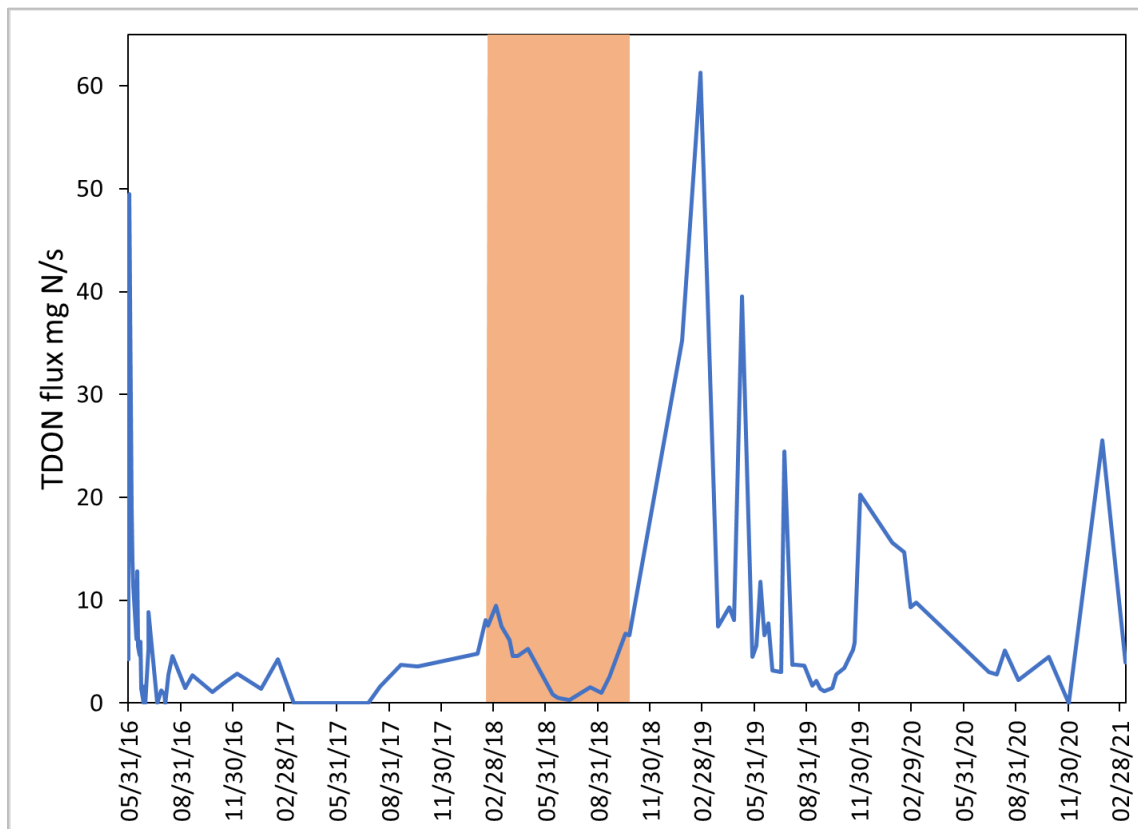


Figure 3.21 Dissolved organic nitrogen flux (mg N/s) over time at the R1 site. The during-restoration period is indicated by the orange bar. Data represent the sampling period from May 2016 – March 2021. Zeros are non-detects and were left as zeros in the dataset.

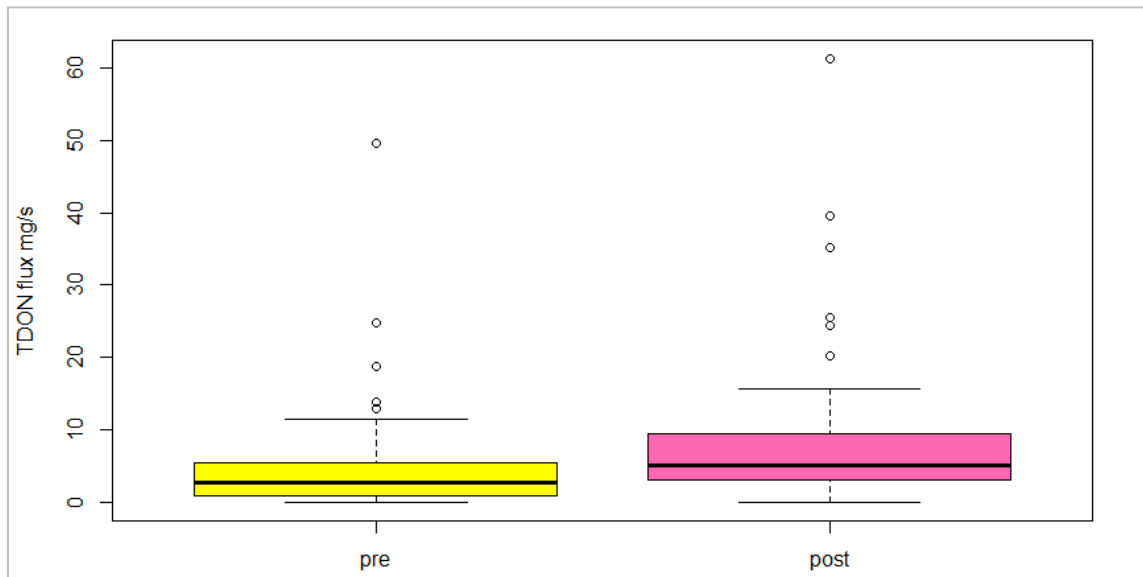


Figure 3.22 Box and whiskers plot for dissolved organic nitrogen flux (mg/s) in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from May 2016 – March 2021.

3.4.1.7.2.1 Dissolved organic Nitrogen Flux Regression Results

Dissolved organic nitrogen flux was measured before and after restoration and were compared to percent canopy cover and percent AFDM. Results of the multiple linear regression indicate that there was not a collective significant effect between the dissolved organic nitrogen flux, canopy cover, and AFDM, ($F(2, 12) = 0.11$, $p > 0.9$, $R^2 = 0.02$) pre-restoration. Post-restoration, results of the multiple linear regression indicate that there was not a significant effect between the dissolved organic nitrogen flux, canopy cover, and AFDM, ($F(2, 3) = 0.75$, $p < 0.54$, $R^2 = 0.33$).

3.4.1.8 Overall Changes at the Watershed Outlet

Overall, changes at the watershed outlet were not consistent in across measured parameters. For standard water quality measurements, pH and DO decreased while specific conductivity and temperature increased (Table 3.14). For watershed characteristics, canopy cover decreased while AFDM increased (Table 3.14). Measured

nitrate and TDN concentrations and flux showed a decreased over time (Table 3.14).

Finally, Ammonium and DON concentrations and flux increased from the pre-restoration period to the post-restoration period (Table 3.14). Findings are consistent with the hypotheses for nitrate, TDN, and DON (RQ3.1).

*Table 3.14 Changes in all measured parameters from the pre-restoration to the post-restoration period. Statistical significance ($p < 0.05$) is indicated by *.*

Parameter	Change from pre to post	Statistical Significance
pH	Decrease	*
Specific Conductivity	Increased	*
Temperature	Increased	*
Dissolved Oxygen	Decreased	/
Canopy Cover	Decreased	/
AFDM	Increased	*
Nitrate (concentration)	Decreased	*
Total Dissolved Nitrogen (concentration)	Decreased	*
Ammonium (concentration)	Increased	/
Dissolved Organic Nitrogen (concentration)	Increased	*
Baseflow Nitrate (flux)	Decreased	*
Baseflow Total Dissolved Nitrogen (flux)	Decreased	/
Baseflow Ammonium (flux)	Increased	/
Baseflow Dissolved organic Nitrogen (flux)	Increased	*
Discharge	Increased	/

Results from regression analysis on seasonal data indicated that canopy cover and AFDM had no significant effect on nitrogen concentrations or flux during both the pre-restoration and post-restoration periods for all measured parameters besides dissolved organic nitrogen flux and ammonium flux (Table 3.15). Here, regression analysis results indicated that canopy cover and AFDM has a significant effect of total organic nitrogen flux in the post-restoration watershed.

Table 3.15 Seasonal data regression significance at the watershed outlet from the pre-restoration to the post-restoration period. Statistical significance ($p < 0.05$) is indicated by *.

Parameter	Significant Effect of Canopy Cover and AFDM - Pre- Restoration	Significant Effect of Canopy Cover and AFDM - Post-Restoration
Nitrate (concentration)	/	/
Total Dissolved Nitrogen (concentration)	/	/
Ammonium (concentration)	/	/
Dissolved Organic Nitrogen (concentration)	/	/
Nitrate (flux)	/	/
Total Dissolved Nitrogen (flux)	/	/
Ammonium (flux)	/	*
Dissolved Organic Nitrogen (flux)	/	*

3.4.2 (RQ3.2) Is there a dormant season component to organic and inorganic nitrogen flux in the pre- or post-restoration system?

3.4.2.1 Discharge

Dormant season average discharge at the watershed outlet increased from a pre-restoration average of $0.01 \text{ m}^3/\text{s}$ to a post-restoration average of $0.06 \text{ m}^3/\text{s}$ (Table 3.16). Here, pre-restoration discharge variation was lower ($0.009 \text{ m}^3/\text{s}$) than the post-restoration period ($0.206 \text{ m}^3/\text{s}$). Results of a Mann-Whitney test indicate that the increase in discharge was statistically significant ($W = 8, p > 0.008$).

Table 3.16 Descriptive statistics for dormant season discharge (Q) (m^3/s) at site R1 in the pre- and post-restoration period. Data represent the sampling period from January 2016 – March 2021. Q = discharge

Dormant Season		
	Pre- Q (m^3/s)	Post- Q (m^3/s)
Mean	0.010	0.060
Standard Error	0.002	0.014
Median	0.009	0.055
Mode	/	/
Standard Deviation	0.004	0.053
Range	0.009	0.206
Minimum	0.007	0.005
Maximum	0.016	0.211

3.4.2.2 Nitrate

Baseflow nitrate flux measurements were compared before and after restoration during the dormant season. Data indicate that nitrate flux increased from a pre-restoration average of 4.28 mg N/s to a post-restoration average of 9.54 mg N/s (Table 3.17). Average nitrate flux was lower (4.28 mg N/s) and less variable (15.35 mg N/s) in the pre-restoration period (Figure 3.23). Average nitrate flux in the post-restoration period was higher (9.54 mg N/s) and more variable (18.35 mg N/s) than the pre-restoration period (Figure 3.23). Overall, restoration led to an increase in nitrate flux during the dormant season over time (Figure 3.24). Results of a Mann-Whitney test indicate that this increase in nitrate flux, 5.26 mg N/s, was not statistically significant $W = 24$, $p > 0.21$.

Table 3.17 Descriptive statistics for nitrate flux (mg N/s) in the pre- and post-restoration dormant periods

Summary Statistic	Pre-restoration	Post-restoration
Mean	4.28	9.54
Standard Error	2.14	1.95
Median	1.32	11.69
Mode	/	/
Standard Deviation	5.67	6.46
Range	15.35	18.35
Minimum	0.78	0.22
Maximum	16.13	18.56

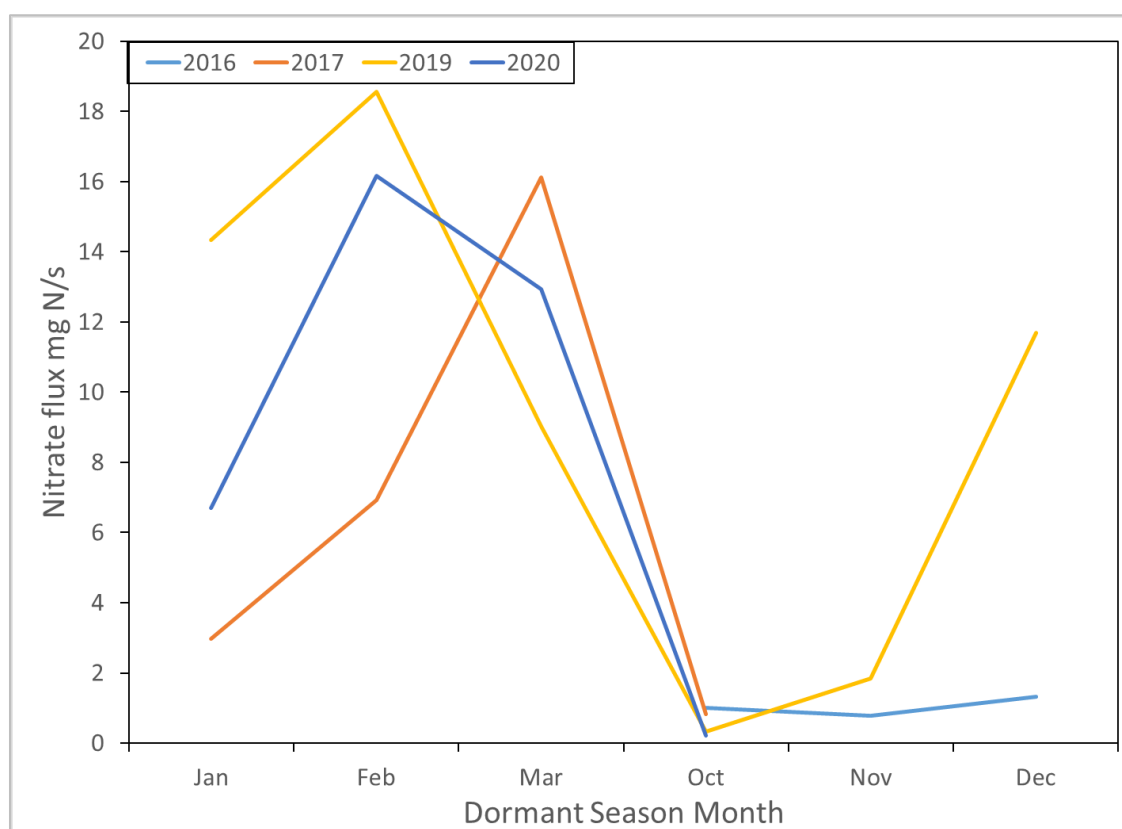


Figure 3.23 Monthly analysis of dormant season (January-March and October-December) nitrate flux (mg N/s) at the R1 site. Data represent the sampling period from 2016 – 2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (yellow) – 2020 (dark blue).

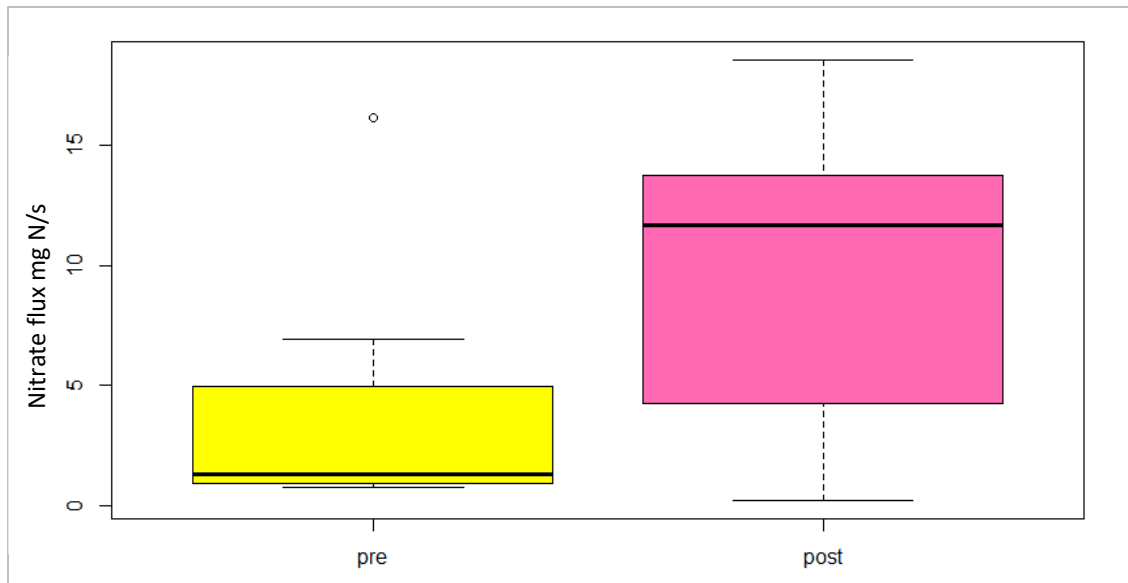


Figure 3.24 Box and whiskers plot for nitrate flux (mg N/s) in the pre- and post-restoration dormant (January-March and October-December) periods at the R1 site. Data represent the sampling period from 2016-2021.

Data were compared using a seasonal index to assess the impact of seasonal months on nitrogen flux before and after restoration during the dormant season. Seasonal indices for the pre-restoration dormant season indicate that nitrate flux was highest in March (377 %) followed by February (162 %) (Table 3.18). Seasonal nitrate flux pre-restoration was lowest in November (18 %) followed by December (31 %) (Table 3.18). Seasonal indices for the post-restoration dormant season show that nitrate flux was highest in February (364 %) followed by December (261 %) (Table 3.18). Seasonal nitrate flux post-restoration was lowest in October (6 %) followed by November (19 %) (Table 3.18). Overall, pre-restoration had the highest seasonal indices represented by February (377 %) while post-restoration had the lowest seasonal indices represented by October (6 %). During both the pre-and post-restoration periods, the majority of baseflow

nitrate flux happened during the period encompassing January through March with a slight shift to include the month of December in the post-restoration period.

Table 3.18 Seasonal nitrate flux indices for dormant season (January-March and October-December) months in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Jan	0.70	2.20
Feb	1.62	3.64
Mar	3.77	2.30
Oct	0.43	0.06
Nov	0.18	0.19
Dec	0.31	2.61

3.4.2.3 Ammonium

Ammonium flux measurements were compared before and after restoration. Data indicate that ammonium flux increased from a pre-restoration average of 0.06 mg N/s to a post-restoration average of 2.61 mg N/s (Table 3.19). Average ammonium flux was lowest (0.06 mg N/s) and least variable (0.15 mg N/s) in the pre-restoration period (Figure 3.25). Average ammonium flux in the post-restoration period was higher (2.61 mg N/s) and more variable (6.13 mg N/s) than the pre-restoration period (Figure 3.25). This increase in ammonium flux, 2.55 mg N/s, was statistically significant $W = 12.5$, $p < 0.02$. Overall, restoration led to an increase in ammonium flux over time (Figure 3.26).

Table 3.19 Descriptive statistics for ammonium flux (mg N/s) in the pre- and post-restoration dormant (January-March and October-December) periods at the R1 site. Data represent the sampling period from 2016-2021.

Summary Statistic	Pre-restoration	Post-restoration
Mean	0.06	2.61
Standard Error	0.02	0.68
Median	0.05	2.75
Mode	/	/
Standard Deviation	0.05	2.24
Range	0.15	6.13
Minimum	0.00	0.00
Maximum	0.15	6.13

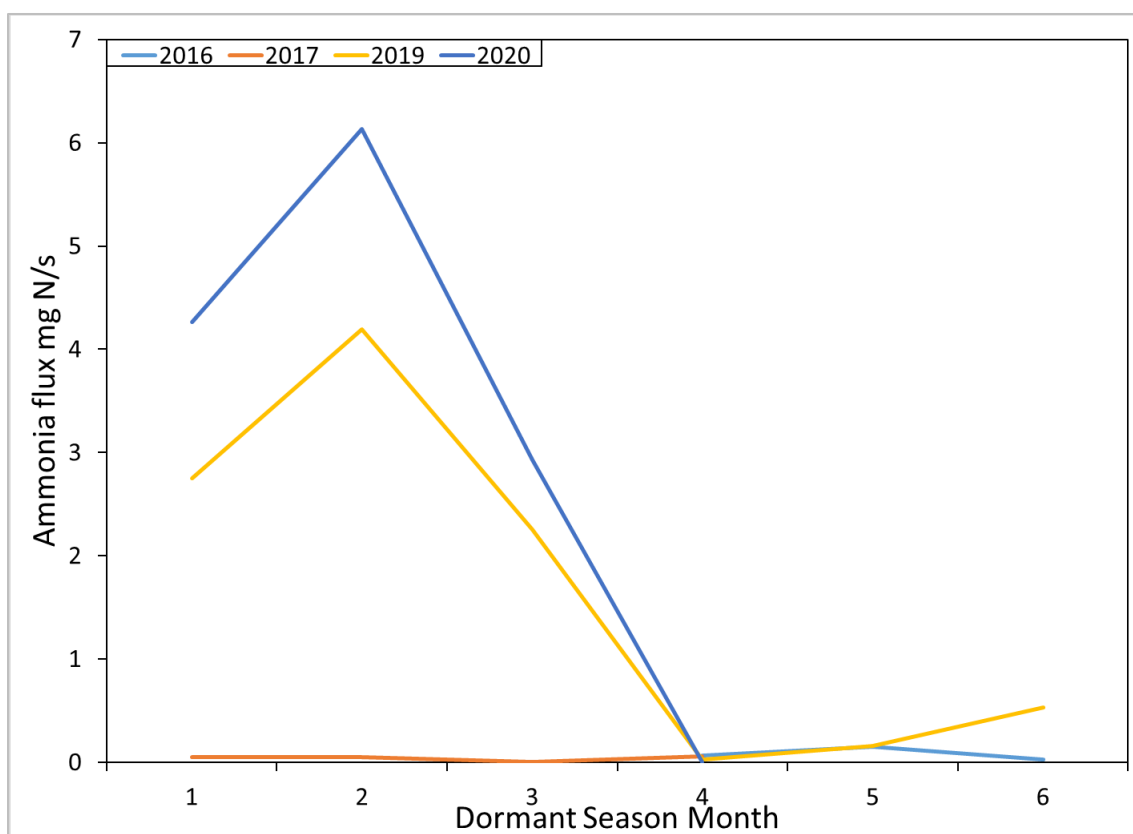


Figure 3.25 Monthly analysis of dormant (January-March and October-December) season ammonium flux (mg N/s) at the R1 site. Data represent the sampling period from 2016-2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (yellow) – 2020 (dark blue).

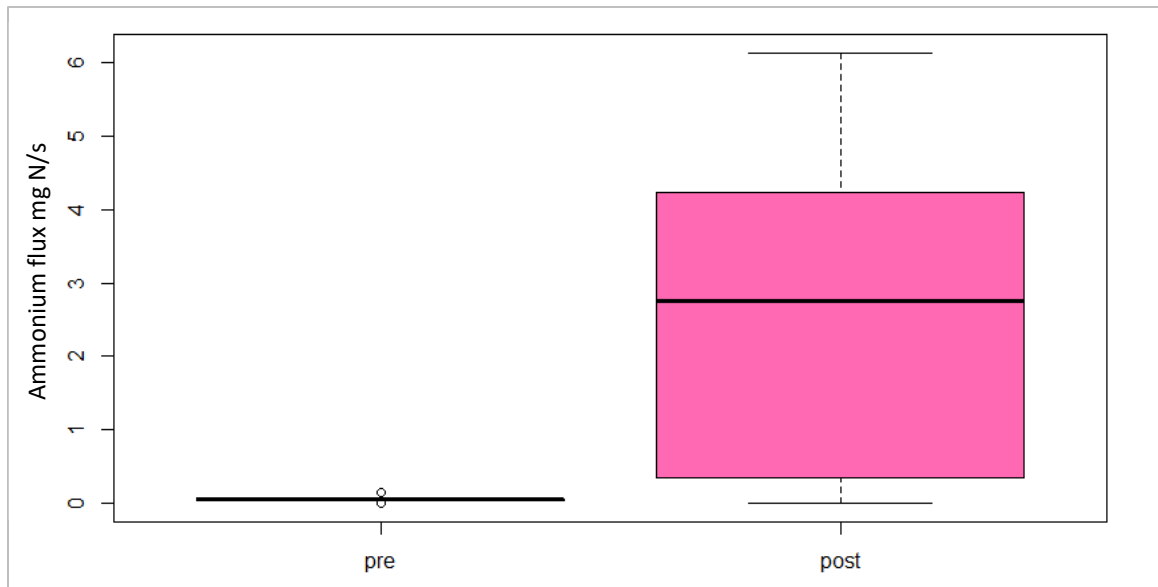


Figure 3.26 Box and whiskers plot for ammonium flux (mg/s) in the pre- and post-restoration dormant (January-March and October-December) periods at the R1 site. Data represent the sampling period from 2016-2021.

Seasonal indices for the pre-restoration dormant season indicate that ammonium flux was highest in November (264 %) followed by October (208 %) (Table 3.20). Seasonal ammonium flux pre-restoration was lowest in March (0 %, undetectable) followed by December (54 %) (Table 3.20). Seasonal indices for the post-restoration dormant season show that ammonium flux was highest in February (395 %) followed by January (268 %) (Table 3.20). Seasonal ammonium flux post-restoration was lowest in October (1 %) followed by November (6 %) (Table 3.20). Overall, post-restoration had the highest seasonal indices represented by February (395 %) while pre-restoration had the third highest seasonal indices represented by November (264 %). During the pre-restoration period, the majority of ammonium flux was moved during the months of October and November while, in the post-restoration period, the majority of ammonium flux happened during the period encompassing December through February.

Table 3.20 Seasonal ammonium flux indices for dormant (January-March and October-December) season months in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Jan	0.82	2.68
Feb	0.92	3.95
Mar	0.00	1.98
Oct	2.08	0.01
Nov	2.64	0.06
Dec	0.54	2.31

3.4.2.4 Total Dissolved Nitrogen

Total dissolved nitrogen flux measurements were compared before and after restoration during the dormant season. Data indicate that TDN flux during the dormant season increased from a pre-restoration average of 5.50 mg N/s to a post-restoration average of 27.50 mg N/s (Table 3.21). Average total dissolved nitrogen flux was lower (5.50 mg N/s) and less variable (9.08 mg N/s) in the pre-restoration period (Figure 3.24). Average total dissolved nitrogen flux in the post-restoration period was higher (27.50 mg N/s) and more variable (81.61 mg N/s) than the pre-restoration period (Figure 3.24). This increase in total dissolved nitrogen flux, 22 mg N/s, was statistically significant, $W = 10$, $p < 0.008$. Overall, restoration led to an increase in total dissolved nitrogen flux during the dormant season over time (Figure 3.28).

Table 3.21 Descriptive statistics for total dissolved nitrogen flux (mg N/s) in the pre- and post-restoration dormant (January-March and October-December) periods at the R1 site. Data represent the sampling period from 2016-2021.

Summary Statistic	Pre-restoration	Post-restoration
Mean	5.50	27.50
Standard Error	1.27	7.21
Median	4.41	25.62
Mode	/	/
Standard Deviation	3.35	23.90
Range	9.08	81.61
Minimum	2.15	2.46
Maximum	11.24	84.07

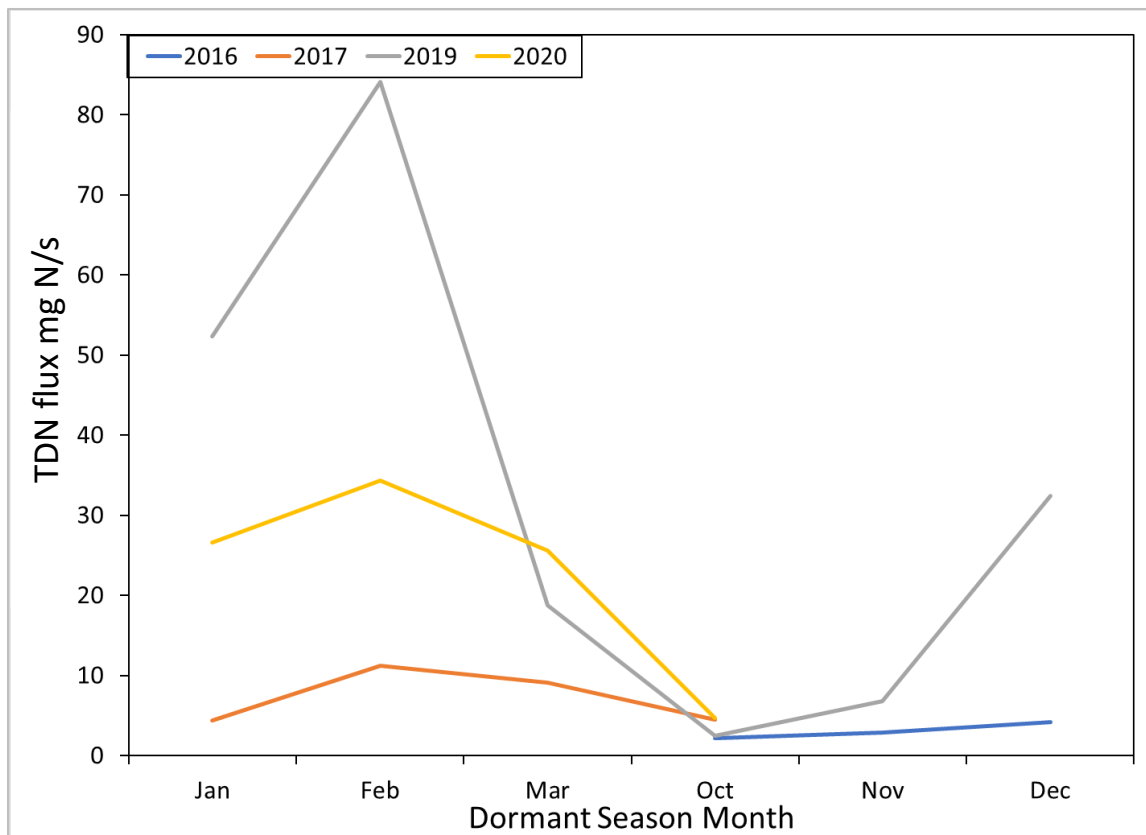


Figure 3.27 Monthly analysis of dormant (January-March and October-December) season total dissolved nitrogen flux (mg N/s) at the R1 site. Data represent the sampling period from 2016-2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (grey) – 2020 (yellow).

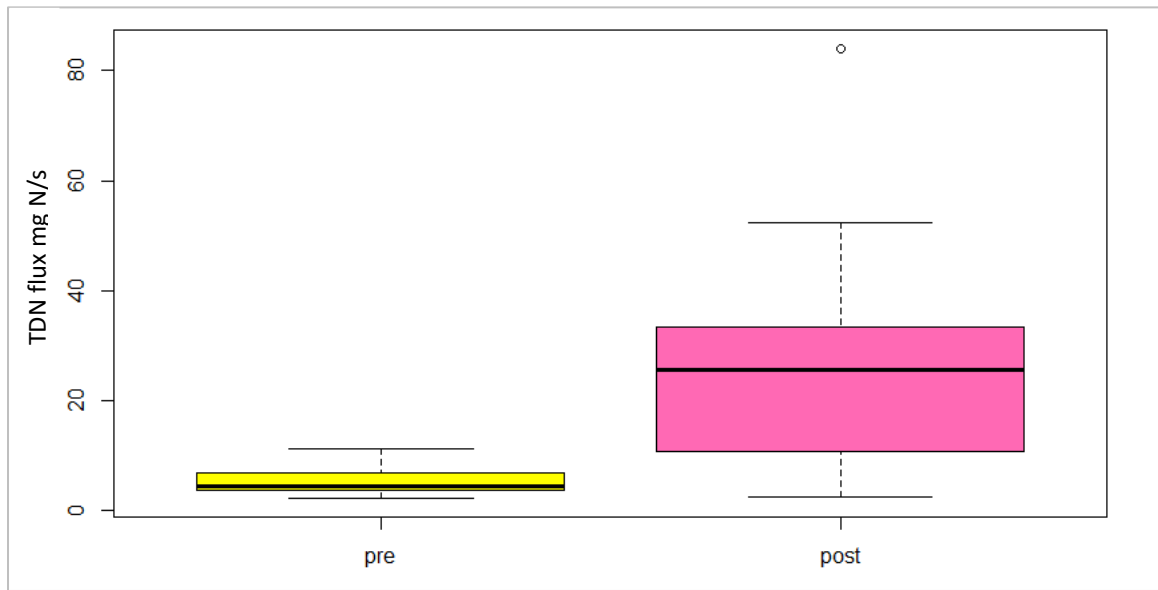


Figure 3.28 Box and whiskers plot for total dissolved nitrogen flux (mg N/s) in the pre- and post-restoration dormant (January-March and October-December) periods at the R1 site. Data represent the sampling period from 2016-2021.

Seasonal indices for the pre-restoration dormant season indicate that total dissolved nitrogen flux was highest in December (77 %) (Table 3.22). Seasonal TDN flux pre-restoration was lowest in October (39 %) (Table 3.22). Seasonal indices for the post-restoration dormant season show that TDN flux was highest in February (41 %) (Table 3.22). Seasonal TDN flux post-restoration was lowest in January and October (16 %) (Table 3.22). Overall, pre-restoration had the highest seasonal indices represented by December (77 %) and pre-restoration had the lowest seasonal indices represented by January and October (16 %). Findings indicate that during the pre-restoration period, the majority of TDN flux happened during the period encompassing November through December. In the post-restoration period, the majority of TDN flux was transported during the February through March period.

Table 3.22 Seasonal total dissolved nitrogen flux indices for dormant (January-March and October-December) season months in the pre- and post-restoration periods at the R1 site. / indicates months in which no data were collected for the period. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Jan	/	0.16
Feb	/	0.41
Mar	/	0.33
Oct	0.39	0.16
Nov	0.53	/
Dec	0.77	/

3.4.2.5 Dissolved organic Nitrogen

Dissolved organic nitrogen flux measurements were compared before and after restoration during the dormant season. Data indicate that DON flux during the dormant season increased from a pre-restoration average of 2.17 mg N/s to a post-restoration average of 15.73 mg N/s (Table 3.23). Average dissolved organic nitrogen flux was lower (2.17 mg N/s) and less variable (4.27 mg N/s) in the pre-restoration period (Figure 3.29). Average dissolved organic nitrogen flux in the post-restoration period was higher (15.73 mg N/s) and more variable (61.32 mg N/s) than the pre-restoration period (Figure 3.29). This increase in total organic nitrogen flux, 13.56 mg N/s, was statistically significant, $W = 9.5$, $p < 0.01$. Overall, restoration led to an increase in dissolved organic nitrogen flux during the dormant season over time (Figure 3.30).

Table 3.23 Descriptive statistics for dissolved organic nitrogen flux (mg N/s) in the pre- and post-restoration dormant (January-March and October-December) periods at the R1 site. Data represent the sampling period from 2016-2021.

Summary Statistic	Pre-restoration	Post-restoration
Mean	2.17	15.73
Standard Error	0.57	5.46
Median	1.98	9.75
Mode	/	/
Standard Deviation	1.50	18.10
Range	4.27	61.32
Minimum	0.00	0.00
Maximum	4.27	61.32

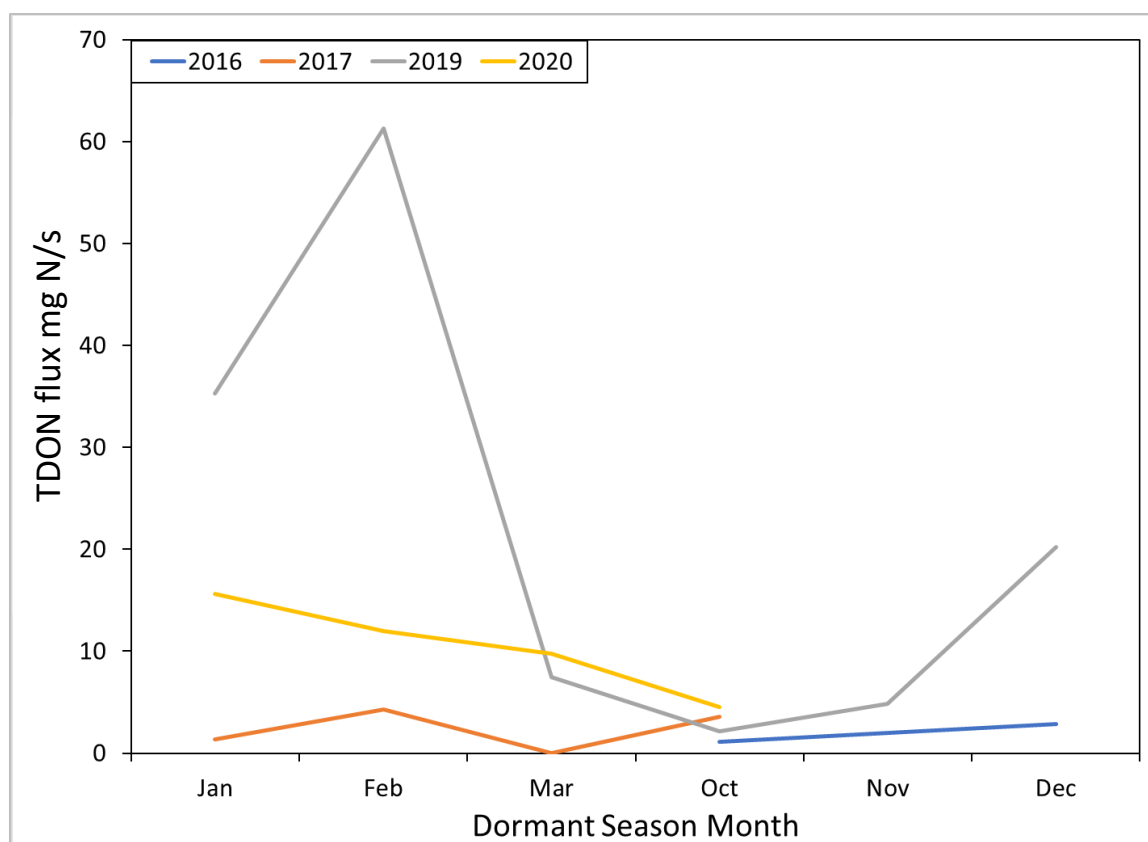


Figure 3.29 Monthly analysis of dormant (January-March and October-December) season dissolved organic nitrogen flux (mg N/s) at the R1 site. Data represent the sampling period from 2016-2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (grey) – 2020 (yellow).

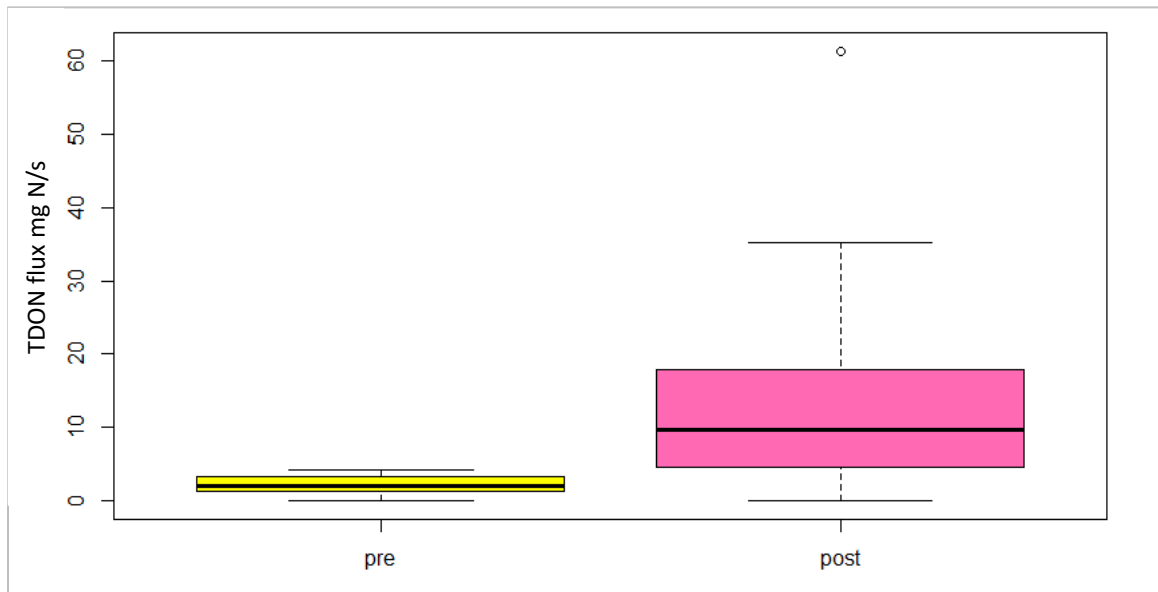


Figure 3.30 Box and whiskers plot for total organic nitrogen flux (mg N/s) in the pre- and post-restoration dormant (January-March and October-December) periods at the R1 site. Data represent the sampling period from 2016-2021.

Seasonal indices for the pre-restoration dormant season indicate that dissolved organic nitrogen flux was highest in December (132 %) (Table 3.24). Seasonal DON flux pre-restoration was lowest in October (50 %) (Table 3.24). Seasonal indices for the post-restoration dormant season show that dissolved organic nitrogen flux was highest in February (27 %) (Table 3.24). Seasonal DON flux post-restoration was lowest in March (0 % undetectable) (Table 3.24). Overall, pre-restoration had the highest seasonal indices represented by December (132 %) while post-restoration had the lowest seasonal indices represented by March (0 % undetectable). Findings indicate that during the pre-restoration periods, the majority of total organic nitrogen flux happened during the period encompassing November through December. In the post-restoration period, the majority of DON was moved in the month of March.

Table 3.24 Seasonal total organic nitrogen flux indices for dormant (January-March and October-December) season months in the pre- and post-restoration periods at the R1 site. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Jan	/	0.09
Feb	/	0.27
Mar	/	0.00
Oct	0.50	0.23
Nov	0.91	/
Dec	1.32	/

3.4.2.6 Dormant Season Changes at the Watershed Outlet

Overall, measured flux at the watershed outlet during the dormant season all changed similarly from the pre-restoration to the post-restoration period. Here, all parameters were seen to increase over time (Table 3.25). Moreover, all parameters, besides nitrate, has statistically significant increased from their pre-restoration levels to post-restoration levels (Table 3.25). These findings for total inorganic nitrogen and total dissolved nitrogen are not consistent with the hypotheses (RQ3.2) which stated that restoration would lead to a decreased in these parameters during the dormant season. However, these findings for dissolved organic nitrogen are consistent with the hypotheses that restoration would lead to an increase during the dormant season (RQ3.2).

Table 3.25 Changes in all measured parameters at the watershed outlet from the pre-restoration to the post-restoration period during the dormant (January-March and October-December) season. Statistical significance ($p < 0.05$) is indicated by *. Data represent the sampling period from 2016-2021

Parameter	Change from pre to post-restoration	Statistical Significance
Nitrate (flux)	Increased	/
Total Dissolved Nitrogen (flux)	Increased	*
Ammonium (flux)	Increased	*
Dissolved organic Nitrogen (flux)	Increased	*
Discharge (Q)	Increased	*

3.4.3 (RQ3.3) Is there a growing season component to organic and inorganic nitrogen flux in the pre- or post-restoration system?

3.4.3.1 Discharge

Growing season average discharge at the watershed outlet decreased from a pre-restoration average of $0.04 \text{ m}^3/\text{s}$ to a post-restoration average of $0.03 \text{ m}^3/\text{s}$ (Table 3.26). Here, pre-restoration discharge variation was lower ($0.09 \text{ m}^3/\text{s}$) than the post-restoration period ($0.16 \text{ m}^3/\text{s}$). Results of a Mann-Whitney test indicate that the decrease in discharge was statistically significant ($W = 256, p < 0.01$).

Table 3.26 Descriptive statistics for growing season discharge (m^3/s) at site R1 in the pre- and post-restoration period. Data represent the sampling period from 2016-2021.

Growing Season		
	Pre -Q (m^3/s)	Post-Q (m^3/s)
Mean	0.045	0.032
Standard Error	0.005	0.008
Median	0.042	0.018
Mode	0.058	0.016
Standard Deviation	0.021	0.037
Range	0.095	0.164
Minimum	0.014	0.005
Maximum	0.109	0.169

3.4.3.2 Nitrate

Nitrate flux measurements were compared before and after restoration. Data indicate that nitrate flux decreased from a pre-restoration average of 13.50 mg N/s to a post-restoration average of 2.01 mg N/s (Table 3.27). Average nitrate flux was highest (13.50 mg N/s) and most variable (35 mg N/s) in the pre-restoration period (Figure 3.31). Average nitrate flux in the post-restoration period was lower (2.01 mg N/s) and less variable (9.63 mg N/s) than the pre-restoration period (Figure 3.31). This decrease in nitrate flux, 11.49 mg N/s, was statistically significant, $W = 91$, $p < 0.002$. Overall, restoration led to a decrease in nitrate flux over time (Figure 3.32).

Table 3.27 Descriptive statistics for nitrate flux (mg N/s) in the pre- and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Summary Statistic	Pre-restoration	Post-restoration
Mean	13.50	2.01
Standard Error	3.18	1.14
Median	10.64	0.37
Mode	/	0.00
Standard Deviation	10.54	3.43
Range	35.00	9.63
Minimum	0.56	0.00
Maximum	35.57	9.63

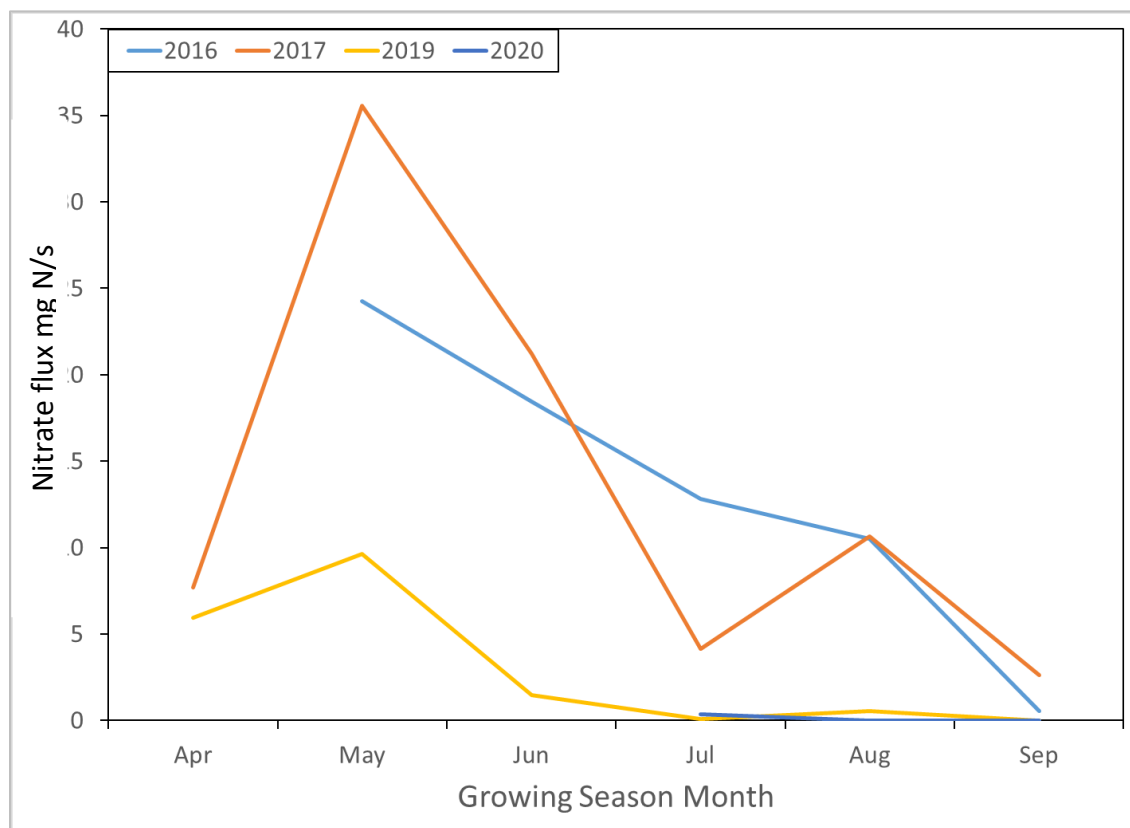


Figure 3.31 Monthly analysis of growing (April-September) season nitrate flux (mg N/s) at the R1 site. Data represent the sampling period from 2016-2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (yellow) – 2020 (dark blue).

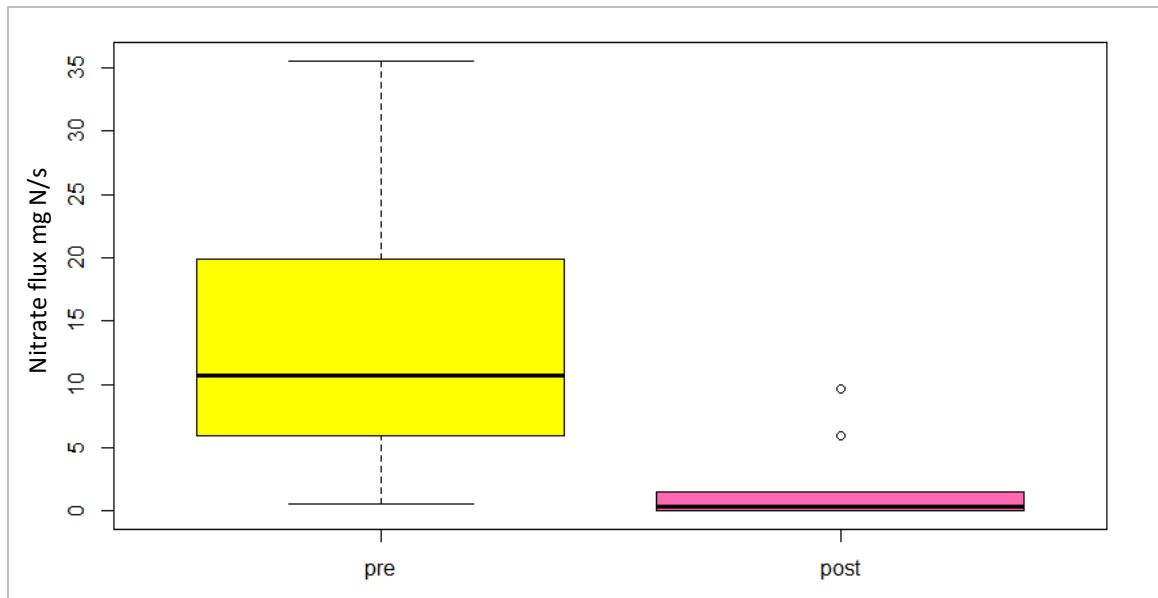


Figure 3.32 Box and whiskers plot for nitrate flux (mg N/s) in the pre- and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Seasonal indices for the pre-restoration growing season indicate that nitrate flux was highest in May (443 %) followed by June (294 %) (Table 3.28). Seasonal nitrate flux pre-restoration was lowest in September (24 %) followed by April (57 %) (Table 3.28). Seasonal indices for the post-restoration growing season show that nitrate flux was highest in May (479 %) followed by April (296 %) (Table 3.28). Seasonal nitrate flux post-restoration was lowest in September (0 %) (Table 3.28). Overall, post-restoration had the highest seasonal indices represented by May (479 %) while post-restoration had the second highest seasonal indices represented by May (443 %). Findings indicate that during both the pre-and post-restoration period growing seasons, the majority of nitrate flux happened during the month of May.

Table 3.28 Seasonal nitrate flux indices for growing (April-September) season months in the pre- and post-restoration periods at the R1 site. / indicates months in which no data were collected for the period. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Apr	0.57	2.96
May	4.43	4.79
Jun	2.94	0.74
Jul	1.25	0.23
Aug	1.57	0.28
Sep	0.24	0.00

3.4.3.3 Ammonium

Ammonium flux measurements were compared before and after restoration. Data indicate that ammonium flux increased from a pre-restoration average of 0.48 mg N/s to a post-restoration average of 1.80 mg N/s (Table 3.29). Average ammonium flux was lower (0.48 mg N/s) and less variable (1.75 mg N/s) in the pre-restoration period when compared to post-restoration (Figure 3.33). Average ammonium flux in the post-restoration period was higher (1.80 mg N/s) and more variable (10.75 mg N/s) than the pre-restoration period (Figure 3.33). This increase in ammonium flux, 1.32 mg N/s, was not statistically significant, $W = 56.5$, $p > 0.62$. Overall, restoration led to an increase in ammonium flux over time (Figure 3.34).

Table 3.29 Descriptive statistics for ammonium flux (mg N/s) in the pre- and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Summary Statistic	Pre-restoration	Post-restoration
Mean	0.48	1.80
Standard Error	0.15	1.20
Median	0.34	0.08
Mode	/	0.00
Standard Deviation	0.51	3.59
Range	1.75	10.75
Minimum	0.00	0.00
Maximum	0.15	6.13

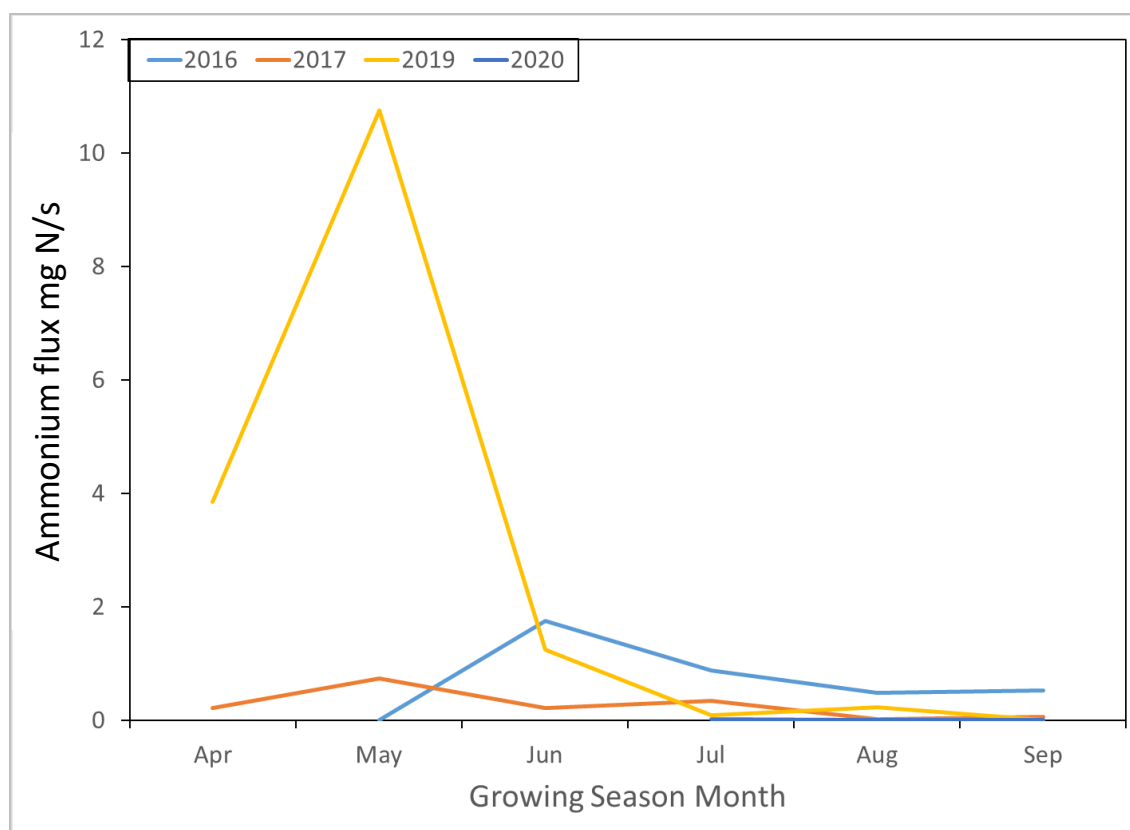


Figure 3.33 Monthly analysis of growing (April-September) season ammonium flux (mg N/s) at the R1 site. Data represent the sampling period from 2016-2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (yellow) – 2020 (dark blue).

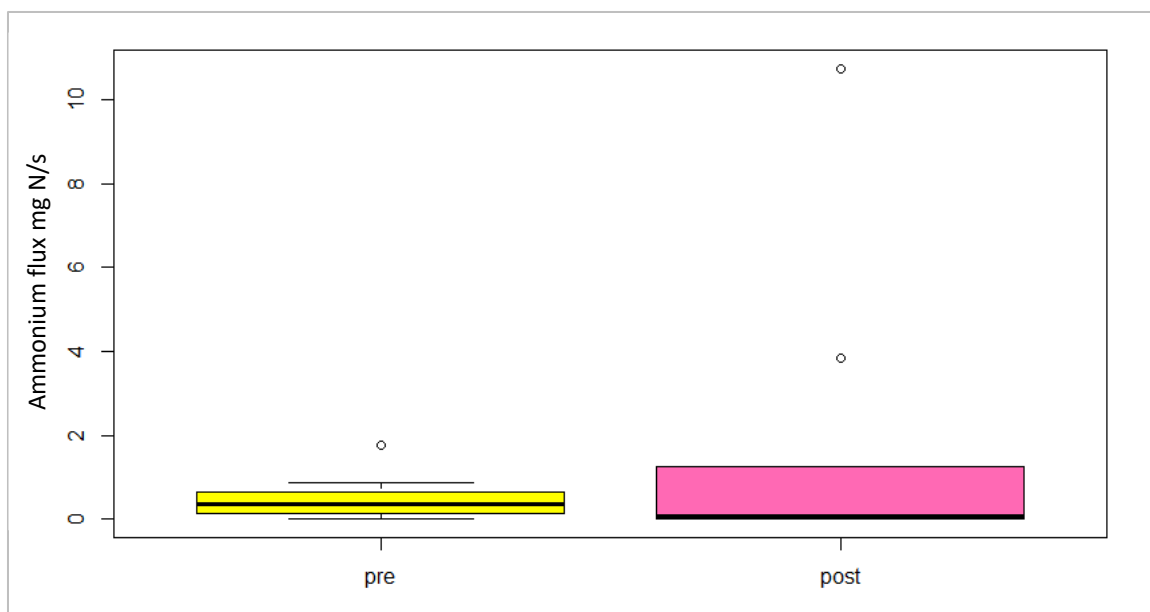


Figure 3.34 Box and whiskers plot for ammonium flux (mg N/s) in the pre- and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Seasonal indices for the pre-restoration growing season indicate that ammonium flux was highest in June (414 %) (Table 3.30). Seasonal ammonium flux pre-restoration was lowest in April (46 %) followed by August (107 %) (Table 3.30). Seasonal indices for the post-restoration growing season show that ammonium flux was highest in May (598 %) followed by April (214 %) (Table 3.30). Seasonal ammonium flux post-restoration was lowest in September (0 %, undetectable) (Table 3.30). Overall, post-restoration had the highest seasonal indices represented by May (598 %) while post-restoration had the second highest seasonal indices represented by June (414 %). Findings indicate that during the pre-restoration period, the majority of ammonium flux occurred during the months of June and July while during the post-restoration period, the majority of ammonium flux happened during the months of April and May.

Table 3.30 Seasonal ammonium flux indices for growing (April-September) season months in the pre- and post-restoration periods at the R1 site. / indicates months in which no data were collected for the period. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Apr	0.46	2.14
May	1.54	5.98
Jun	4.14	0.69
Jul	2.58	0.06
Aug	1.07	0.12
Sep	1.22	0.00

3.4.3.4 Total Dissolved Nitrogen

Total dissolved nitrogen measurements were compared before and after restoration during the growing season. Data indicate that TDN increased from a pre-restoration average of 10.21 mg N/s to a post-restoration average of 10.98 mg N/s (Table 3.31). Average total nitrogen was lowest (10.21 mg N/s) and less variable (31.05 mg N/s) in the pre-restoration period (Figure 3.35). Average total nitrogen in the post-restoration period was highest (10.98 mg N/s) and most variable (40.77 mg N/s) than the pre-restoration period (Figure 3.35). This increase in total dissolved nitrogen, 0.77 mg N/s, was not statistically significant, $W = 45$, $p > 0.76$. Overall, restoration led to a increase in growing season total dissolved nitrogen over time (Figure 3.36).

Table 3.31 Descriptive statistics for total dissolved nitrogen flux (mg N/s) in the pre-, during, and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Summary Statistic	Pre-restoration	Post-restoration
Mean	10.21	10.98
Standard Error	3.39	4.32
Median	6.39	5.15
Mode	0.00	/
Standard Deviation	11.23	12.95
Range	31.05	40.77
Minimum	0.00	1.61
Maximum	31.05	42.38

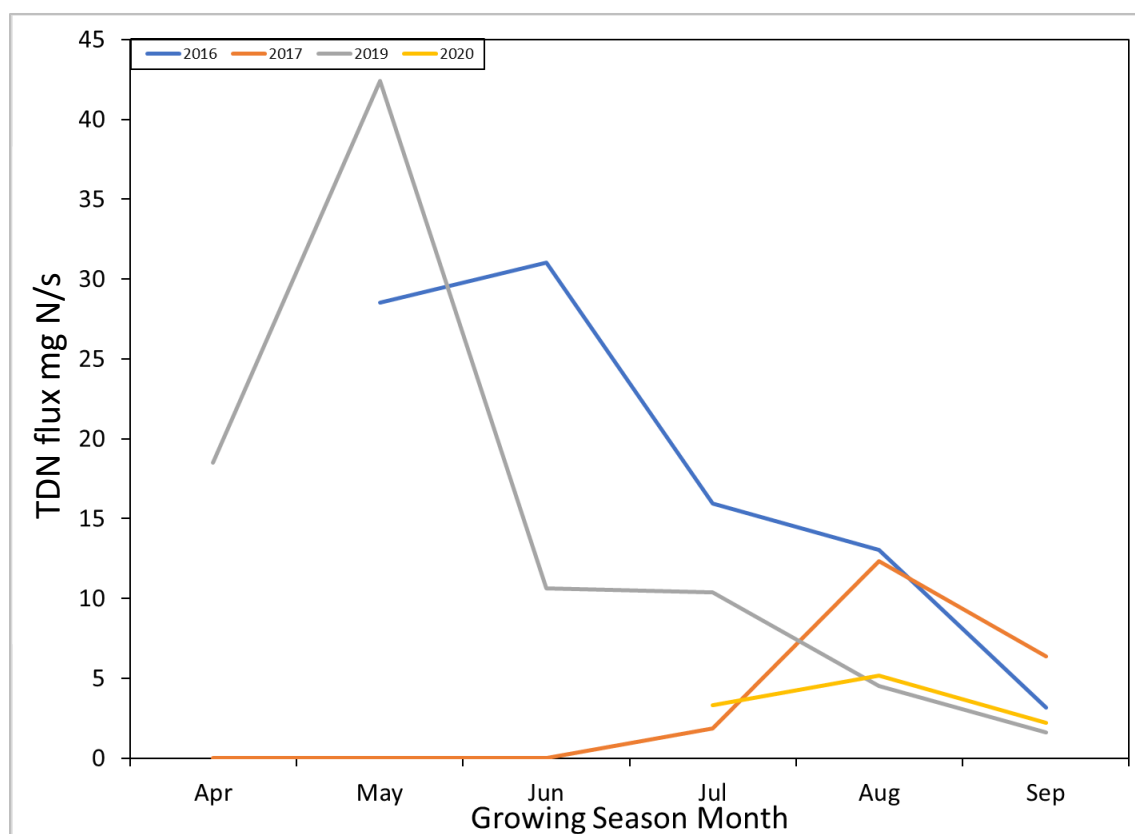


Figure 3.35 Monthly analysis of growing (April-September) season total dissolved nitrogen flux (mg N/s) at the R1 site. Data represent the sampling period from 2016-2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (grey) – 2020 (yellow).

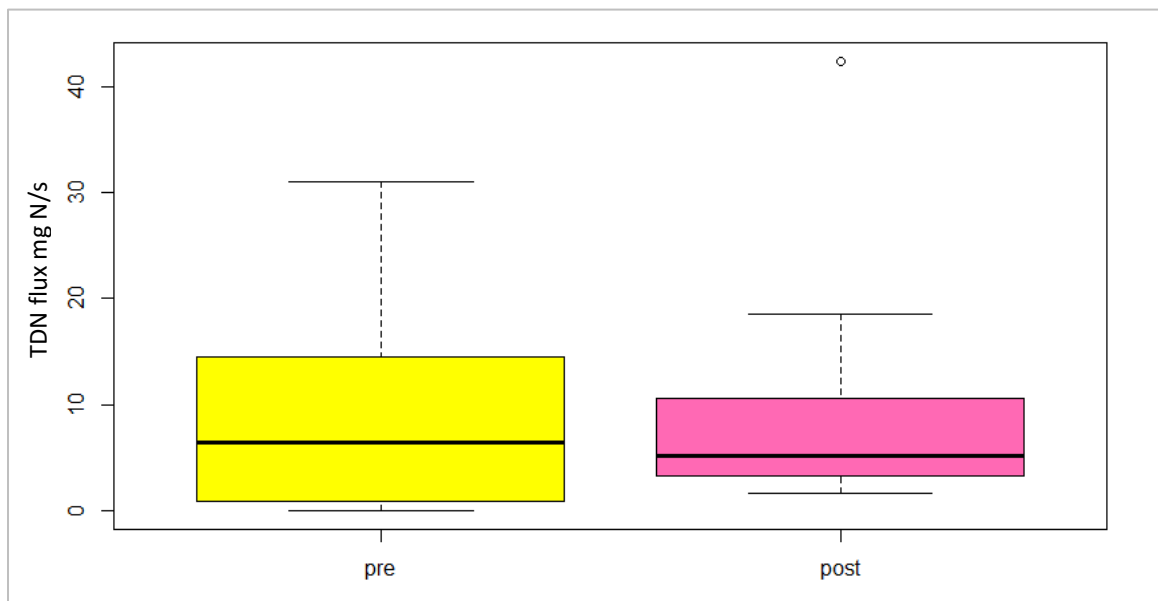


Figure 3.36 Box and whiskers plot for total dissolved nitrogen flux (mg N/s) in the pre- and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Seasonal indices for the pre-restoration growing season indicate that total dissolved nitrogen flux was highest in June (304 %) followed by May (279 %) (Table 3.32). Seasonal total dissolved nitrogen flux pre-restoration was lowest in September (31 %) followed by August (128 %) (Table 3.32). Seasonal indices for the post-restoration growing season show that total dissolved nitrogen flux was highest in May (386 %) (Table 3.32). Seasonal total dissolved nitrogen flux post-restoration was lowest in September (15 %) (Table 3.32). Overall, pre-restoration had the highest seasonal indices represented by June (304 %) while post-restoration had the highest seasonal indices represented by May (386 %). Findings indicate that the majority of total dissolved nitrogen flux happened during the period encompassing May through August in the pre-restoration period and the months of April through June in the post-restoration period.

Table 3.32 Seasonal total dissolved nitrogen flux indices for growing (April-September) season months in the pre- and post-restoration periods at the R1 site. / indicates months in which no data were collected for the period. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Apr	/	1.69
May	2.79	3.86
Jun	3.04	0.97
Jul	1.56	0.95
Aug	1.28	0.41
Sep	0.31	0.15

3.4.3.5 Dissolved organic Nitrogen

Dissolved organic nitrogen measurements were compared before and after restoration during the growing season. Data indicate that dissolved organic nitrogen increased from a pre-restoration average of 2.62 mg N/s to a post-restoration average of 87.17 mg N/s (Table 3.33). Average DON was lower (2.62 mg N/s) and less variable (11.18 mg N/s) in the pre-restoration period when compared to post-restoration (Figure 3.37). Average DON in the post-restoration period was higher (7.17 mg N/s) and more variable (20.40 mg N/s) than the pre-restoration period (Figure 3.37). This increase in dissolved organic nitrogen, 4.55 mg N/s, was statistically significant, $W = 22$, $p < 0.04$. Overall, restoration led to an increase in dissolved organic nitrogen over time (Figure 3.38).

Table 3.33 Descriptive statistics for total organic nitrogen flux (mg N/s) in the pre- and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Summary Statistic	Pre-restoration	Post-restoration
Mean	2.62	7.17
Standard Error	0.99	2.11
Median	2.04	5.15
Mode	0.00	/
Standard Deviation	3.28	6.34
Range	11.18	20.40
Minimum	0.00	1.61
Maximum	11.18	22.01

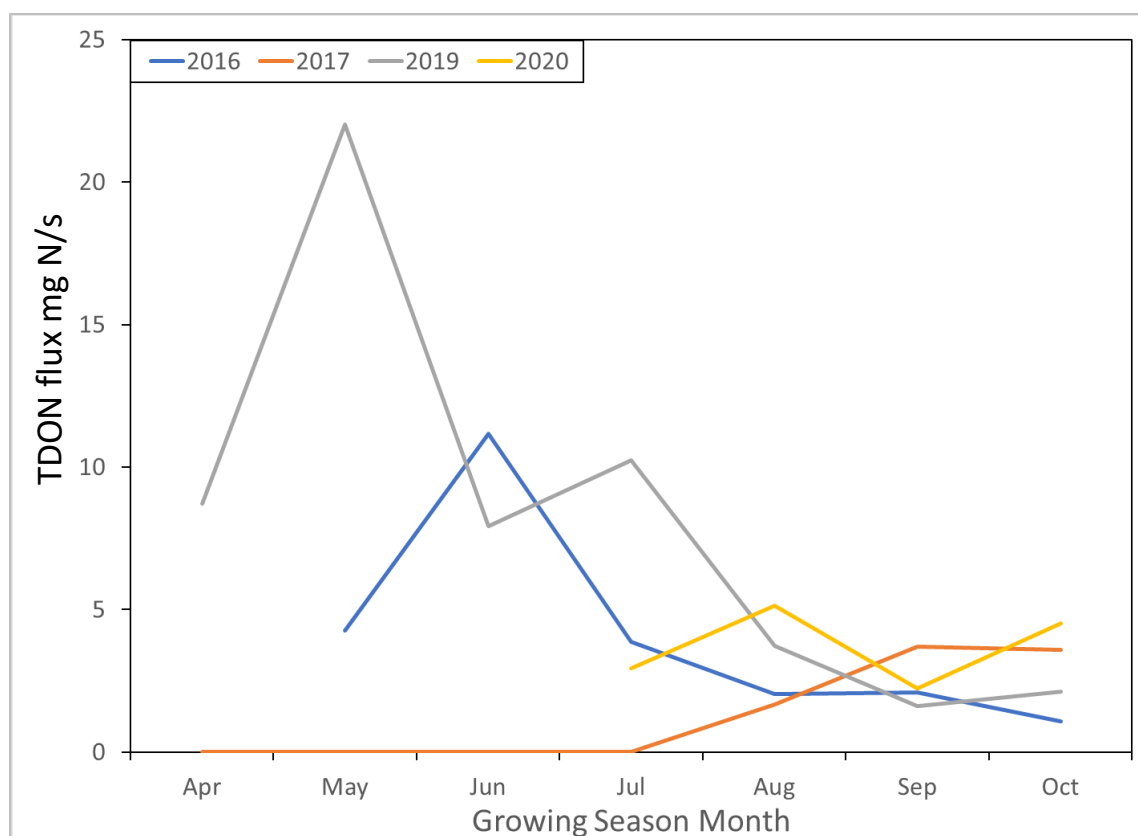


Figure 3.37 Monthly analysis of growing (April-September) season dissolved organic nitrogen flux (mg N/s) at the R1 site. Data represent the sampling period from 2016-2021. Pre-restoration includes 2016 (light blue) – 2017 (orange). Post-restoration includes 2019 (grey) – 2020 (yellow).

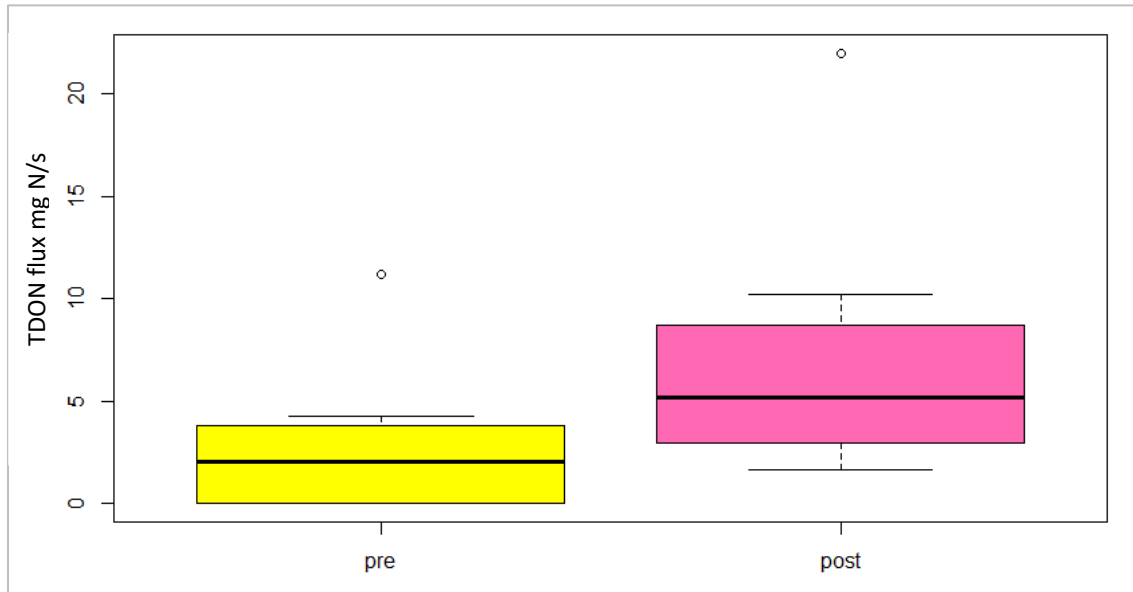


Figure 3.38 Box and whiskers plot for dissolved organic nitrogen flux (mg N/s) in the pre- and post-restoration growing (April-September) periods at the R1 site. Data represent the sampling period from 2016-2021.

Seasonal indices for the pre-restoration growing season indicate that dissolved organic nitrogen flux was highest in June (427 %) followed by May (163 %) (Table 3.34). Seasonal DON flux pre-restoration was lowest in August (78 %) followed by September (80 %) (Table 3.34). Seasonal indices for the post-restoration growing season show that DON flux was highest in May (307 %) followed by April (122 %) (Table 3.34). Seasonal DON flux post-restoration was lowest in September (22 %) (Table 3.34). Overall, pre-restoration had the highest seasonal indices represented by June (427 %) while post-restoration had the second highest seasonal indices represented by May (307 %). Findings indicate that during the pre-restoration period, the majority of DON flux happened during the period encompassing May through July. In the post-restoration period, the majority of DON flux occurred during the months of April through June.

Table 3.34 Seasonal dissolved organic nitrogen flux indices for growing (April-September) season months in the pre- and post-restoration periods at the R1 site. / indicates months in which no data were collected for the period. Data represent the sampling period from 2016-2021.

Month	Pre- Restoration Seasonal Indices	Post- Restoration Seasonal Indices
Apr	/	1.22
May	1.63	3.07
Jun	4.27	1.11
Jul	1.48	1.43
Aug	0.78	0.52
Sep	0.80	0.22

3.4.3.6 Growing Season Changes at the Watershed Outlet

Overall, measured flux at the watershed outlet during the growing season all changed similarly from the pre-restoration to the post-restoration period. Here, nitrate was seen to decrease over time while ammonium increased (Table 3.35). Moreover, nitrate was the only parameter to have a statistically significant decrease from pre-restoration levels to post-restoration levels (Table 3.35). These findings for nitrate are consistent with the hypotheses (RQ3.3) which stated that restoration would lead to a decreased flux during the growing season. Moreover, these findings for dissolved organic nitrogen were also consistent with the hypotheses that restoration would lead to an increase during the growing season (RQ3.2). However, findings for ammonium and TDN were not consistent with hypotheses that flux would decrease over time (Table 3.35).

*Table 3.35 Changes in all measured parameters at the watershed outlet from the pre-restoration to the post-restoration period during the growing (April-September) season. Statistical significance ($p < 0.05$) is indicated by *. Data represent the sampling period from 2016-2021.*

Parameter	Change from pre to post	Statistical Significance
Nitrate (baseflow flux)	Decreased	*
TDN (baseflow flux)	Increased	/
Ammonium (baseflow flux)	Increased	/
DON (baseflow flux)	Increased	*
Discharge (Q)	Decreased	*

3.5 Discussion

3.5.1 RQ3.1

Trends in stream chemistry suggest that restoration of streams at the whole watershed-level, where streams are connected to their surrounding floodplains, has the potential to decrease the concentrations and flux of inorganic nitrate and TDN. This is indicative of increased denitrification, and possibly decreased nitrate inputs, especially as streams in the post-restoration period flow through channels allowing for floodplain inundation and no longer flow through channels with high, incised stream banks (Pinay et al., 2000; Hasselquist et al., 2017; Mayer et al., 2022). Moreover, similar to findings by Zarnetske et al. (2011a), nitrate concentration and flux decreases may be attributed to increases in residence times. As for TDN, overall decreases may be attributed to the significant decrease in nitrate as nitrate makes up a significant portion of TDN.

Restoration decreased the TDN and inorganic nitrate concentration flux indicating that the watershed is retaining/transforming more N. For instance, the bacteria responsible for denitrification are oxygen sensitive and may have become more active as dissolved oxygen levels decreased in the post-restoration period (Reisinger et al., 2016). Moreover, decreases may also be attributed to dilution driven by increased discharge (Zhou et al.,

2017). Overall, findings provide evidence that the restoration approach used was successful at changing the TDN and inorganic nitrate concentration and flux at the watershed outlet.

Trends suggest that restoration has the potential to increase ammonium and DON concentration and flux. Restoration increased the DON concentration and flux slightly possibly providing evidence of alteration to organic nitrogen sources such as a potential increase in organic biomass (i.e.: increased instream flora and fauna) which may contribute to organic N as they complete their lifecycles and the in-washing of plant and animal wastes from newly reconnected floodplains. Furthermore, restoration led to increases in ammonium concentration and flux indicating that changes within the watershed may not be favorable for nitrification. For instance, bacteria that perform nitrification are temperature, pH, and oxygen sensitive and may not have adapted to increased temperature and decreased pH and dissolved oxygen levels within the watershed.

Multiple regression analyses were found to be insignificant, indicating that altered watershed characteristics (e.g.: changes to canopy cover and algal biomass) do not play an important role in organic and inorganic N concentration and inorganic flux. However, this may be due to the small dataset used which was based on seasonal data collected in the pre-restoration (canopy cover N = 7, ADFM N = 7) and post-restoration periods (canopy cover N = 6, ADFM N = 6). Future investigations should include high-frequency data collection of all parameters to ensure a more robust regression analysis can take place. Moreover, this study compared watershed average canopy cover and ADFM to nitrogen concentrations and flux at the watershed outlet. This was done to determine how

changes throughout the watershed impacted downstream transport of nitrogen. However, this may be a limitation and may have caused reduced potential of relational effects due to comparing two different scales of data. Future research could investigate if relationships exist between the watershed characteristics and nitrogen data collected solely at the watershed outlet. The multiple regression analysis was significant for DON and ammonium flux providing evidence that altered watershed characteristics (e.g.: changes to canopy cover and AFDM) play an important role in DON and ammonium flux.

Planning future restoration projects around these findings could have significant implications for organic and inorganic N concentration and flux within restored watersheds. For instance, as the decrease in canopy cover and increase in AFDM had no significant effect on the majority of parameters assessed, restoration planners and practitioners may clear the canopy along the stream reaches to increase accessibility which may reduce the amount of time needed for restoration to be completed – and ultimately decrease the time the stream channel itself is disturbed.

3.5.2 RQ3.2

For all parameters, flux increased from pre-restoration to post-restoration during the dormant season with all parameters beside nitrate exhibiting a statistically significant increase. Increases in TDN and DON may be due to the streams being more connected to their floodplains allowing for the in-washing of newly fallen particulate and dissolved organic N from the floodplain and additions from increased instream AFDM (Pinay et al., 2000). Here, post-restoration, the stream channel waters were able to reach the surrounding floodplains and bring with them sources of nitrogen inaccessible in the pre-

restoration period. Thus, overall sources of nitrogen to the stream were increased significantly in the post-restoration period including ammonium. Furthermore, a reduction in ammonium could also be attributed to a decrease in biomass respiration leading to an increase in stream oxygen content (Minaudo et al., 2015) Increases in nitrate may indicate limited denitrification potential and reduced transient storage during the dormant season allowing for a slight, non-significant increase. Similarly, Bernhardt & Likens (2004) found that streams located within the Hubbard Brooke Experimental Forest exhibited higher N during the dormant season than the growing season. If restoration must take place over the dormant season and there is a restoration goal to decreased N within the stream, restoration planners and practitioners may seek to remove excess sources of N from the floodplain (e.g.: reducing fertilizer use, excluding livestock, ensuring sewer systems are functioning) to ensure that goal of decreased N is met. Furthermore, increases in TDN, DON, and ammonium flux may also be attributed to increases in dormant season Q. Here, ammonium that was formed in the previously dry unsaturated zone may be pushed into the stream as baseflow as the unsaturated zone become saturated.

Findings from the seasonal indices analysis suggest that the majority of N flux for total dissolved nitrogen and dissolved organic nitrogen during the dormant season occur in December while ammonium occurred in November and nitrate in March. During the post-restoration period, the majority of N flux for all species occurred in the month of February. Results indicate that, environmentally, the watershed exports the majority of nitrogen during the end of the dormant season when vegetation has yet to emerge from dormancy. Understanding the seasonal component of organic and inorganic N flux can

help restoration planners determine the best time to begin projects during the dormant season to cause the least alteration to the natural N cycle. Furthermore, understanding how restoration could lead to more consistent N maximums could help watershed managers know when to best regulate N within the watershed.

3.5.3 RQ3.3

Findings indicate that TDN, DON, and ammonium flux increased from pre-restoration to post-restoration during the growing season with TDN and ammonium exhibiting a non-statistically significant increase and DON exhibiting a statistically significant increase. Increases in TDN, DON, and ammonium may be due to changes in characteristics of the watershed allowing for increased connection to floodplains (e.g.: allowing for the in-washing of particulate and dissolved forms of N), increased stream complexity (e.g.: increasing retention in pools and back eddies), and increased stream and riparian temperatures (e.g.: warmer riparian zone may inhibit the nitrification process by lowering oxygen and reducing bacteria activity and may increase the mineralization of organic N).

Statistically significant decreases in nitrate may be due to changes throughout the watershed that allowed for denitrification to occur. Similarly, Arheimer et al. (1996) and Bernhardt & Likens (2004) found that inorganic N decreased during the growing season. Mechanisms for reducing nitrate may include increasing hyporheic exchange and GW-SW interactions and improving conditions after restoration such as increased light availability and increased algae biomass (Minaudo et al., 2015). For instance, the addition of meanders allowed for an increase in surface area where the stream water has extended contact with surrounding hyporheic zone (Zarnetske et al., 2011a). These zones are

known hotspots for denitrification. Furthermore, the addition of stream features such as rock and wood cross vanes allow for hyporheic exchange to occur as changes in hydraulic head are increased. Together, these changes may have contributed significantly to the reduction of nitrate within the watershed. If restoration must take place over the growing season and there is a restoration goal to decreased nitrate within the stream, restoration planners and practitioners should seek to ensure features are added to restoration plans that aid the retention time and transformation of N. Furthermore, decreases in nitrate flux may also be attributed to decreases in growing season Q.

Findings from the seasonal indices analysis suggest that the majority of N flux for ammonium, TDN, and DON during the growing season occurred in June during the pre-restoration period with a slight shift to May during the post-restoration period. For nitrate, the majority was moved in the month of May for both the pre- and post-restoration periods. Results indicate that, environmentally, the watershed exported the majority of nitrogen during the start of the growing season and is further reduced as the growing season continues as riparian and instream vegetation increase and biogeochemical processes occur with favorable conditions. Understanding the seasonal component of organic and inorganic N concentration and flux can help restoration planners determine the best time to begin projects so as to cause the least alteration to the natural nitrogen cycle. Moreover, this can also begin to shed light on how restoration at the watershed-level can impact N flux by shifting N flux seasonal highs over time and help watershed managers better prepare nutrient management plans.

3.6 Conclusion

As nitrogen pollution is a substantial part of water quality degradation worldwide (Grimm et al., 2003; Drever, 2005; Kaushal et al., 2014; Zhou et al., 2017; Journey et al., 2018; Lavelle et al., 2019), it is essential that restoration planners and practitioners work to create and implement restoration projects with the goal of managing N concentrations and flux. While N is an essential component of ecosystem functioning, and overabundance of N can cause ecosystem degradation (Drever, 2005) and must be addressed to ensure ecosystem functioning. This study documents stream restoration at the watershed scale has the potential to decrease nitrate and TDN concentrations and flux while simultaneously increasing DON and ammonium concentration and flux.

A vital challenge for sustainable ecosystems is ensuring that all stages of the N cycle are functioning. To begin to address nutrient cycling within stream exhibiting characteristics of the USS, restoration projects must incorporate strategies (e.g.: reconnect floodplains with channels, reduce channelization, and alter channel features with the addition of rock or log cross vanes, back eddies) which create hotspots for retention, uptake, and removal (e.g.: denitrification and biological uptake) and which reduce aquatic N concentrations. Furthermore, as N additions to streams continue to increase worldwide, researchers must continuously examine the benefits of various restoration approaches to determine the best strategies for remedying nutrient pollution. This study aimed to determine if the whole-watershed stream restoration approach used within the Reedy Creek watershed had the ability to alter N processing. Overall, the efficacy of using the whole-watershed stream restoration approach was successful at reducing N at the watershed outlet within the Reedy Creek watershed. This study

supports the concept that restoration practices that enhance instream processes and groundwater-surface water connections provide the conditions needed to impact N cycling.

As this study did not set out to identify specific restoration design features that may enhance nitrogen processes and alter N, a limitation of this study includes a need to assess the mechanisms responsible for N changes. Future research is needed to further identify, assess, and quantify the mechanisms responsible for alterations to N within restored watersheds. Findings require future research to investigate if changes in watershed characteristics were adequate for addressing organic N retention/transformation and to determine any changes in organic N sources. Furthermore, an additional limitation of this study is the focus on baseflow N concentration and flux. To fully understand the impact that restoration has on N cycling, future research could investigate how effective restoration features (i.e.: the addition of cross vanes, changes in sinuosity, addition of riffle-pools, and the addition of bank woody debris) are at retaining/transforming N during differing flow events and at differing N concentrations. Additionally, as streams behave differently based on their surrounding land-uses, future research could compare restoration projects across land-use types to determine if streams exhibiting different characteristics respond differently to restoration with regard to N processing. Finally, extended monitoring of the Reedy Creek watershed could elucidate how streams respond to restoration and how long and short-term nutrient dynamics compare.

4 SUSPENDED SOLIDS DYNAMICS IN AN URBAN FORESTED STREAM BEFORE AND AFTER WHOLE-WATERSHED STREAM RESTORATION

4.1 Introduction

Small forest streams are dynamic, constantly changing size and shape with regard to flow rates and sediment loads and act as conduits transporting materials derived from upstream terrestrial landscapes to downstream locations and ultimately to the ocean or other large bodies of water. Sediment loads are the amount of sediment dissolved in, carried by, and pushed along the bed of a stream channel (Bierman & Montgomery, 2014) and their movement is regulated by sediment sources and flow regimes. It is important to note that the movement of water making up a watershed's flow regime is more complex and bidirectional than the "pipe" model implied by the watershed water balance (Bencala, 1993; Bencala et al., 2011) with all flow paths controlling the movement of sediments within the watershed. Potential flow paths of water include stream flow from higher to lower elevations, flooding onto the surrounding floodplain, and hydrologic retention at multiple time scales (Bencala, 1993; Bencala et al., 2011) as well as overland flow bringing with it a plethora of sediments to and from the surrounding landscape. The amount of sediment entering the stream through those flow paths is controlled by surrounding land-use, topography, and geology (Filoso et al., 2015).

In small forest streams, sediment loads can be derived from the natural floodplain, hillslope erosion, bank erosion, and shear stress, as well as the flushing of organic matter (e.g.: leaves) that is broken down by macroinvertebrates instream and water flows (Charlton, 2008; Holden, 2014). Sediment load consists of dissolved load, suspended

load, and bed loads; these loads are distinguished from each other with regard to size and coarseness (Charlton, 2008). Dissolved loads consist of ions and molecules that are carried in solution (Bierman & Montgomery, 2014). Suspended sediments consist of small (i.e.: less than 2mm – 0.002mm), fine grain particles including clay particles, silt, and sand which are easily suspended within the water column (Charlton, 2008; Bierman & Montgomery, 2014; Holden, 2014). Bedload sediments tend to be larger (i.e.: greater than 2mm), more coarse sediments that are unable to be lifted in suspension and instead travel along the stream bed over short distances (Charlton, 2008; Bierman & Montgomery, 2014).

According to Simon & Klimetz (2008), sediments are a leading form of surface water quality degradation in the United States. Suspended sediments can cause environmental damage and impair chemical and ecological water quality in several ways including: reduction of light transmission leading to decreased photosynthesis potential and reduced visual range of aquatic organisms; decreased dissolved oxygen released from aquatic flora; increases in water temperature as sediment absorbs heat from sunlight (Holden, 2014); transporting sorbed trace elements; smothering of benthic species; damage to water supply systems (Davies-Colley & Smith, 2001); and sediment deposition and accumulation on floodplains leading to a reduction in overland flow rates and the ability of sediment to be effectively moved downstream (Hupp et al., 2009). Chemical water quality degradation does occur through the addition of sediments and trace elements to the stream environment (Horowitz, 1985; Copaja et al., 2020). Here, trace metals can be sorbed to suspended sediments or precipitated to bed sediments (Horowitz, 1985; Copaja et al., 2020) where they may take different paths such as being

retained, bioaccumulated, or leaving the watershed as sediment yield. These issues exacerbate water quality declines as nutrient processing is further impaired leading to issues such as eutrophication or increased toxicity.

For a stream channel to be considered stable and for water quality to be acceptable, there must be a balance between sediment load and discharge; in other words, a stabilized channel is one in which erosion and deposition rates are balanced (Simon & Klimetz, 2008; Holden, 2014). Transitioning from forested to urban land-uses requires construction which disrupts natural sediment dynamics and may deposit more sediments within streams than would naturally occur over several decades (U.S. EPA, 2018a). Within urban-suburban areas, impervious surfaces disrupt natural stream functioning and alter stream sediment processing (Thompson et al., 2018). Moreover, historical stream alteration, such as channelization, typical of urbanized areas, also alters stream sediment dynamics by decreasing residence times and limiting the ability of streams to retain sediments (Walsh et al., 2005; Thompson et al., 2018). Alteration of streams, and the issues associated with alteration, becomes increasingly significant in headwaters, as headwater streams make up 60-80% of the total stream length and drain 70-80% of all watershed area (MacDonald & Coe, 2007). Restoration has become an increasingly popular tool used to reduce streams' sediment loads (Filoso et al., 2015).

The natural channel design (NCD) method, implemented across North Carolina restoration projects, aims to restore degraded streams to match reference streams for various stream characteristics, including natural sediment dynamics, by focusing on restoring stream form and function (Rosgen, 2011). The NCD approach seeks to restore streams to an equilibrium state where the stream is able to adjust to its new flow path

(Rosgen, 2010, 2011). To do so, alterations of stream width to depth ratios, sinuosity, and slope may take place leading to altered hydraulic characteristics such as stream power, velocity, and shear stress which in turn alter stream sediment dynamics (Rosgen, 2011). Here, the newly restored stream is allowed to adjust along its restored reach where flow regimes work to deposit, scour, and erode stream bed and bank sediments so long as the base-level of the stream is not disturbed and rates are not accelerated (Rosgen, 2011). In so doing, the NCD aims to recreate natural sediment patterns of reference streams at the restored site. In degraded systems, it is essential that natural sediment dynamics are restored to provide the conditions needed for nutrient transformations, enhance groundwater-surface water (GW-SW) interactions, to ensure the survival of aquatic species, and to ensure the stream channel itself is stabilized. This study aims to determine whole-watershed stream restoration's ability to alter baseflow and stormflow TSS concentrations and flux in the Reedy Creek watershed.

4.1.1 Total Suspended Solids

Suspended solids, a subset of total sediments, are the quantity (by weight per unit volume) of suspended, undissolved particles in the water column that can be collected and trapped by a filter of a nominal pore size. Suspended solids generally range in size from ~0.005 to 1.0 mm and are made up of materials such as silt, sand, clay, algae, particulate organic matter, and other small particles within the water column (Holden, 2014) with flow velocity dictating which will be suspended at a given time. Suspended sediments make up the majority of annual sediments within a stream (Simon et al., 2004) and, according to Glysson (2002), “fluvial sediment [are] the single most widespread pollutant in the Nation's rivers and streams, affecting aquatic habitat, drinking water

treatment processes, and recreational uses of rivers, lakes, and estuaries”. Excess suspended sediments can degrade ecosystem form, function, and biodiversity. If a system contains excessive total suspended solids (TSS), biodiversity may be impaired, with excess sediment leading to fish fry mortality and behavioral changes (Rowe et al., 2003; Simon et al., 2004). Excess sediments, especially once settled, may also clog streambeds and floodplains by filling interstitial openings leading to decreased hyporheic and GW-SW exchange (Bunte & Abt, 2001). However, suspended sediments are needed to maintain proper ecosystem functioning as sediments provide direct and indirect benefits to streams. Sediments provide nutrients necessary for floodplain function and aquatic vegetation growth which forms the basis of the aquatic food web. Moreover, sediments are necessary for the building and changing of channel morphology. According to Simon et al. (2004), suspended sediment transport rates can be grouped into three categories: 1) natural conditions characteristic of stable channels, 2) impacted conditions characteristic of increased erosion and disturbance, and 3) impaired conditions where erosion and sediment transport rates are so high that they disturb aquatic biodiversity and stream functioning.

4.1.2 Stream Sediment Dynamics at Baseflow

Unimpacted streams need a steady supply of sediment to maintain ecosystem functioning. While baseflow carries ~10 times less sediment concentration per unit time than carried by stormflows (Edelmann et al., 2002), baseflow is the hydrological condition at which streams spend most of their time, providing the ambient conditions and flow regimes at which stream ecology is based. Moreover, in a study conducted by Ismail (2000), as much as 60% of TSS flux are carried by stormflow. This indicates that

baseflow could be responsible for transporting at least 40% of TSS flux, and therefore baseflow conditions represent a significant part of the restoration success equation for controlling erosion, especially from the bed and banks. Moreover, smaller flow, such as occur during baseflow, can move smaller particles that are easily suspended.

Sediment dynamics in streams are regulated by land-use, topography, and sediment sources (Ankers et al., 2003; Simon et al., 2004). Within these streams, sediments are derived from the terrestrial landscape, instream processes, or some combination of the two (MacDonald & Coe, 2007) depending on weather conditions. As natural streams within the Piedmont region on North Carolina tend to be groundwater-driven at baseflow, sediment dynamics at baseflow tend to be dominated by stream channel, bank, and bed erosional processes (Table 4.1) (Rosgen, 2010). A study conducted by Langendoen et al. (2012) found that bank erosion accounted for 36% of TSS along the Missisquoi River. Moreover, groundwater driven streams tend to have characteristically low sediment concentrations due to stable hydrologic dynamics (Estrany et al., 2009). Groundwater additions to the streams tend to be sediment-free and work to dilute the sediments that may be suspended during storms (Estrany et al., 2009).

Urbanized/impaired streams produce and transport more sediment than their natural counterparts (Simon et al., 2004). Here, sediments that were carried to, and deposited within, the stream by overland flow can be resuspended and moved at baseflow. According to Houser et al. (2006), land-disturbances lead to an increase in TSS concentrations during baseflow with increasing disturbances leading to increase TSS concentrations. Urban/impaired streams typically have higher levels of suspended sediments during baseflow (O'Driscoll et al., 2010) due to changes in hydrology leading

to increased instream erosion (Table 4.1). Changes to land-use typical of urbanized areas lead to altered sediment transport and altered channel morphology (O'Driscoll et al., 2010). Moreover, as urbanization increases, areas are transformed from historical land-use types (i.e.: forested, prairie) to areas with exposed sediments which causes increased sedimentation as the newly exposed sediments are easily suspended and carried to nearby streams during storm events. Once those newly exposed sediments become built-out (e.g.: suburbs, shopping malls), the sediment added to the stream decreases leading to channel erosion, widening, and incision (O'Driscoll et al., 2010).

Historic alteration of streams has led to an increase in erosional processes which resulted in increased TSS. In channels that have become incised and are therefore confined, bank erosion will occur at baseflow (Elliott et al., 2010). However, it is the goal of restoration to return streams to more natural conditions (both appearance and function), and if achieved, restoration should result in decreased TSS as erosional potential is decreased due to natural stream morphology and the addition of riparian and aquatic vegetation (Table 4.1). Results from a restoration project in Spring Creek, Pennsylvania indicate that restoration was successful at reducing TSS concentrations at baseflow via the addition of riparian buffers, bank stabilization, and overall stream channel stabilization (Carline & Walsh, 2007). The TSS within restored streams may be transient depending on the extent of restoration and the time needed for the stream to return to pre-disturbed conditions. It is important to note that while these ecosystems should be dynamic and free to shift naturally, channel morphological success exists where the channel is not deviating toward unstable conditions (Doll et al., 2021). In other words, for channel restoration to be successful, the channel should not aggrade or

degrade substantially. Instead, restoration success should be viewed as a delicate balance of channel sediment transport in which the channel neither aggrades nor degrades

(Harman & Jennings, 2001).

Table 4.1 Processes regulating sediment dynamics of small streams based on stream condition and flow regime.

Stream Type	Baseflow	Stormflow
Unimpacted (forested)	Channel, bank, and bed erosion processes controlled by stream morphology dictated flow rates and diluted by groundwater (Estrany et al., 2009)	Channel, bank, and bed erosion processes regulated by natural infiltration and flushing of old groundwater to streams. Storms are less flashy due to increased infiltration rates which leads to decreased overland flow and thus less erosion than in urban streams. (Hornberger et al., 1998; Holden, 2014)
Urban	Channel, bank, and bed erosion enhanced by channelization induced faster flow rates and decreased discharge leading to increased TSS concentrations (O'Driscoll et al., 2010)	Channel, bank, and bed erosion enhanced and increased by the addition of impervious cover leading to decreased infiltration rates, increased overland flow, and increased flashiness. Sediment issues are further exacerbated by the addition of sediments from construction sites (O'Driscoll et al., 2010; Russell et al., 2019)
Restored	Return of stream to natural conditions where channel, bank, and bed erosion processes are controlled by stream morphology dictated flow rates and diluted by groundwater (Estrany et al., 2009; Doll et al., 2021)	Return of stream to natural conditions where channel, bank, and bed erosion processes are regulated by natural infiltration and flushing of old groundwater to streams. Less flashy = less erosion. (Rosgen, 2011)

4.1.3 Stormflow Suspended Solid Dynamics

Stormflow sediment dynamics are a critical component of watershed sediment dynamics as the majority of sediment is transported during stormflow (Holden, 2014; Ismail, 2000). Suspended sediment concentrations are significantly greater during stormflow than baseflow (Edelmann et al., 2002). These large discharge events, such as are seen during storms and wastewater releases, characteristically exhibit increased erosion and overland flow which ultimately increases stream suspended solid loads (Ellison et al., 2014; Ismail, 2000). Additionally, Filoso et al. (2015) found that sediment inputs tend to be dominated by stormflow events and are typically highest in the late summer and early fall. Moreover, stormflow TSS dynamics differ with surrounding land-use with urbanized areas exhibiting higher surface runoff and suspended sediment yields than their natural counterparts.

Within unimpacted systems, sediment may be derived from the terrestrial landscape or from instream relocation and erosion of bank and bed sediments (Table 4.1). The sediment loads within unimpacted streams are directly related to precipitation amounts and sediment available. Antecedent conditions and precipitation amounts define the source of stormflow and amount of TSS (Hewlett & Hibbert, 1967; Ismail, 2000; MacDonald & Coe, 2007) that enters the stream channel. In unimpacted systems, precipitation has the potential to infiltrate soil while older water is released from soil pores (Hornberger et al., 1998) and enters the channel as stream flow (i.e.: baseflow). When compared to urban watersheds, unimpacted headwater watersheds have increased infiltration capacities and should have less total sediment entering the stream. Additionally, overland flow, or infiltration-excess flow, also occurs in unimpacted

systems in locations where soils are compacted or where precipitation rates exceed infiltration capacities (Holden, 2014). Here soil particles and other debris on the land surface are picked up in the flow and enter the stream.

More than their unimpacted counterparts, urban/impaired systems have anthropogenic stressors that alter sediment dynamics. According to Shinomiya et al. (2020), alterations of the landscape lead to increases in TSS concentrations within disturbed forests. Stormflow TSS discharges were found to be increased after land disturbance at the Fort Benning Military Installation in Georgia (Houser et al., 2006). Furthermore, stormflow TSS concentrations were seen to increase with increasing discharge in a disturbed agricultural watershed in Wisconsin (Rose & Karwan, 2021). Larger discharge events associated with stormflow are characteristic of urban streams in which increased levels of impervious cover and urban storm water systems act as conduits directing water to streams along shorter, more direct flow paths (Table 4.1). Increased stormflow due to urbanization leads to erosion induced channel enlargement (O'Driscoll et al., 2010; Russell et al., 2019). In urbanized systems, sediment may be derived from construction sites, roadways, roofs, and other pervious and impervious surfaces; however, sediment from terrestrial landscapes may be reduced in urban streams making stream banks vital sediment sources (O'Driscoll et al., 2010) – especially in incised channels where flood waters rarely overtop banks (Rosgen, 2010). Restoration practices must take into consideration landscape sediment dynamics as sediments are the building blocks of aquatic ecosystems. Designs such as the natural channel design (NCD) (Rosgen, 2011) focus on bankfull channel conditions as a key component of restoration with overarching goals of improving water quality, stabilizing

streambanks, and reducing sediment and nutrient losses. The NCD uses 67 variables to create a reference profile to which degraded systems are restored (Rosgen, 2011). Here, reference stream profiles focus on the cross-sectional area (i.e.: bankfull channel) of the channel as that area represents the sediment, flow regime, and bank/bed materials necessary for stream functioning (Rosgen, 2011). Bankfull conditions are known to represent channel-forming flows as large amounts of sediment are transported during these events (Simon et al., 2004), are the point at which flooding is imminent (Rosgen, 2010), and are related to channel morphology of channels of various sizes (Rosgen, 2011).

Restored streams should lead to decreased flashiness and decreased watershed erodibility and, ultimately, to decreased TSS (Table 4.1). To do this, the NCD (Rosgen, 2011) approach incorporates stream stabilization and structure enhancements such as the addition of large woody debris to dissipate energy from flows, grade controls to limit stream velocities, and the addition of riparian vegetation which is protected until it establishes and works to reduce bank and riparian erosion. However, depending on the scope of restoration (e.g.: restoration only focusing on a small reach), the overarching issues may not be addressed which may lead to restoration having no beneficial effect on TSS. To have an impact on TSS, restoration should address issues at larger spatial scales.

4.1.4 Sediment in Piedmont Streams

Piedmont streams within NC have a history of degradation caused by channelization, dredging, damming for power generation, urbanization, and bank erosion leading to a loss of riparian vegetation (Harman & Jennings, 2001; Wegmann et al., 2013). The Piedmont of NC has undergone intense erosion since the arrival of European

colonizers when destructive agricultural activities began (Trimble, 2008). According to Trimble (2008), erosion was minimal during pre-European colonization with records indicating many streams across the Piedmont were clear and free of suspended debris. However, as European colonizers moved across the landscape, erosion intensified leaving a wake of altered and degraded landscapes and streams in their path. According to Trimble (2008), erosional land-use intensity for five Piedmont regions reached their peak between the years of 1860-1920. Clear cutting, planting, and abandoning of fields in the Piedmont led to gullying and sediment erosion. These newly eroded, legacy sediments were washed downstream where they buried more fertile sediments and caused stream aggradation along higher-order streams (Trimble, 2008).

Currently, Piedmont streams exhibit heavy incision and carry high sediment loads (Mukundan et al., 2011) as previously aggraded sediments are exported downstream. The incision and high sediment load typical of Piedmont streams negatively affects water quality (Wegmann et al., 2013). According to Mukundan et al. (2011), Piedmont streams have experienced major changes over the last century including periods of decreased hillslope erosion, sediment retention due to flood control reservoirs, and channelization. These changes have increased stream power and ultimately led to channel instability and incision; especially as Piedmont streams tend to flow through areas with easily eroded legacy sediments (Mukundan et al., 2011) such as sand, silt, and clay (O'Driscoll et al., 2010). Piedmont streams also have characteristically wide channels due to shallow bedrock limiting channel bed incision (O'Driscoll et al., 2010). Within North Carolina, the only existing TSS standards are for trout (10 mg/L) waters or high quality waters (20 mg/L) of the state (U.S. Environmental Protection Agency (EPA), 2018) and, therefore,

cannot be applied to the Reedy Creek watershed where waters have been listed as impaired.

4.2 Study Objectives and Research Questions

The objective of this study was to determine whether stream restoration has influenced TSS concentration and flux within the watershed at both baseflow and stormflow. Here, I explored to what extent does storm size and intensity impact TSS concentration and flux. Additionally, this study aimed to better understand the temporal component of TSS concentration and flux to determine if restoration has led to a more stable stream system at baseflow. For this study I asked two questions:

(RQ4.1) *Does restoration impact TSS concentration and flux at stormflow during pre- and post-restoration periods?* I hypothesized that:

Hypothesis	Hypothesized mechanism
Storms before and directly after restoration will have higher TSS concentration and flux compared to the post-restoration stream.	During both the pre- and post-restoration periods, TSS concentration and flux will vary based on the duration and size of the storm. Quick, high precipitation events will have higher TSS concentration and flux (per storm and per unit time) than longer-duration, lower-intensity events with the same precipitation amounts. However, overall, the stream bank and beds within the watershed would become more stable (i.e.: erosional potential would decrease) in the post-restoration period leading to a decrease in stormflow TSS concentration and flux.

(RQ4.2) *Does restoration impact baseflow TSS concentration and flux during pre- and post-restoration periods?* I hypothesized that:

Hypothesis	Hypothesized mechanism
Baseflow TSS concentrations and flux would decrease post-restoration.	TSS concentration and flux at baseflow will be less variable but higher before restoration because of a relatively uniform rate of bank erosion in the incised, straightened pre-restoration stream. After restoration, there will be a transient period of increased TSS concentration and flux as the sediments newly added to the streambed and bank settle. After the settling period, post-restoration TSS concentration and flux will return to more stable conditions and will overall be lower than the pre-restoration period. Here, changes in soil type will lead to increased soil cohesion thus requiring higher shear stress to dislodge, move, and suspend bank and bed particles.

To test these hypotheses, baseflow and stormflow surface water samples were collected and analyzed for TSS.

4.3 Methods

4.3.1 External Data Sources

Several datasets were collected and used for this study from outside sources. First, precipitation data were used from the USGS RC Rain gage (US Geological Survey, 2022b) (351540080430045 CRN-16) located within the Reedy Creek Park and Nature Preserve. This monitoring station records precipitation data continuously and provides a daily sum of precipitation. Second, stream discharge data were taken from the USGS RC Discharge gage #0212427947 (US Geological Survey, 2022a) located at the outlet of the Reedy Creek watershed. This monitoring station records data continuously at five-minute

increments. Third, composite samples of storm total suspended solids were obtained from the Charlotte-Mecklenburg Storm Water Services (CMSWS) monitoring program (Charlotte-Mecklenburg Storm Water Services (CMSWS), 2021a). This monitoring program collects flow-weighted stormwater samples using a composite ISCO sampler. Samples are collected from the ISCO at the end of each storm.

4.3.2 Field Methods

For baseflow, samples were taken monthly at the watershed outlet, site R1 (Figure 1.5), within the Reedy Creek watershed. The analysis period was bound from December 2016 through February 2021. This sample range was chosen as it represents the two-year period before restoration started and the two-year period after restoration was completed (Table 1; site R1). Baseflow samples were taken using 1-liter polyethylene bottles which were acid washed and rinsed thoroughly with deionized water prior to use. Bottles and caps were rinsed three times in surface water directly downstream from the sampling site before sample was taken. The samples were taken by submerging the bottle upstream and allowing water to flow in, leaving no headspace. The TSS samples were transported to the lab for immediate filtering. Discharge measurements were taken from a USGS gauging station located at the watershed outlet, approximately 200m downstream of R1, the surface water sampling location.

4.3.3 Lab Methods

For stormflow, a dataset was created that consists of date, time, storm duration, Q , and TSS concentration retrieved from CMSWS. Here, there was one data point for each parameter per storm. Additionally, antecedent precipitation was calculated for 48 hours prior to the start of each event; this data were retrieved from the USGS continuous

monitoring network (US Geological Survey, 2022b). Antecedent precipitation was determined by creating hydrographs of individual storms, determining the hydrograph rising limb start time, and taking the average of precipitation from 48 hours prior to that point.

Baseflow TSS was determined gravimetrically. Samples were filtered immediately upon returning from the field site and samples were shaken to resuspend particles before filtering. During filtering, water samples of a known volume (i.e.: volumes filtered varied but variations were taken into account when calculating TSS concentrations and flux (Equation 4.1)) were vacuum filtered through pre-combusted pre-weighed 0.7-micrometer glass fiber filters (GF/F). After filtering, GF/Fs were placed in a drying oven at 50°C for 24-48 hours. Once the filters dried, they were reweighed for TSS before discarding.

4.3.3 Data Processing

The TSS concentration and flux of samples were calculated for the watershed outlet (Figure 1.5) using Equation 4.1 in which PW is the initial weight of the filter and dry weight is the weight of the filter after the sample of a known volume is passed through and the filter has been dried at 50°C for 24-48 hours. Baseflow TSS flux, at the R1 site, was determined using Equation 4.2 in which TSS is the total suspended solids measured in milligrams per liter and Q is discharge in m^3s^{-1} . Baseflow discharge for each sample was set to the average discharge on that day. Equation 4.3 was used to determine the TSS flux for the entire study watershed in kg per hectare per year. The TSS for baseflow was collected weekly using the method described above. Equation 4.4 was used to determine the TSS flux per storm. Discharge measurements for each event were

retrieved from the CMSWS (Charlotte-Mecklenburg Storm Water Services (CMSWS), 2021a). To determine Q for each TSS sample, Q was determined from the average of daily Q at the watershed outlet. If Q could not be determined the sample was discarded.

$$TSS (g L^{-1}) = (dry\ weight (g) - PW(g)) / ((water\ volume (mL))/1000) \quad \text{Equation 4.1}$$

$$Flux (g s^{-1}) = TSS (g L^{-1}) * discharge (m^3 s^{-1}) * (1000 L / 1 m^3) \quad \text{Equation 4.2}$$

$$Flux (kg\ hectare^{-1}\ year^{-1}) = ((Flux (g s^{-1}) / 1000) * 3.154 \times 10^8) / 600 \quad \text{Equation 4.3}$$

$$Flux (per\ storm\ kg) = TSS (g L^{-1}) * Per\ storm\ Discharge (m^3) * (1000 L / 1 m^3) \quad \text{Equation 4.4}$$

Storm water total suspended solids (TSS) measurements were retrieved from CMSWS's instream stormwater monitoring program. The CMSWS monitoring program uses automated ISCO samplers to collect composite TSS samples representative of the entire storm hydrograph. Data from 2016-2021 were requested from CMSWS. For quality control, only storms that had data representing 70%+ of the storm volume were used for analysis. Storms were grouped into before and after restoration periods.

4.3.4 Statistical Analysis

Basic descriptive statistics (e.g.: mean, median, mode, standard error, standard deviation, range, minimum, and maximum) were calculated for baseflow (pre- and post-

restoration) and stormflow (individual storms) TSS concentrations and flux. Histograms were created for baseflow and stormflow TSS concentration and flux before and after restoration and visually examined for apparent distribution types. Scatter plots were created to show X-Y relationships between discharge and TSS concentrations, at baseflow and stormflow, before and after restoration. Time series plots were constructed and visually examined for stormflow discharge amounts and storm duration. All statistical analyses were conducted in R (R Core Team, 2019). R packages used include base R (R Core Team, 2019) and ggpubr (Kassambara, 2020). ggpubr was used for general correlation analyses (Kassambara, 2020).

To answer RQ4.1, *does restoration impact TSS concentration and flux at stormflow during pre- and post-restoration periods*, Mann-Whitney tests were used to assess differences in TSS concentration and flux at stormflow before and after restoration (Whitlock & Schluter, 2009). The Mann-Whitney test was chosen as it is a non-parametric test that does not assume normality of data. A correlation analysis was conducted between antecedent discharge and TSS concentration and flux to understand the relationship between the two parameters.

To answer RQ4.2, *does restoration impact TSS concentration and flux at baseflow during pre- and post-restoration periods*, Mann-Whitney tests were used to assess differences in TSS concentration and flux at baseflow before and after restoration (Whitlock & Schluter, 2009). The Mann-Whitney test was chosen as it is a non-parametric test that does not assume normality of data.

4.4 Results

4.4.1 (RQ4.1) Does restoration impact TSS concentration and flux at stormflow during pre- and post-restoration periods?

4.4.1.1 Storm Dynamics

Stormflow dynamics (e.g.: total stormflow runoff (m^3), total storm precipitation (mm), lag times (minutes), and total per storm TSS flux (g)) were assessed for 16 storms with available TSS data from April 2016- April 2021 (see Appendix A and Appendix B). Storms were grouped and compared before and after restoration. Total precipitation was recorded in the pre-, during, and post-restoration periods. Data indicate that total precipitation (mm) amounts increased from a pre-restoration average of 22 mm to post-restoration average of 15 (mm). Parsing the data out further, data indicate that per-storm precipitation (mm) amounts decreased from a pre-restoration average of 23 mm ($N = 6$) to post-restoration average of 15 (mm) ($N = 7$). Average precipitation was higher (22 mm) and less variable (18 mm) in the pre-restoration period when compared to post-restoration (Table 4.2). Post-restoration, average precipitation was lower (15 mm) and more variable (23 mm) than the pre-restoration period. Overall, for the 13 individual storms assessed in this study, storms tended to be smaller and more variable over time (Figure 4.1). Results of a Mann-Whitney U test indicate that there was not a statistically significant difference in the total per storm precipitation in the pre- versus post-restoration comparisons ($W = 31, p > 0.18$).

Table 4.2 Descriptive statistics for total per storm precipitation (mm) for selected storms in the pre-, during, and post-restoration periods. Data represent the storms sampled during the period from 2016-2021

Summary Statistic	Pre-Restoration	During Restoration	Post-Restoration
Number of Storms	6	3	7
Mean	23.24	20.83	14.99
Standard Error	3.61	1.92	2.73
Median	21.80	19.60	14.50
Mode	/	/	/
Standard Deviation	8.07	3.33	7.22
Range	18.60	6.30	23.20
Minimum	15.20	18.30	3.00
Maximum	116.2	24.60	26.20

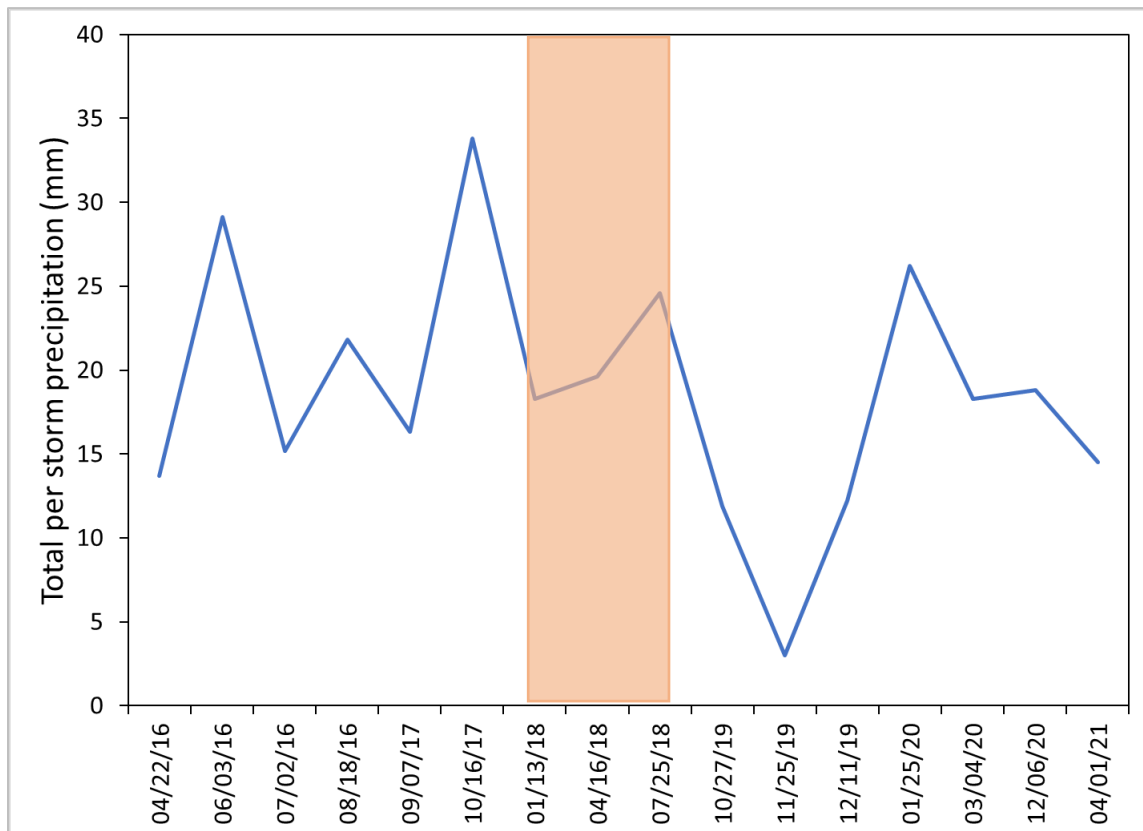


Figure 4.1 Total per storm precipitation (mm) for 13 selected and assessed pre- and post-restoration storms. Orange bar represents the during construction period. Data represent the storms sampled during the period from 2016-2021.

Stormflow dynamics were further explored by assessing difference in total storm discharge in the pre- and post-restoration periods. For total stormflow runoff (m^3), overall, there was an increase from the pre-restoration period average of 5035 m^3 to the post-restoration period average of 51500 m^3 (Table 4.3). Results of a Mann Whitney U test indicate that this increase in stormflow runoff, 46500 m^3 , was not statistically significant, $W = 7$, $p > 0.05$. Here, pre-restoration average stormflow runoff (m^3) was lower at 5000 m^3 and was less variable at 6600 m^3 . Post-restoration, average stormflow runoff was highest at 51500 m^3 and most variable at 263000 m^3 . Overall, stormflow runoff, and therefor storm size, was increased in the post-restoration period (Figure 4.2). There was an overall trend that indicated as storm discharge increased, so too did TSS flux (Figure 4.3).

Table 4.3 Stormflow Runoff (m^3) in the pre-, during, and post-restoration periods at the watershed outlet (R1). Data represent the storms sampled during the period from 2016-2021.

Summary Statistic	Pre-Restoration	During Restoration	Post-Restoration
Number of Storms	6	3	7
Mean	5000	6000	51500
Standard Error	1100	300	36000
Median	5000	6300	20000
Mode	/	6300	/
Standard Deviation	2600	450	95000
Range	6500	800	263000
Minimum	2000	5500	3400
Maximum	8500	6200	270000

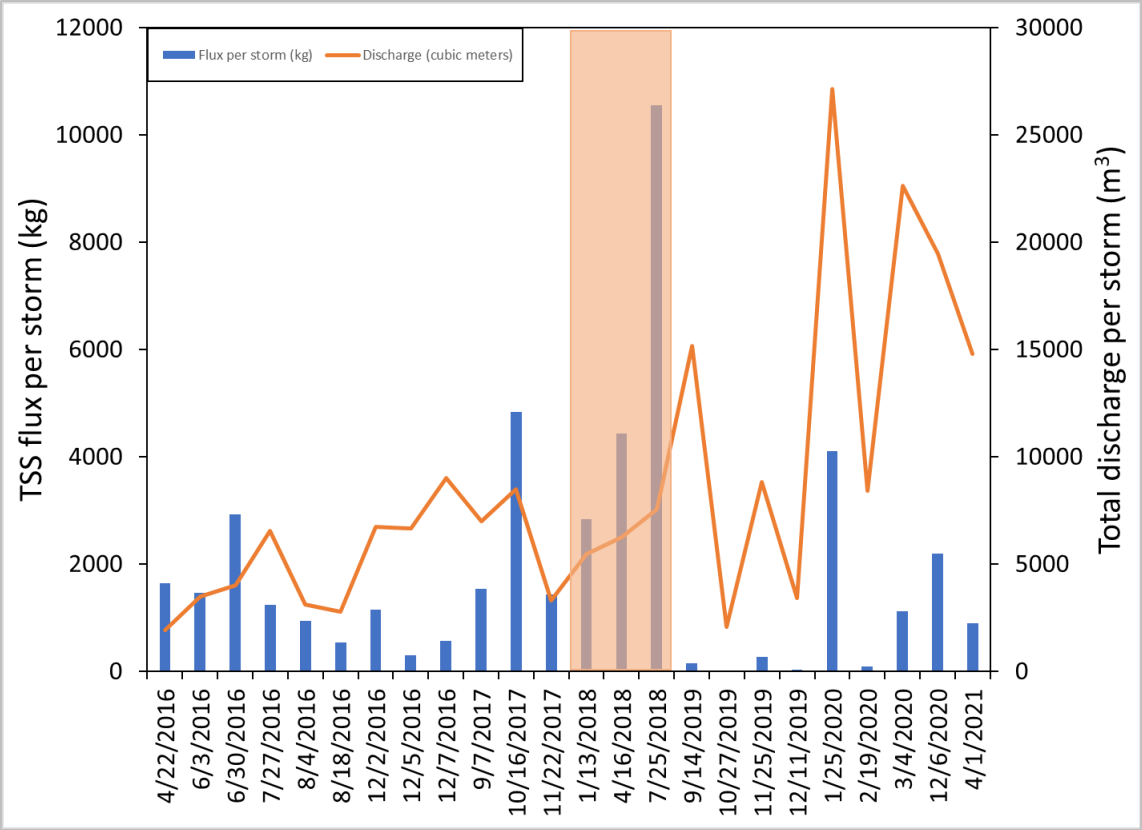


Figure 4.2 TSS flux per storm (g) (blue bars) and total storm discharge (m³) (orange line) at the watershed outlet (R1). Data represent the storms selected and sampled during the period from 2016-2021. Orange bar represents the during-construction period.

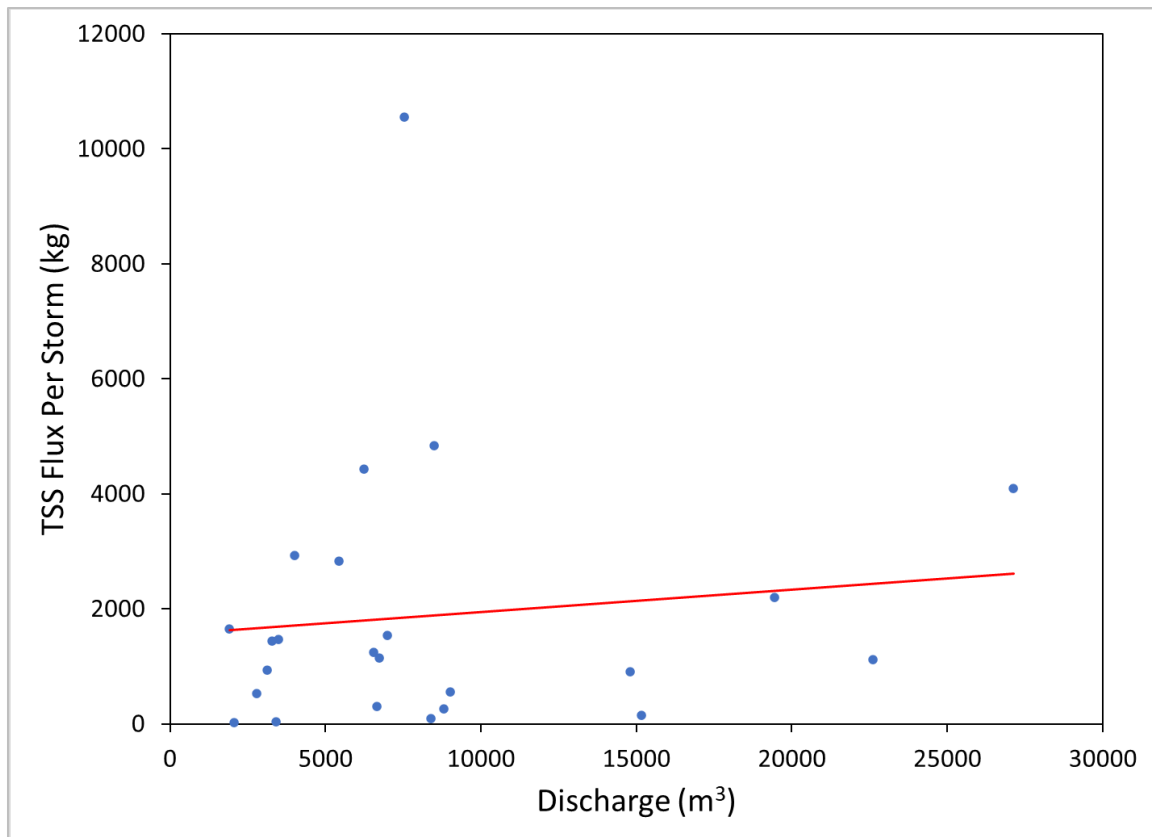


Figure 4.3 TSS flux per storm and total storm discharge (m³) at the watershed outlet (R1). Data represent the storms selected and sampled during the period from 2016-2021. Blue line is the line of best fit.

Storms not only varied in size but also structure with regard to individual storm lag to peak times (minutes). Data indicate that mean storm lag to peak increased from a pre-restoration average of 85 minutes to a post-restoration average of 126 minutes. Results of a Mann-Whitney U test indicate there was not a statistically significant difference in average lag to peak times for the pre- and post-restoration comparisons ($W = 11.5, p > 0.19$). Similarly, results of a Mann-Whitney U test indicate that there was not a statistically significant difference in total lag to peak times either ($W = 9.5, p > 0.11$) Pre-restoration, average lag to peak times were shorter (85 minutes) and less variable (70 minutes) than during the post-restoration period (Table 4.4). During the post-restoration

period, average lag to peak times were longer (126 minutes) and more variable (158 minutes) when compared to the pre-restoration period (Table 4.4). Overall, the average lag to peak time increased over time (Figure 4.4).

Table 4.4 Descriptive statistics for lag to peak times (minutes) for selected storms in the pre-, during, and post-restoration period at the watershed outlet. Data represent the storms sampled during the period from 2016-2021.

Summary Statistic	Pre-Restoration	During Restoration	Post-Restoration
Number of Storms	6	3	7
Mean	85	83	126
Standard Error	12	14	23
Median	80	95	90
Mode	/	/	/
Standard Deviation	28	25	61
Range	70	45	158
Minimum	50	55	73
Maximum	120	100	230

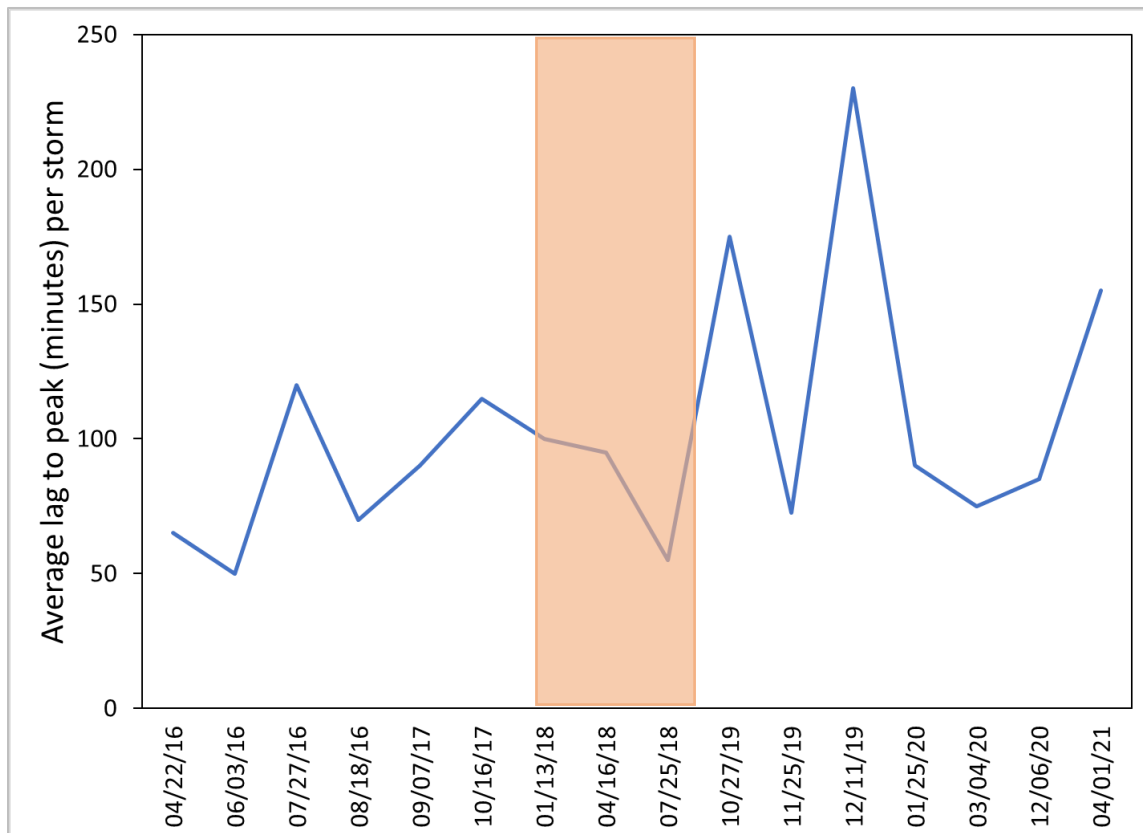


Figure 4.4 Average lag to peak (minutes) per storm at the watershed outlet (R1). Data represent the storms sampled during the period from 2016-2021. Orange bar represents the during-restoration period.

4.4.1.2 Stormflow TSS Concentrations

Stormflow TSS concentrations were compared before and after restoration ($N = 13$). Data indicate that stormflow TSS concentrations decreased from a pre-restoration average of 331 mg/L to a post-restoration average of 67.2 mg/L (Table 4.5). Results of Mann-Whitney U test indicate that this decreased in stormflow TSS concentrations, 263.8 mg/L, was statistically significant, $W = 42$, $p = 0.003$.

Pre-restoration storms were split between low intensity, short-duration (2 storms), low intensity, long-duration (1 storm), high intensity, short-duration (2 storms), and high intensity, long-duration (1 storm) storms (Table 4.11). Post-restoration storms were split between low intensity, short-duration (1 storm), low intensity, long-duration (2 storms),

high intensity, short-duration (2 storms), and high intensity, long-duration (2 storms) storms (Table 4.11). Here, the highest TSS concentration (1400 mg/L) was moved during a high intensity, short-duration event which occurred during the post-restoration period and was the only storm to exceed 1000 mg/L (Table 4.11; 10/27/2019). All storms moving more than 100 mg/L TSS (N = 9) were high intensity storms besides storms that occurred on 4/22/2016, 7/27/2016, and 8/18/2016 which moved 860 mg/L, 189.9 mg/L and 190 mg/L and were classified as low intensity long, duration, low intensity, short-duration, and low intensity, short-duration storms respectively (Table 4.11). Storms moving less than 100 mg/L TSS (N = 4) all occurred in the post-restoration period and were low intensity except for the storm occurring on 12/06/2020 which moved 46.9 mg/L TSS and was classified as a high intensity, long-duration storm.

Table 4.5 Stormflow TSS concentrations (mg/L) in the pre-, during, and post-restoration periods at the watershed outlet. Data represent the storms sampled during the period from 2016-2021.

Summary Statistic	Pre-Restoration	During Restoration	Post-Restoration
Number of Storms	6	3	7
Mean	331.0	876.7	67.2
Standard Error	71.5	267.4	23.1
Median	220.0	710.0	40.0
Mode	190.0	/	/
Standard Deviation	257.6	463.1	73.2
Range	815.0	880.0	214.6
Minimum	45.0	520.0	9.4
Maximum	860.0	1400.0	224.0

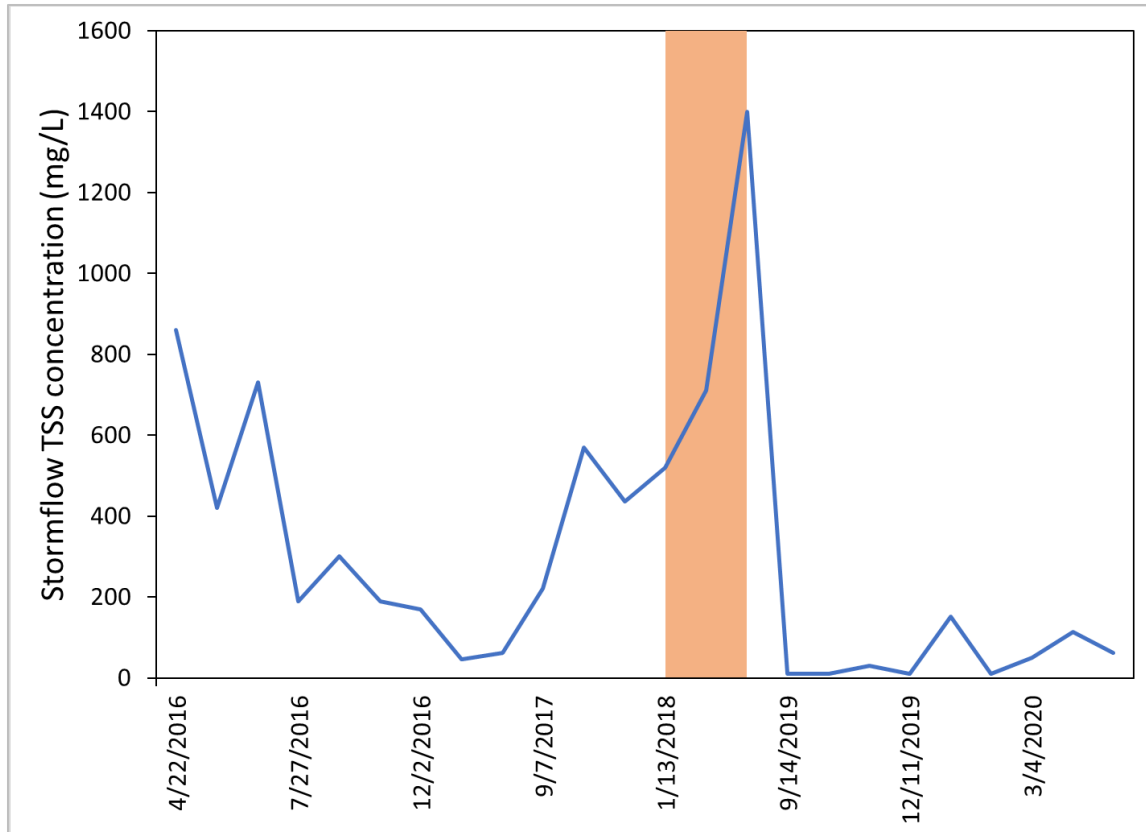


Figure 4.5 Stormflow TSS concentration (mg/L) at the watershed outlet. Orange bar represents the during-restoration period. Data represent the storms sampled during the period from 2016-2021.

A correlation analysis was complete between stormflow TSS concentration and 48-hour antecedent moisture in the pre- and post-restoration periods. Results indicate that there was a non-significant, slightly positive relationship $r(4)$, 0.45, $p = 0.38$ (Figure 4.6) during the pre-restoration period. During the post-restoration period, results indicate that there was a non-significant, slightly negative relationship $r(4)$, -0.36, $p = 0.49$ (Figure 4.7).

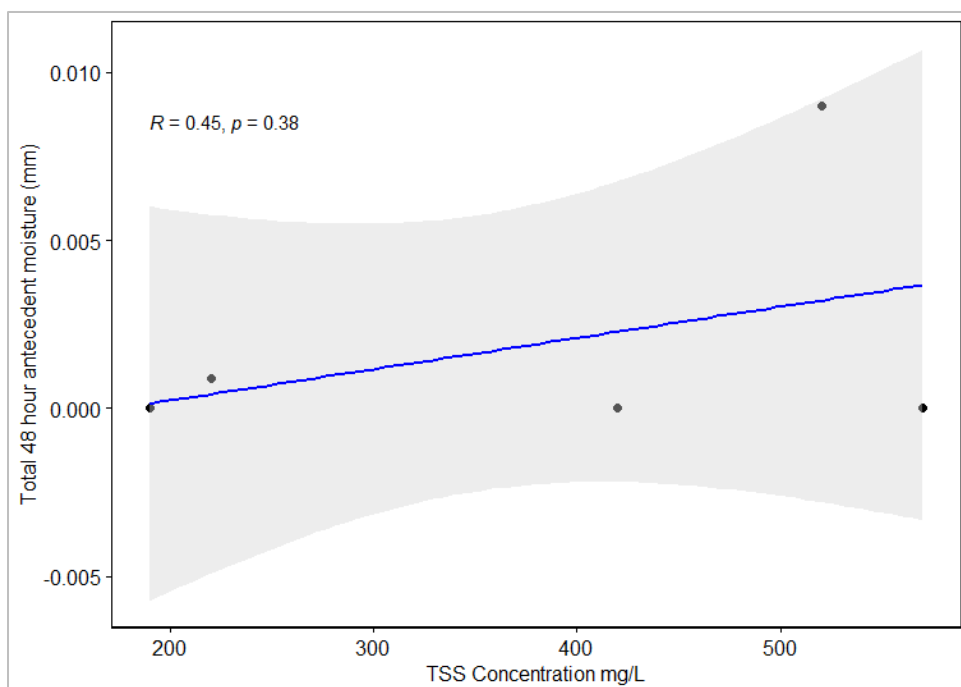


Figure 4.6 Correlation between TSS concentration (mg/L) and total antecedent precipitation (mm) in the pre-restoration period at the watershed outlet. Data represent the storms sampled during the period from 2016-2017.

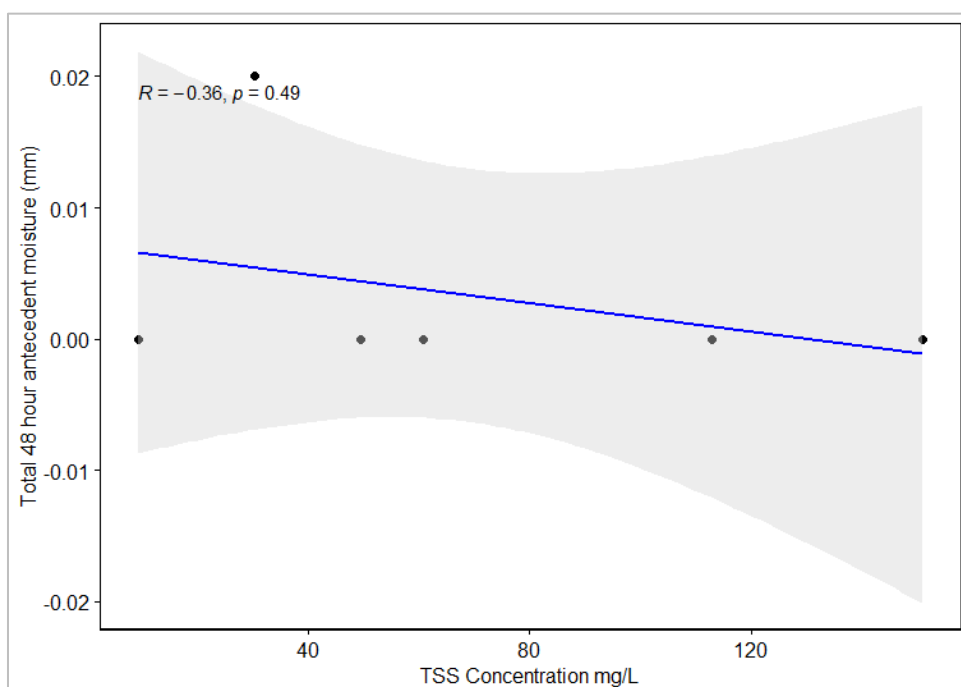


Figure 4.7 Correlation between TSS concentration (mg/L) and antecedent precipitation (mm) in the post-restoration period at the watershed outlet. Data represent the storms sampled during the period from 2019-2021.

4.4.1.3 Stormflow Flux

Overall, there was a decrease in per storm TSS flux (kg) from a pre-restoration average of 1880 kg to a post-restoration average of 1190 kg. Pre-restoration, average per storm TSS flux was higher (1880 kg) and more variable (4310 kg) when compared to the post-restoration period (Table 4.6). Average per storm TSS flux was lower (1190 kg) and less variable (4090 kg) in the post-restoration period when compared to pre-restoration (Table 4.6). This decrease in per storm TSS flux (g), 690 kg, was not statistically significant, $W = 30$, $p > 0.23$. Overall, restoration led to a decrease in total storm TSS flux over time but was not a statistically significant decrease (Figure 4.8).

Here, the highest TSS flux (4840 kg) was moved during a high intensity, short-duration event which occurred pre-restoration (Table 4.11; 10/16/2017). All storms moving more than 1000 kg TSS ($N = 8$) were high intensity storms besides storms that occurs on 4/22/2016, 7/27/2016, and 1/25/2020 which moved 1650 kg, 1240 kg and 4100 kg and were classified as low intensity long, duration, low intensity, short-duration, and low intensity, long-duration storms respectively (Table 4.11). Storms moving less than 1 million g TSS ($N = 5$) were low intensity storms besides storms that occurred on 10/27/2019 and 4/01/2021 which moved 2.06 kg and 899 kg respectively and were both classified as high intensity, short-duration storms. The storm occurring on 8/18/2016 was the only pre-restoration storm to move less than 1000 kg TSS (530 kg).

Table 4.6 Descriptive statistics for per storm TSS flux (kg) in the pre-, during, and post-restorations periods at the watershed outlet. Data represent the storms sampled during the period from 2016-2021.

Summary Statistic	Pre-Restoration	During Restoration	Post-Restoration
Mean	1880	5940	1190
Standard Error	614	2350	571
Median	1500	4430	899
Mode	/	/	/
Standard Deviation	1500	4080	1510
Range	4310	7720	4090
Minimum	530	2830	2.06
Maximum	4840	10600	4100

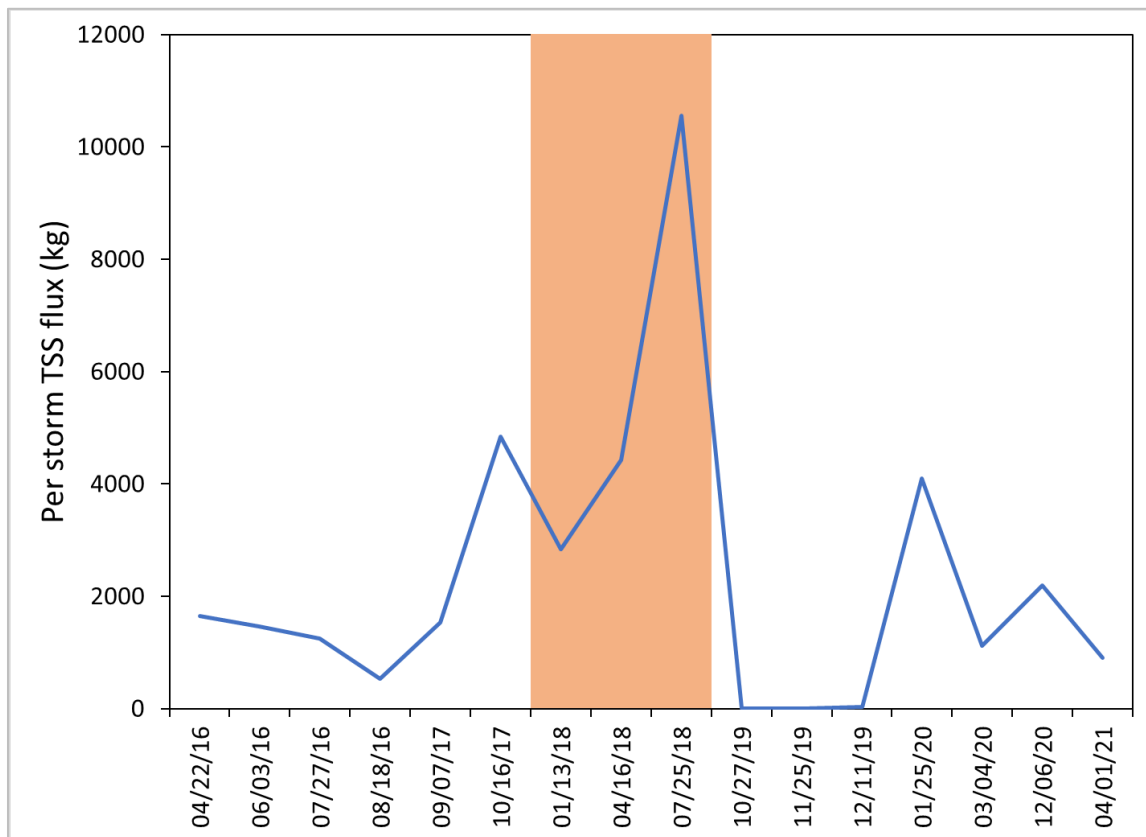


Figure 4.8 Per storm TSS (kg) dynamics for all storms at the watershed outlet. Orange bar represents the during-restoration period. Data represent the storms sampled during the period from 2016-2021.

A correlation analysis was complete between stormflow TSS flux and 48-hour antecedent moisture for both the pre-restoration and post-restoration periods. Results

indicate that there was a non-significant, slightly negative relationship $r(5)$, -0.11 , $p = 0.84$ (Figure 4.9) during the pre-restoration period. During the post-restoration period, results indicate that there was a non-significant, slightly negative relationship $r(5)$, -0.29 , $p = 0.53$ (Figure 4.10).

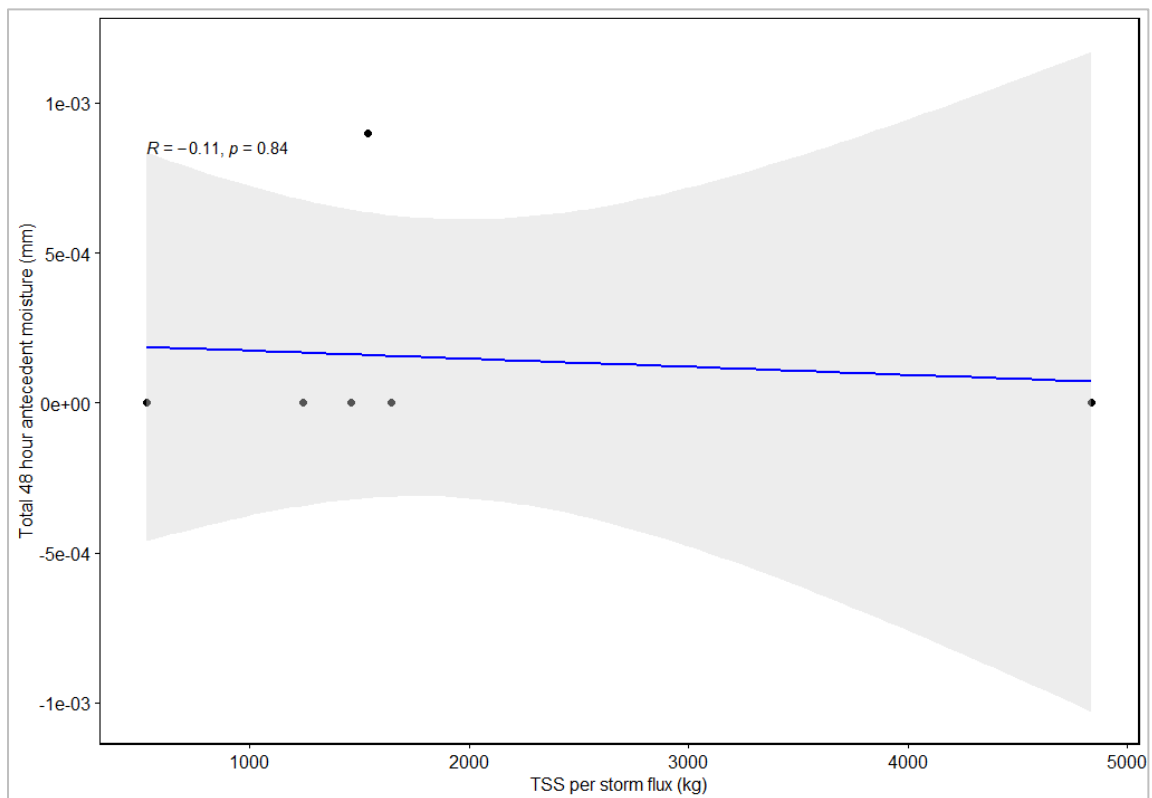


Figure 4.9 Correlation analysis for TSS per storm flux (kg) and 48-hour antecedent precipitation (mm) in the pre-restoration period at the watershed outlet. Data represent the storms sampled during the period from 2016-2017.

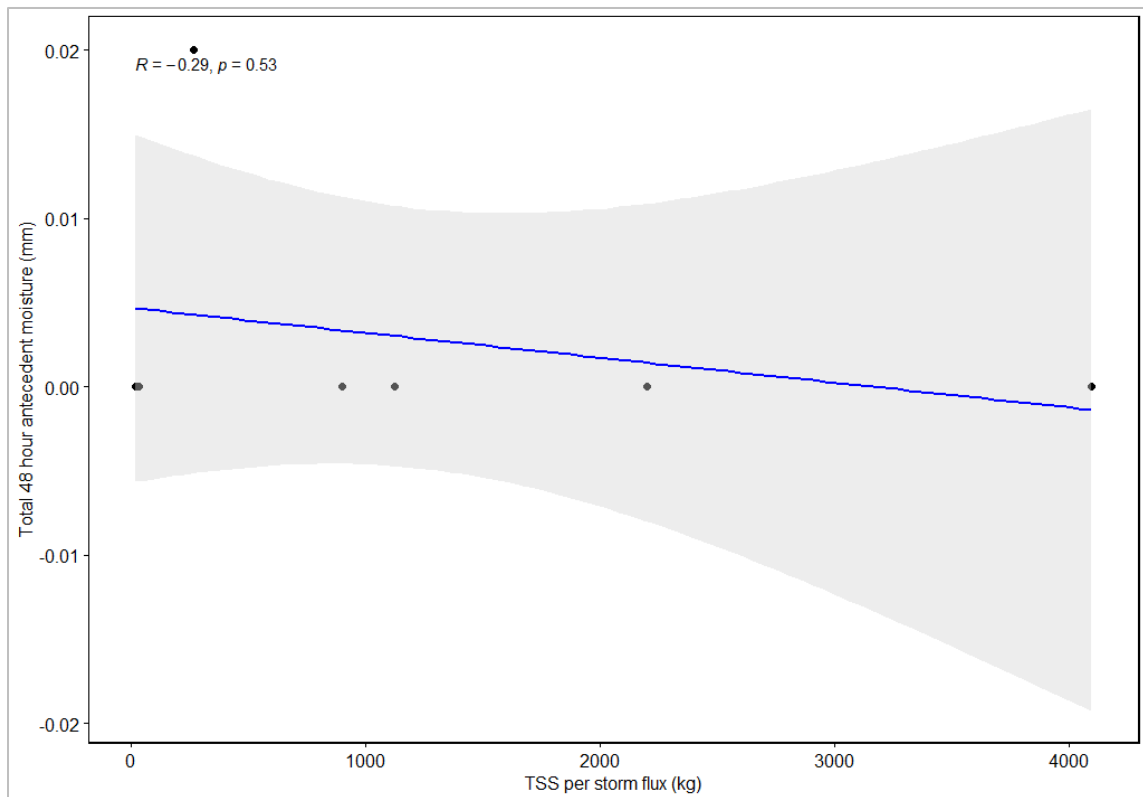


Figure 4.10 Correlation analysis for TSS per storm flux (kg) and 48-hour antecedent precipitation (mm) in the post-restoration period at the watershed outlet. Data represent the storms sampled during the period from 2019-2021.

4.4.1.4 Watershed Changes in Stormflow TSS Dynamics

Watershed stormflow dynamics shifted from pre-restoration to the post-restoration period. The per storm discharge and storm duration increased in the post-restoration period (Table 4.7). Other parameters such as per storm precipitation, TSS concentrations, and TSS flux decreased in the post-restoration period (Table 4.7). Overall, the post-restoration watershed exhibited storms which produced less precipitation per storm over longer time periods resulting in increased storm discharge and a reduction in TSS concentrations and flux. These findings are consistent with the hypotheses (RQ4.1).

Table 4.7 Storm characteristics changes from pre- to post-restoration at the watershed outlet. Asterisk * indicates statistically significant data ($p < 0.05$). Data represent the storms sampled during the period from 2016-2021.

Storm Characteristics	Direction of Change from Pre- to Post-Restoration	Significance
Storm Runoff (Q m ³)	Increased	/
Duration	Increased	/
Storm Precipitation (mm)	Decreased	/
TSS Storm Concentration	Decreased	*
TSS Storm Flux	Decreased	/

4.4.2 (RQ4.2) Does restoration impact TSS concentration and flux at baseflow during pre- and post-restoration periods?

4.4.2.1 Baseflow Concentrations

Baseflow TSS concentrations were compared before and after restoration. Data indicate that baseflow TSS concentrations decreased from a pre-restoration average of 19.7 mg/L to a post-restoration average of 7.3 mg/L (Table 4.5). Results from a Mann-Whitney test indicate that this decrease in baseflow TSS concentrations, 12.4 mg/L, was statistically significant $W = 1008$, $p < 0.02$. Average baseflow TSS highest (19.7 mg/L) and most variable (81.3 mg/L) in the pre-restoration period. Post-restoration, average baseflow TSS was lowest (7.3 mg/L) and less variable (36.3 mg/L) when compared to the pre-restoration period. Overall, restoration led to a decreased in baseflow TSS concentrations over time (Figure 4.11). Moreover, yearly, 2016 had the highest average baseflow TSS concentration (23.0 mg/L) followed by 2018 (restoration year; 13.0 mg/L). Overall, 2020 has the lowest average baseflow TSS concentration at 5.0 mg/L. Here, one data point collected on 06/04/2019 was removed from the post-restoration dataset on as it was 1084.5 mg/L which is too high to have been validly recorded.

Table 4.8 Descriptive statistics for baseflow TSS concentrations (mg/L) in the pre-, during, and post-restoration periods at the watershed outlet. Data represent the sampling period from December 2015 – March 2021.

Summary Statistic	Pre-Restoration	During Restoration	Post-Restoration
Mean	19.7	12.6	7.3
Standard Error	3.2	1.5	1.3
Median	10.7	10.2	5.0
Mode	/	-	5.0
Standard Deviation	21.4	6.4	7.5
Range	81.3	19.7	36.3
Minimum	1.1	4.0	1.0
Maximum	82.4	23.8	37.4

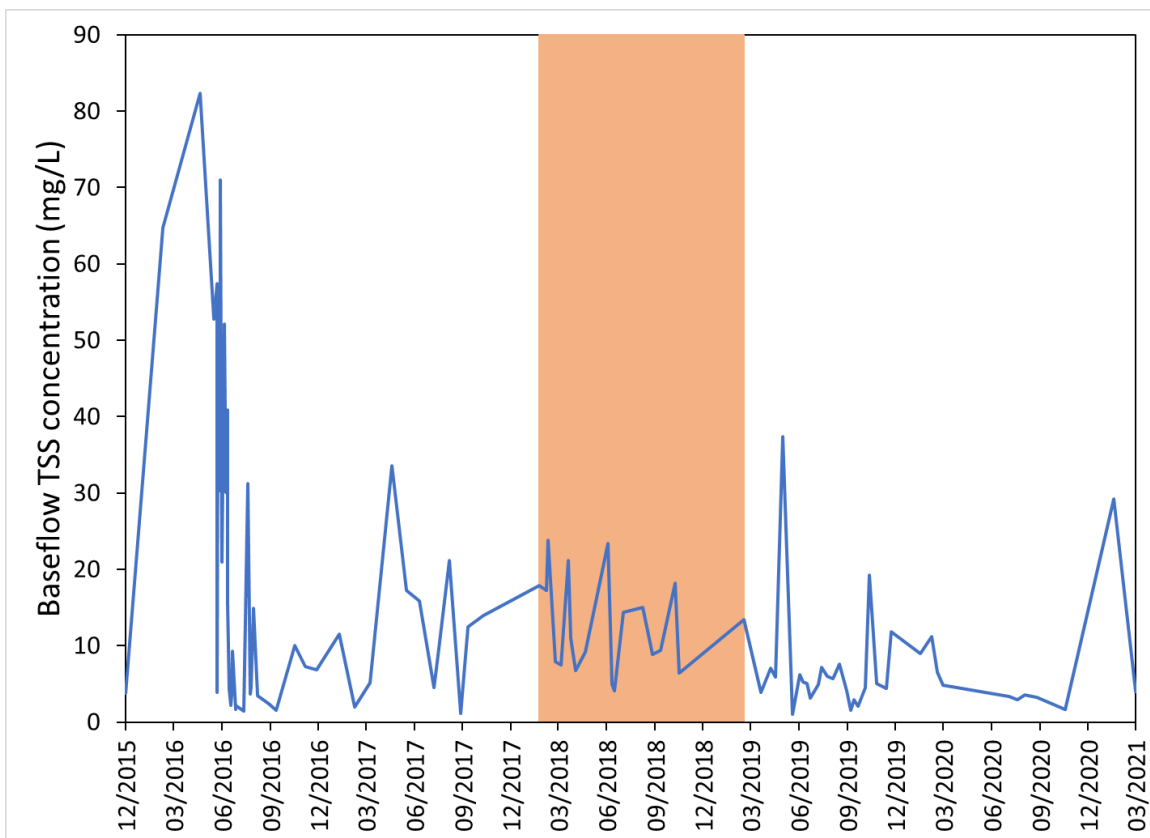


Figure 4.11 Baseflow TSS concentration (mg/L) at the watershed outlet. The during-restoration period is indicated with orange bar. Data represent the sampling period from December 2015 – March 2021.

4.4.2.2 Baseflow TSS Flux

Overall, baseflow flux (kg/ha/year) were larger during the pre-restoration period when compared to the post-restoration period. Data indicate that baseflow flux (kg/ha/year) decreased from a pre-restoration average of 48.8 (kg/ha/year) to a post-restoration average of 23.3 (kg/ha/year) (Table 4.9). This decrease in baseflow TSS flux, 19.5 (kg/ha/year), was statistically significant, $W = 980$, $p < 0.04$. Average baseflow TSS (kg/ha/year) was higher (48.8 kg/ha/year) and less variable (246.7) than during the post-restoration period. Post-restoration, average baseflow TSS (kg/ha/year) was lower (29.3 kg/ha/year) and more variable (330.9 kg/ha/year) then during the pre-restoration period. Overall, restoration led to a decreasing pattern of baseflow TSS over time (Figure 4.12). Moreover, yearly average baseflow TSS flux (kg/ha/year) was highest during pre-restoration in 2016 (56.9 kg/ha/year) followed by 2019 (28.6 kg/ha/year) in the post-restoration period. Overall, 2020 had the lowest average baseflow TSS flux (13.9 kg/ha/year).

Table 4.9 Descriptive statistics for baseflow TSS flux (kg/ha/year) in the pre-, during, and post-restoration periods at the watershed outlet. Data represent the sampling period from December 2015 – March 2021.

Summary Statistic	Pre-Restoration	During Restoration	Post-Restoration
Mean	48.8	14.6	29.3
Standard Error	9.8	3.0	11.3
Median	11.8	11.8	6.1
Mode	/	/	/
Standard Deviation	65.0	12.7	66.9
Range	246.7	45.0	330.9
Minimum	1.2	1.1	0.5
Maximum	247.9	46.0	331.4

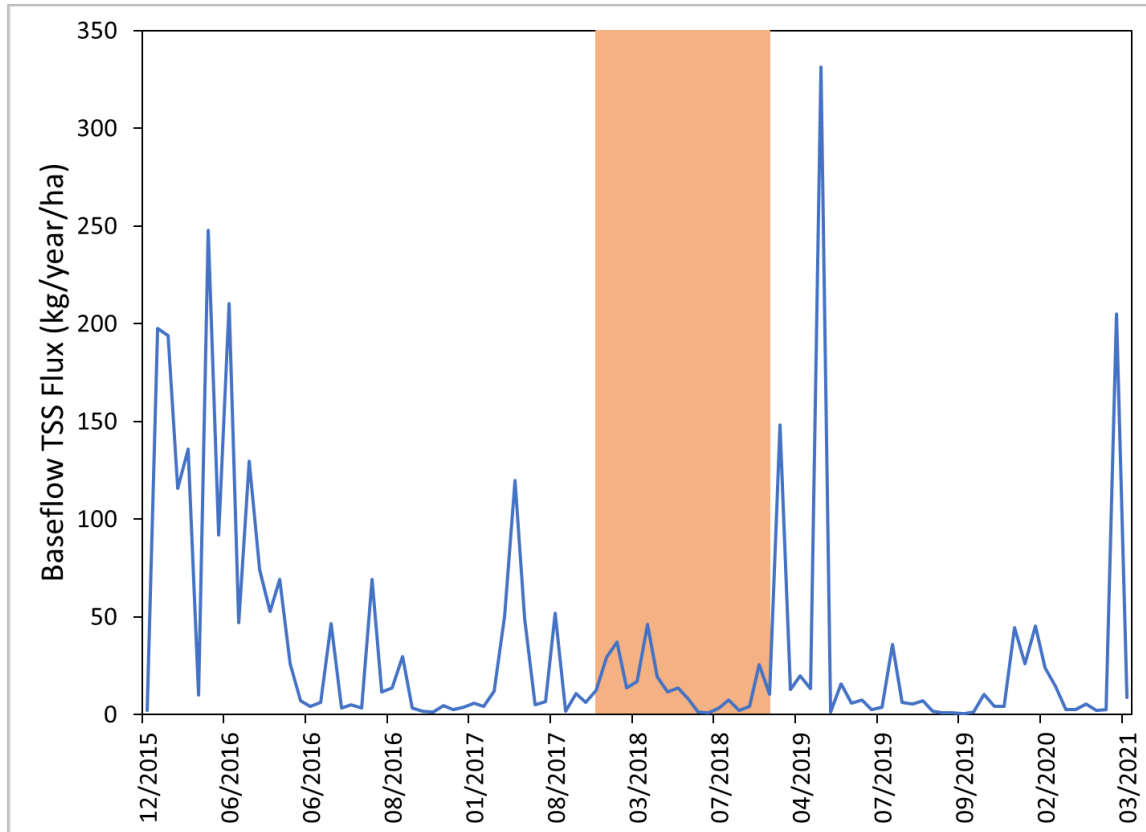


Figure 4.12 Baseflow TSS Flux (kg/ha/year) in the pre-, during, and post-restoration period at the watershed outlet. During restoration period is indicated by orange bar. Data represent the sampling period from December 2015 – March 2021.

4.4.2.3 Watershed Changes in Baseflow TSS Dynamics

Overall, baseflow TSS dynamics changed from pre-restoration levels to post-restoration levels. TSS concentrations and flux both decreased statistically in the post-restoration period (Table 4.10). Furthermore, results from RQ2.3 indicate that baseflow Q increased in the post-restoration period.

Table 4.10 Baseflow characteristic changes from pre- to post-restoration at the watershed outlet. Asterisk * indicates statistically significant ($p < 0.05$) data. Data represent the sampling period from December 2015 – March 2021.

Baseflow Characteristics	Direction of Change from Pre- to Post-Restoration	Significance
Baseflow Concentration	Decreased	*
Baseflow Flux	Decreased	*

4.5 Discussion

4.5.1 RQ4.1

RQ4.1 set out to determine the impact restoration had on stormflow TSS concentration and flux dynamics. Mann-Whitney tests resulted in a statistically significant difference in average stormflow TSS concentration before and after restoration which suggests that the restoration approach used within the Reedy Creek watershed was successful at reducing stormflow TSS concentration during the period of the study. Here, recreating stream channels in such a way that incorporated increased sinuosity and the addition of riffle-pool sequences may have led to decreasing erosional forces during stormflow. Moreover, a shift from sand to clay dominated bank and beds may have further decreased erosional forces by increasing the shear stress needed to dislodge, move, and suspend bank and bed materials. Overall, restoration led to a more stable stream.

Storm dynamics did vary from the pre- to post-restoration periods, however, this variation was not significant and may not have contributed substantially to changes in TSS dynamics. Moreover, the increase in stormflow accompanying the decrease in per-storm precipitation from pre- to post-restoration may be attributed to an increase in throughfall as canopy cover along the streams was reduced due to clear-cutting allowing for more water to enter the stream than during the pre-restoration period. A correlation analysis between stormflow TSS concentrations and 48-hour antecedent precipitation resulted in a non-significant, positive relationship indicating that, to a certain extent, increases in stormflow TSS concentrations were driven by increases in 48-hour antecedent moisture during the pre-restoration period. A post-restoration correlation

analysis of the same parameters resulted in a non-significant, negative relationship indicating that increases in TSS concentrations may be slightly related to decreases in 48-hour antecedent precipitation. For TSS stormflow flux, Mann-Whitney test results were not statistically significant between the pre-restoration and post-restoration periods. However, reductions in storm TSS flux did occur indicating that restoration, while not statistically significant, was successful at reducing flux within the watershed even with higher runoff. Overall, post-restoration exhibited statistically significantly lower TSS concentration but no statistically significant difference in TSS flux. Trends suggest that the watershed TSS concentration dynamics shifted from pre-restoration where concentrations were high to post-restoration where concentrations were low. This may indicate a decrease in channel and floodplain erosion, alteration of sediment sources, increased infiltration, increased residence times, and/or a slight change in near-stream topography.

Storm dynamics changed from pre-restoration to post-restoration with non-significant increases in storm runoff indicating larger storms releasing more precipitation moved over the watershed in the post-restoration period. For both the pre-restoration and post-restoration periods, it is important to note that 48-hour antecedent moisture was zero for 12 out of 16 recorded storms. One storm from each timeframe (pre- and post-restoration) had small amounts of antecedent precipitation with the latter having more antecedent precipitation. However, this non-significant increase in precipitation did not lead to increased TSS exports, instead TSS decreased with an increase in precipitation in the post-restoration period.

Results from McMahon et al. (2021) indicate that restoration, at the small-scale (i.e.: restoration of sites was less than 1.2 km in length), was only slightly successful at decreasing sediment loads during storm events. This provides evidence that the scale of restoration is important for altering essential watershed functions such as sediment loads. Furthermore, future restoration projects need to take place on large scales (i.e.: greater than 1 km of stream length), include headwaters, and address upstream land-use practices so they can provide a buffer for downstream habitats as headwaters are extremely susceptible to nutrient pollution (Lavelle et al., 2019; Lewis et al., 1996).

4.5.2 RQ4.2

RQ4.2 set out to determine the impact restoration had on baseflow TSS concentration and flux dynamics. Mann-Whitney tests resulted in a statistically significant decrease in average baseflow TSS concentration before and after restoration which suggests that the restoration approach used within the Reedy Creek watershed was successful at reducing baseflow TSS concentration during the period of the study. This finding indicates that restoration has led to a stream system with less erosion. For baseflow TSS flux, results from a Mann-Whitney test indicated a statistically significant decrease in average baseflow TSS flux before and after restoration which suggests that the restoration approach used within the Reedy Creek watershed was successful at reducing baseflow TSS flux during the period of the study. These findings indicate that changes in watershed characteristics brought on by stream restoration were adequate to address baseflow TSS concentration and flux over time.

Overall, changes within the watershed from the pre-restoration to the post-restoration period led to a decrease in TSS concentrations during both baseflow and

stormflow events. Here, results indicate that stormflow Q increased (Table 4.7) and baseflow Q increased. These increases in Q may have led to a dilution effect of TSS within the watershed either through increased stormflow Q over longer periods or through increases in groundwater Q (Estrany et al., 2009). Pre-restoration elevated TSS flux may have been due to a slightly higher flashiness of the stream as stormflows in the heavily incised channels were unable to access surrounding floodplains, increased stream power and shear stress due to a lack of ‘natural’ morphological characteristics (i.e.: sinuosity, slope), and flushing of groundwater as precipitation infiltrated and left the watershed quickly as channels acted as groundwater seep points – all of which caused high levels of TSS concentrations and flux. Post-restoration decreases in TSS concentration and flux may be due to decreased flashiness as stormflows are able to access the floodplain to dissipate stream power and deposit sediments, decreased stream power and reduced shear stress induced by channel morphological changes, and increased groundwater storage due to elevated stream channels acting to buffer precipitation – all of which may lead to decreased instream erosion. However, the change in flashiness within the Reedy Creek watershed was slight and additional investigations should be completed to determine if slight changes in flashiness led to decreased TSS. Moreover, changes in land-use have been shown to alter stream suspended solid loads through increasing terrestrial and stream erosion (Houser et al., 2006; Shinomiya et al., 2020) and when those alterations are remedies, TSS loads should also decrease.

4.6 Conclusion

As mentioned in the literature review, streams are dynamic and complex as they work suspending, transporting, and depositing sediment loads downstream, in the channel

itself, and to surrounding floodplains (Bencala, 1993; Bencala et al., 2011; Filoso et al., 2015). Sediments are the building blocks of stream channels themselves, the landscapes surrounding the stream, and bring with them nutrients needed to support robust riparian habitat. As such, having a balanced sediment load (i.e.: where erosion and deposition rates are the same) in streams is essential for ensuring a functioning ecosystem (Simon et al., 2004; Holden, 2014). Moreover, as sediments are a leading form of surface water quality degradation (Simon et al., 2004), it is essential that issues with sediments be addressed and, in degraded streams, restoration may be used to alter sediment dynamics. Here, this study aimed to better understand how restoration of streams at the watershed-level altered stream TSS concentrations and flux during both stormflow events and at baseflow. According to findings within the Reedy Creek watershed, restoration was found to be successful at reducing TSS concentration and flux during both baseflow and stormflow events.

This study is useful for better understanding the TSS dynamics within the Reedy Creek urban restored watershed over a two-year pre-post period. Results from this study highlight that large scale (i.e.: 12 km) restoration projects in which streams are reconfigured and reconnected to their surrounding floodplains are effective at decreasing TSS concentrations and flux at baseflow and stormflow. Variability of future climate enhances the need for continued monitoring to determine if succeeding climates will influence restoration's success in the future. Future research could focus on different flow regimes and precipitation events of varying sizes to get a more holistic understanding of how the newly restored watershed functions under high-stress events.

Limitations of this study include the methodology used for stormflow. Here, samples were collected using an ISCO automated composite sampler located at the watershed outlet adjacent to the USGS gauging station. Samples were flow-weighted with increasing frequency of pulls taking place at higher flows with limited capacity to store large flow event samples. Samples were collected from the ISCO monitoring station and taken by CMSWS to a laboratory for analysis. According to CMSWS, in most cases, stormwater samples were retrieved on the day of the event. All stormflow samples were collected and processed by CMSWS and their affiliate labs. A further limitation of this study is that ISCO sampling was not consistent for all parameters throughout the study period. Several storms were removed from the dataset due to missing parameters such as storm size, duration, and TSS. Therefore, future stormflow monitoring should take place and should be conducted with more than 1 ISCO to ensure that all storm parameters are captured and should include backup Q and precipitation collection stations. Additionally, there was a small number of storms with varying amounts of precipitation which may have inhibited the analysis.

A further limitation of this study is that there was not a control study site to compare results from to determine if changes in channel morphology were responsible for alterations of stormflow TSS concentrations and flux. However, as storm dynamics did not change significantly, this may provide additional evidence that the channel reconfiguration has improved TSS dynamics within the watershed. Similarly, results from the neighboring Beaverdam site also indicate restored streams may export less TSS flux than their urban stream counterparts (Gagrani, 2013). Findings such as these can inform restoration practitioners on the efficacy of using the Rosgen natural channel design

restoration approach for altering stormflow TSS concentration and flux. By altering the stream channel and adjacent floodplain, restoration should encourage stormflow sediment storage while reducing sediment erosion and TSS flux.

Furthermore, future research could focus on TSS differences within the individual tributaries to determine which land-use type may be contributing the most TSS. Focusing on the tributaries for more fine-grained analysis may help pin-point sediment sources. Over the duration of the project, several tributaries experienced blowouts along the banks in the post-restoration period that may have been captured in weekly data collected at the watershed outlet.

Understanding how restoration can affect TSS loads is important as TSS impacts all aspects of the stream ecosystem including nutrient processing, food chains, vegetation/light interactions, and water quality. Overall, stream restoration at the watershed level is important for ensuring streams are stable, neither eroding nor aggrading excessively. To do so, watershed managers should identify and prioritize impaired waterways in headwater systems for restoration, create restoration goals and approaches that align with those goals, implement various restoration approaches known to benefit those goals, and monitor water quality using testable hypotheses for an extended period of time after restoration is completed to determine if restoration was successful. Restoration of stream ecosystems is imperative if we are to protect water quality with regard to sediment dynamics.

4.7 Appendix A: Chapter 4 TSS Data for Selected Storms

Table 4.11 Summary data for TSS and stormflow dynamics study 3 for selected storms in the pre- and post-restoration periods at the R1 (watershed outlet) site. Data represent the sampling period from April 2016 – April 2021.

Date	48-hour antecedent precipitation (mm)	Lag Time (minutes) Peak 1	Lag Time (minutes) Peak 2	Lag Time (minutes) Peak 3	Average lag time (minutes)	Storm duration (minutes)	Storm category
04/22/16	0	65			65	360	low intensity long-duration
06/03/16	0	50			50	35	high intensity short-duration
07/27/16	0	120			120	45	low intensity short-duration
08/18/16	0	55	85		70	195	low intensity short-duration
09/07/17	0.0009	90			90	335	high intensity long-duration
10/16/17	0	90	140		115	165	high intensity short-duration
01/13/18	0.009	100			100	285	high intensity short-duration
04/16/18	0	90	100		95	650	high intensity long-duration
07/25/18	0.07	55			55	290	high intensity short-duration
10/27/19	0	240	170	115	175	125	high intensity short-duration
11/25/19	0.02	95	50		72.5	655	low intensity long-duration
12/11/19	0	290	240	160	230	245	low intensity short-duration
01/25/20	0	90			90	750	low intensity long-duration
03/04/20	0	70	80		75	1060	high intensity long-duration
12/06/20	0	85			85	575	high intensity long-duration
04/01/21	0	115	195		155	300	high intensity short-duration

4.8 Appendix B: Chapter 4 Figures of Selected Storms

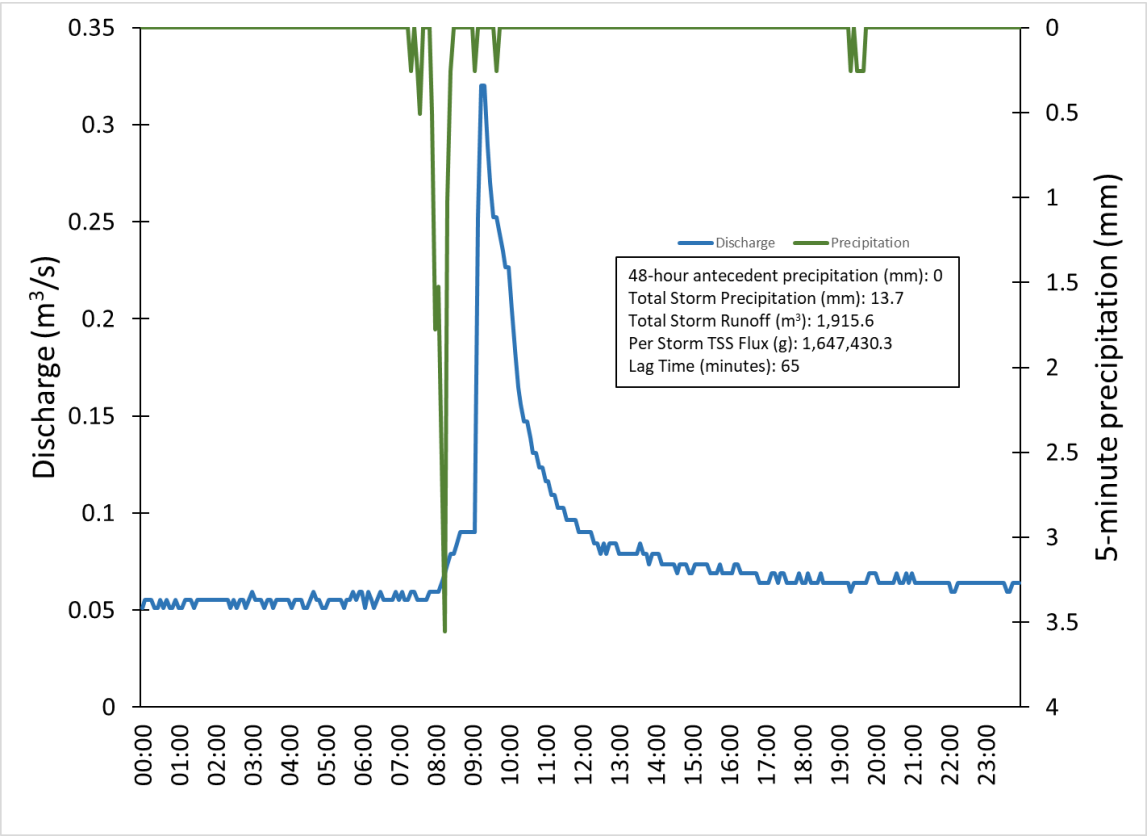


Figure 4.13 Pre-restoration storm from 4/22/2016 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

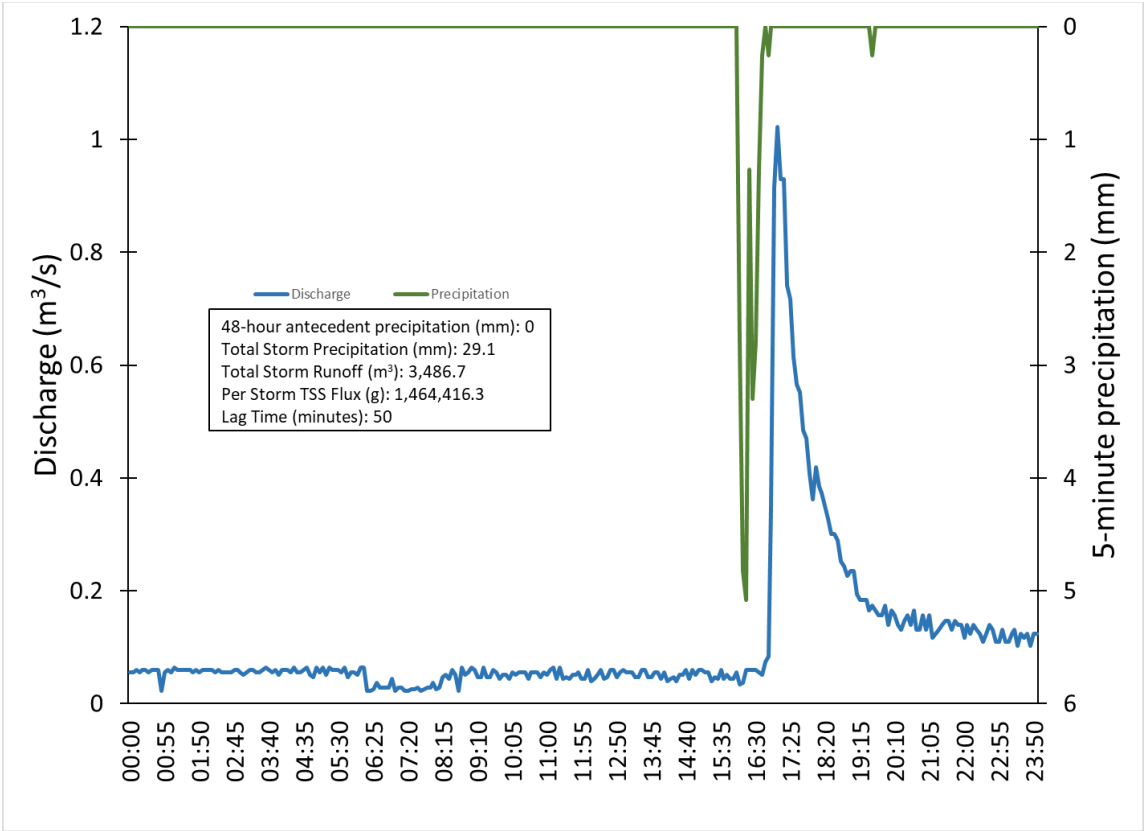


Figure 4.14 Pre-restoration storm from 6/03/2016 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

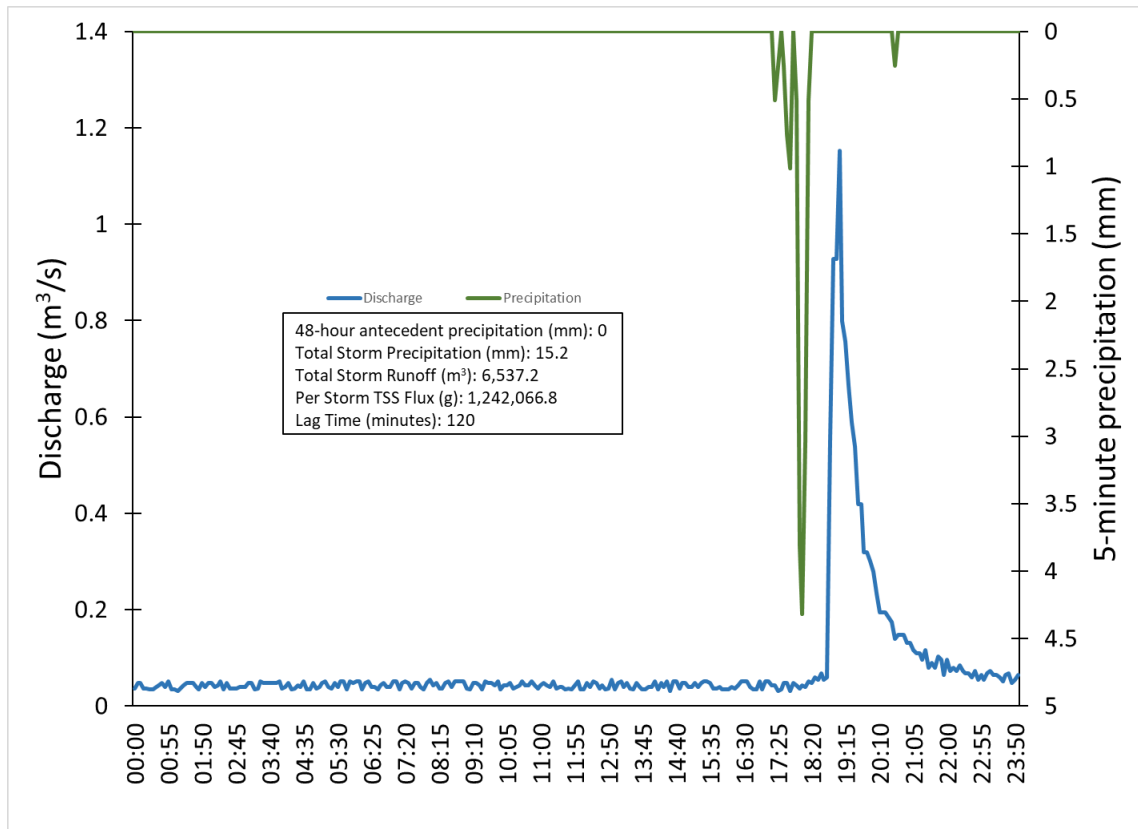


Figure 4.15 Pre-restoration storm from 7/27/2016 showing precipitation (mm) and discharge (m³/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m³), storm TSS flux (g), and lag times (minutes), are presented on figure.

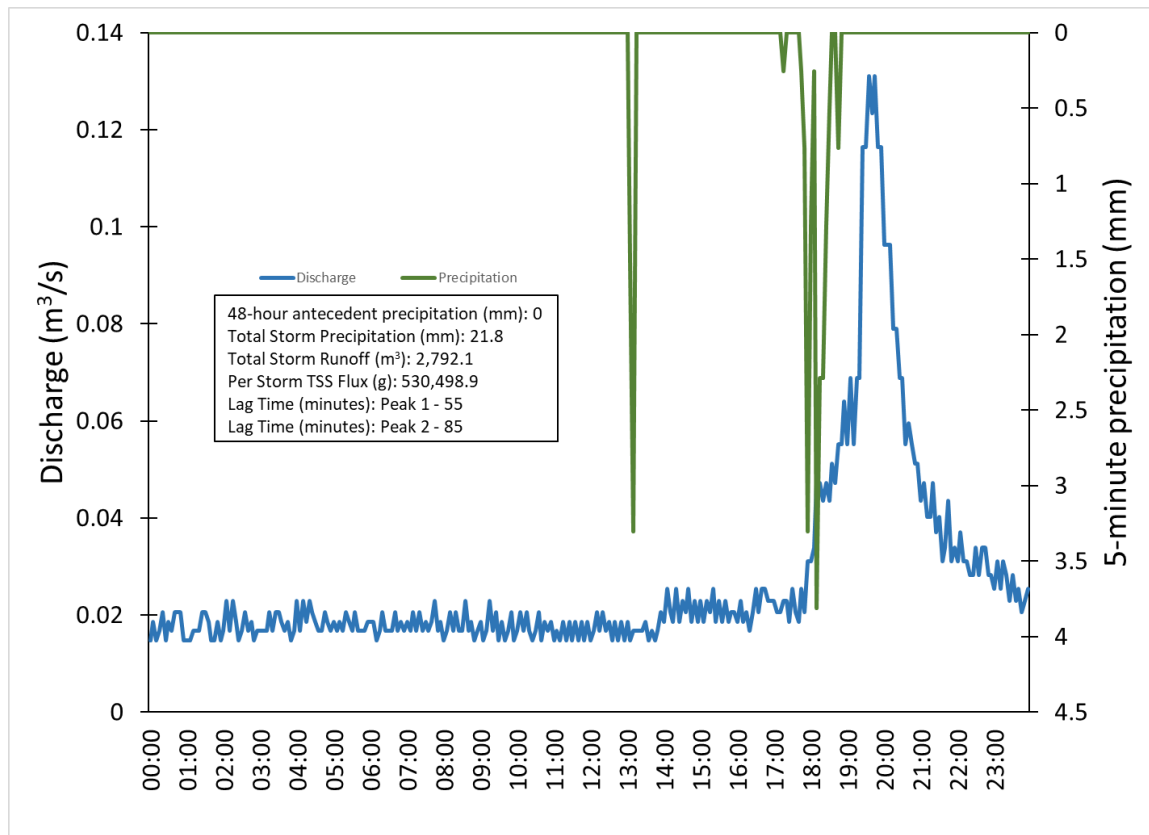


Figure 4.16 Pre-restoration storm from 8/18/2016 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

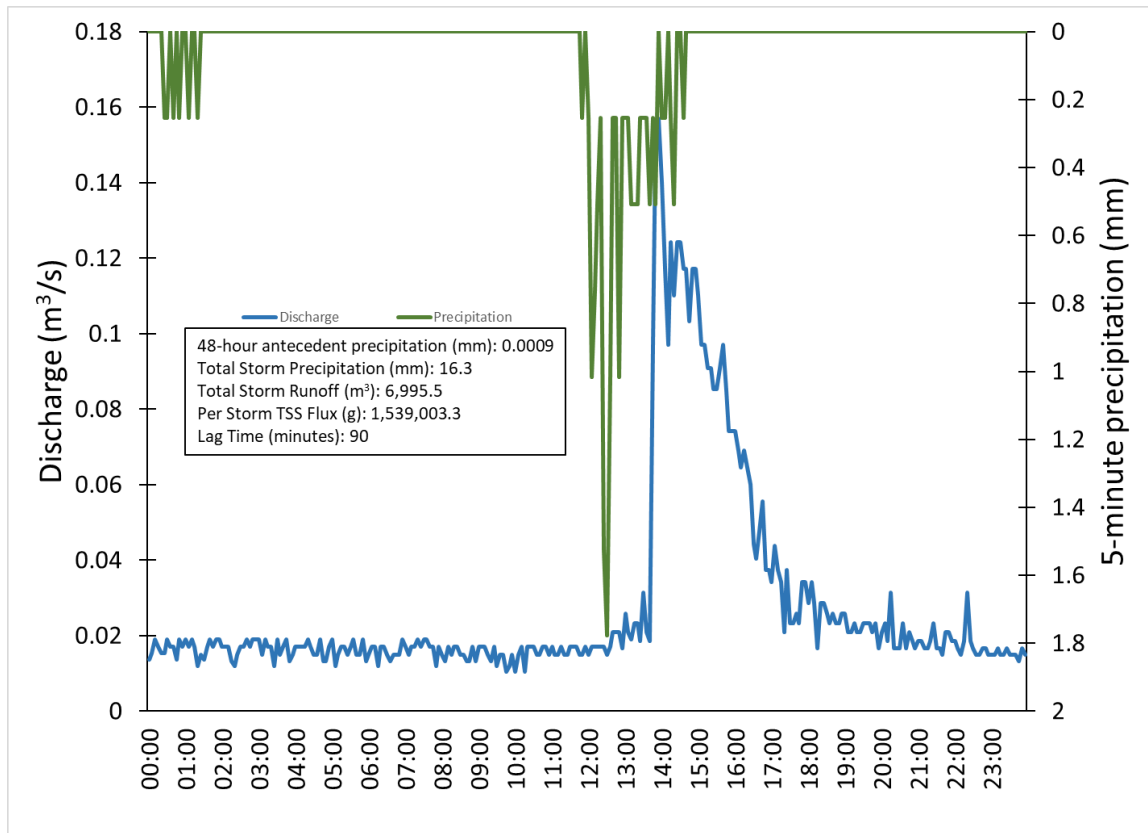


Figure 4.17 Pre-restoration storm from 9/17/2017 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

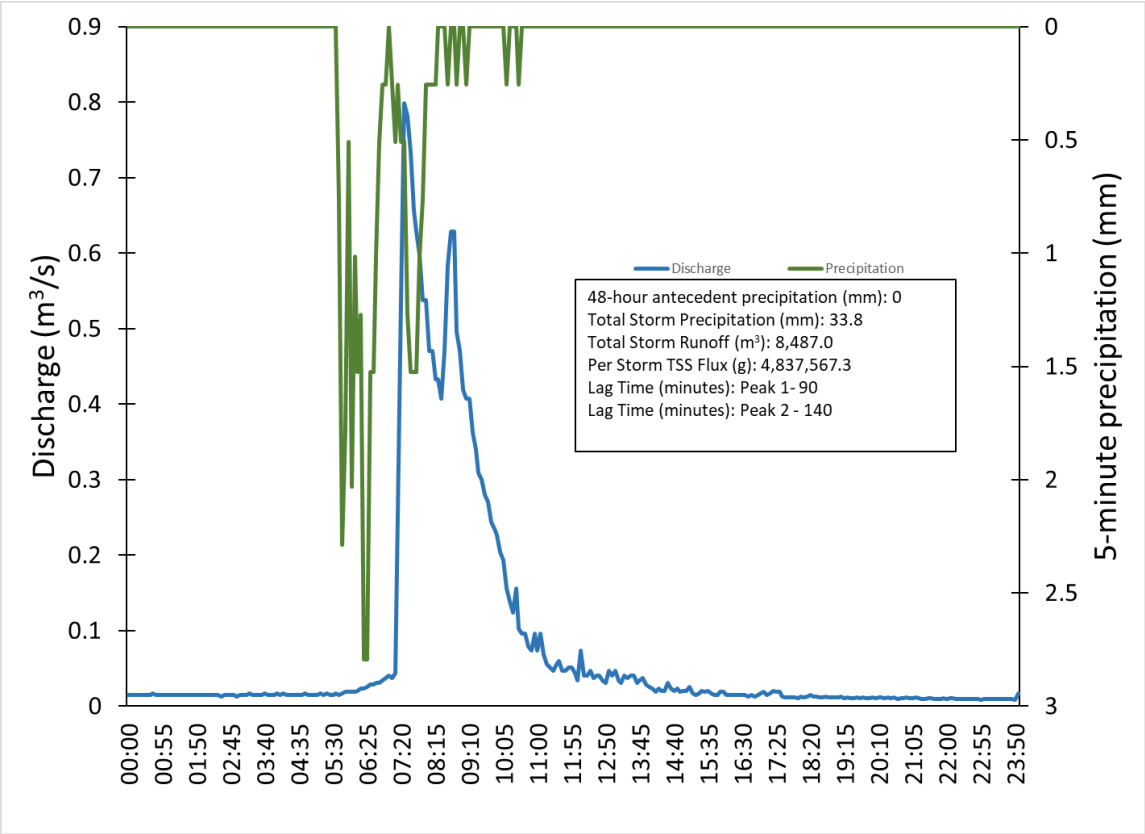


Figure 4.18 Pre-restoration storm from 10/16/2017 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

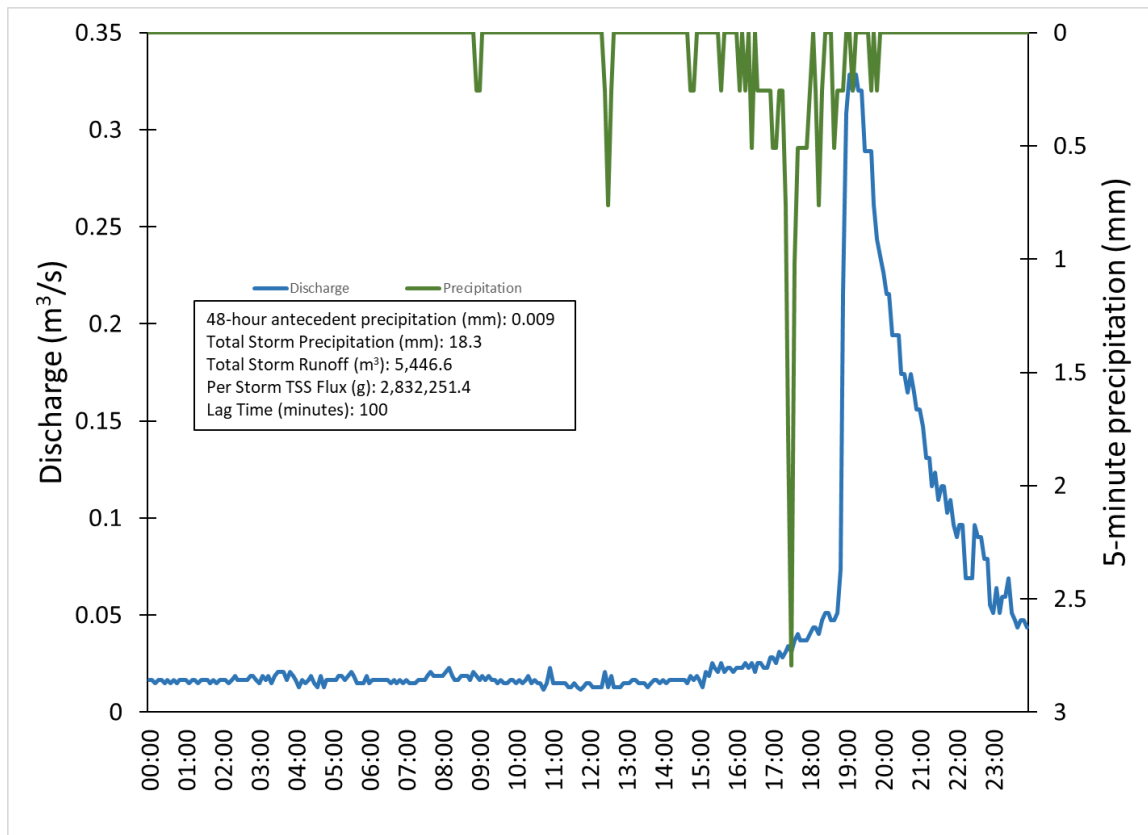


Figure 4.19 During-construction storm from 1/13/2018 showing precipitation (mm) and discharge (m³/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m³), storm TSS flux (g), and lag times (minutes), are presented on figure.

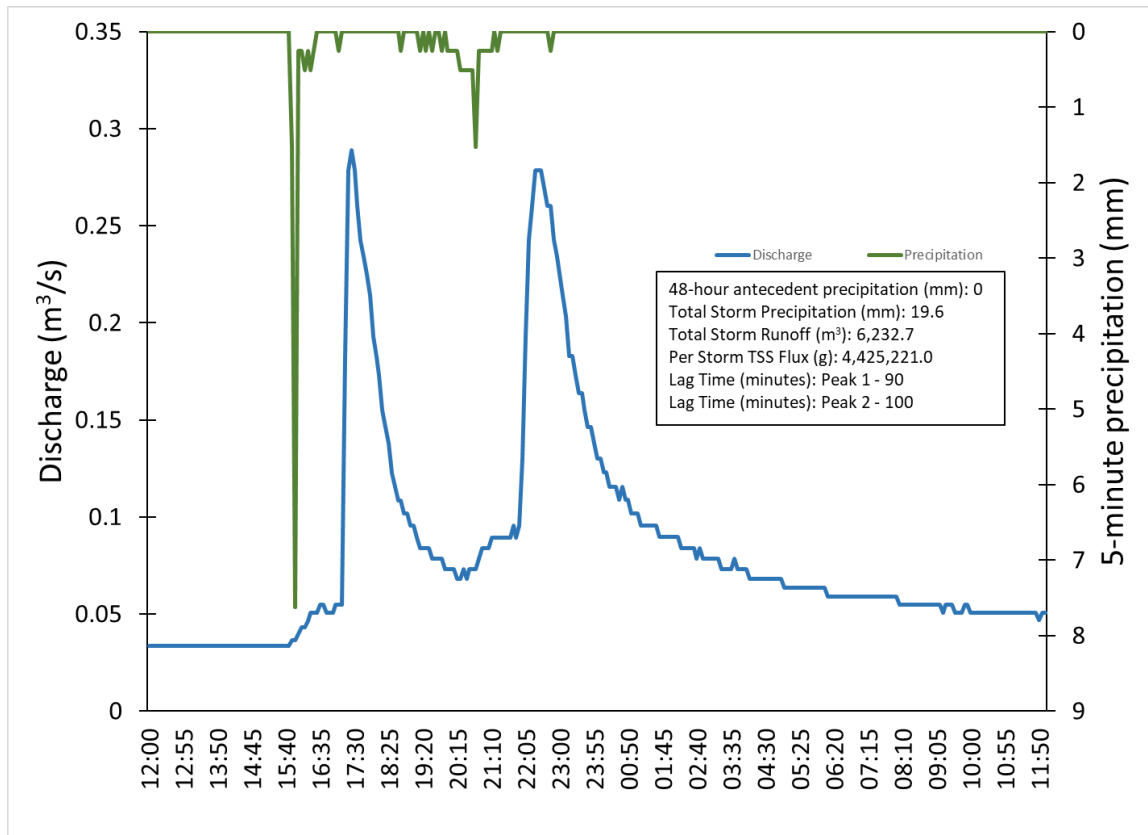


Figure 4.20 During-construction storm from 4/16/2018 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

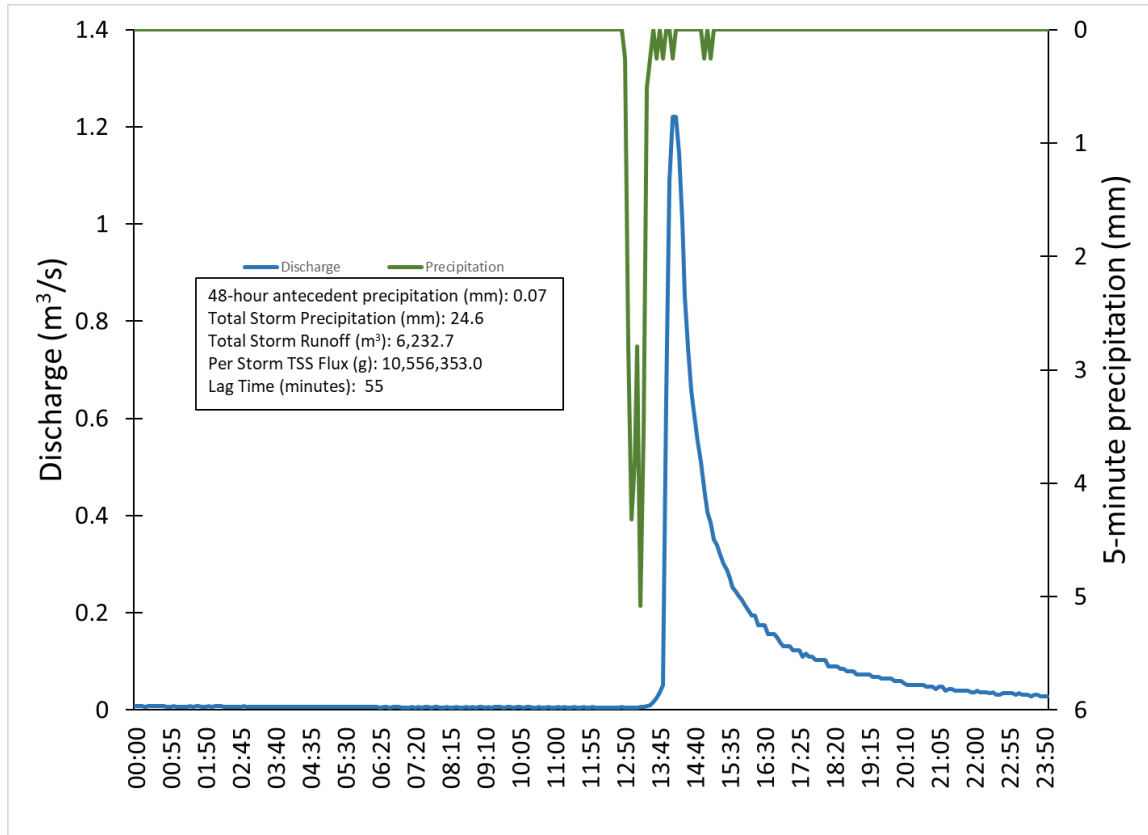


Figure 4.21 During-construction storm from 7/25/2018 showing precipitation (mm) and discharge (m³/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m³), storm TSS flux (g), and lag times (minutes), are presented on figure.

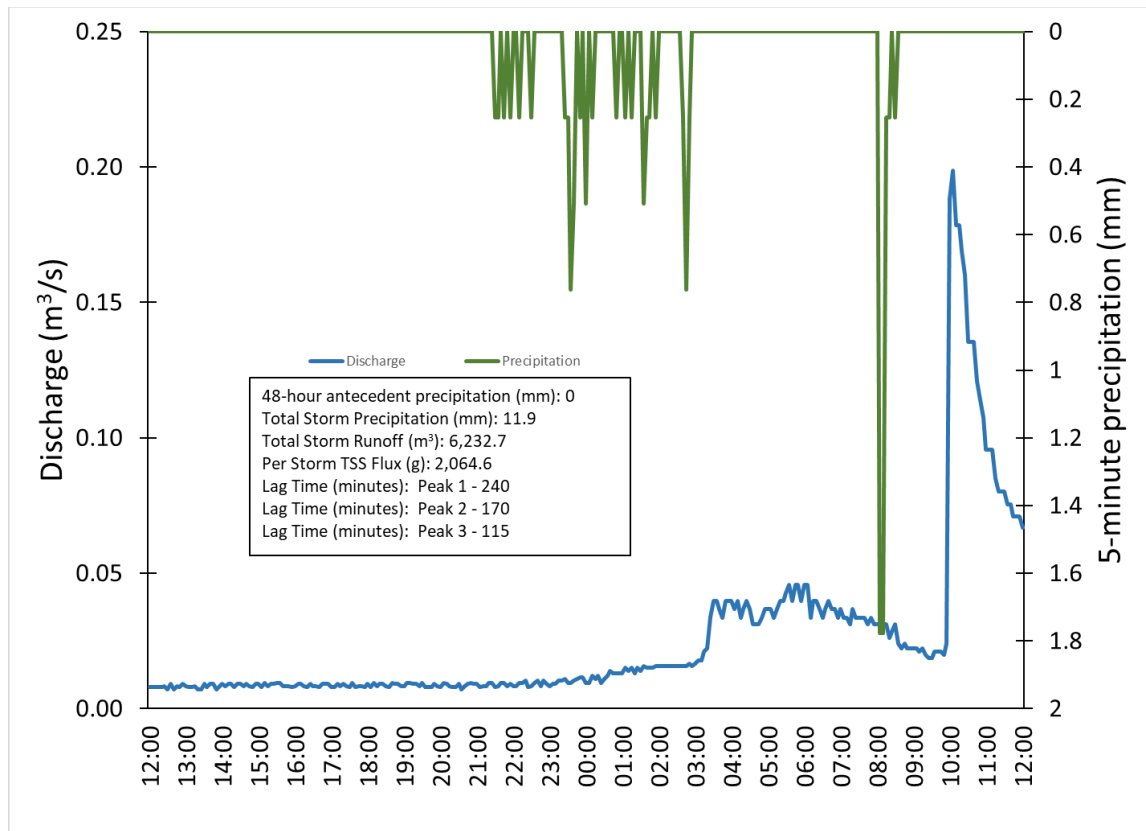


Figure 4.22 Post-restoration storm from 10/27/2019 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

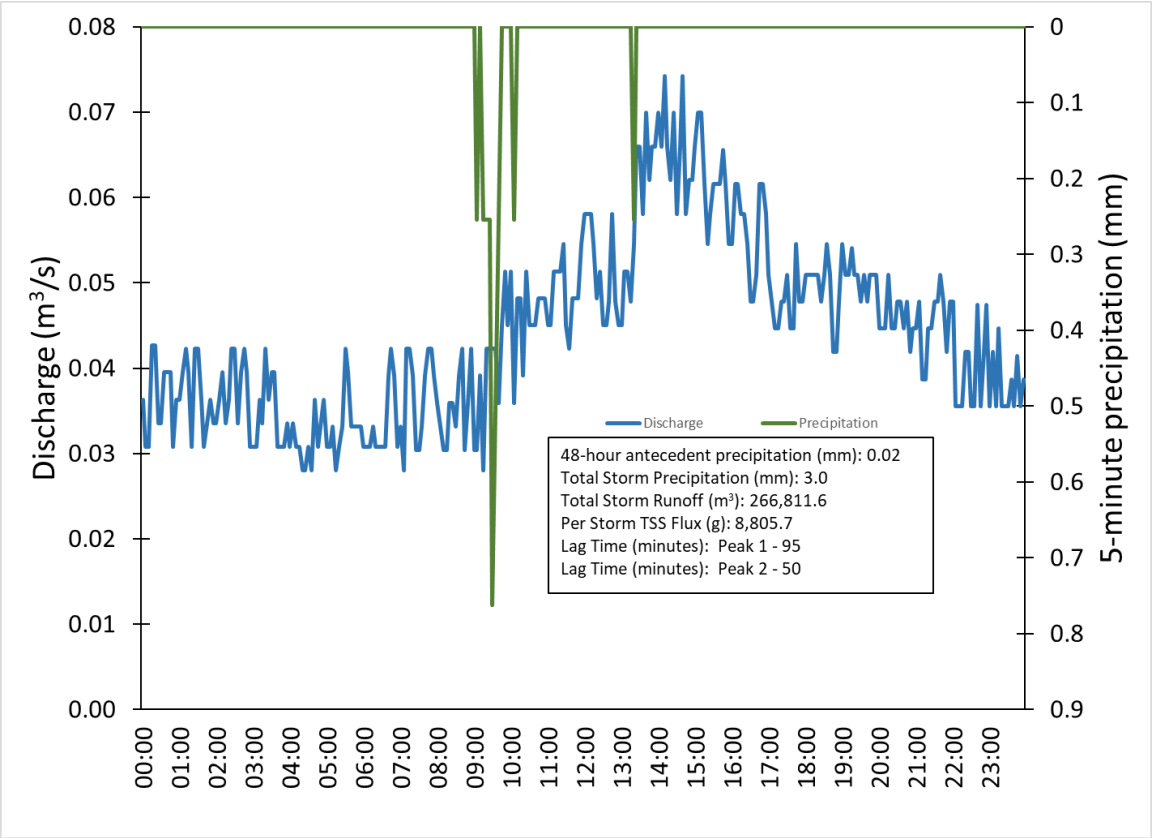


Figure 4.23 Post-restoration storm from 11/25/2019 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

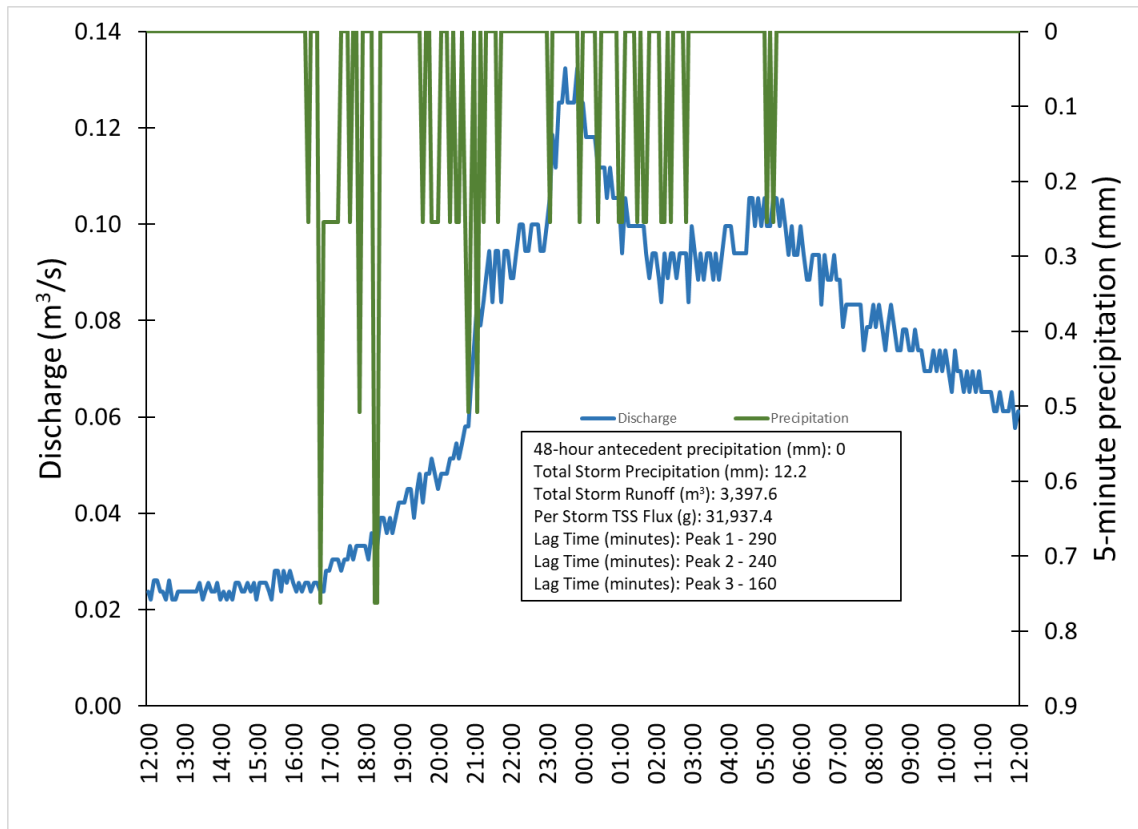


Figure 4.24 Post-restoration storm from 12/11/2019 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

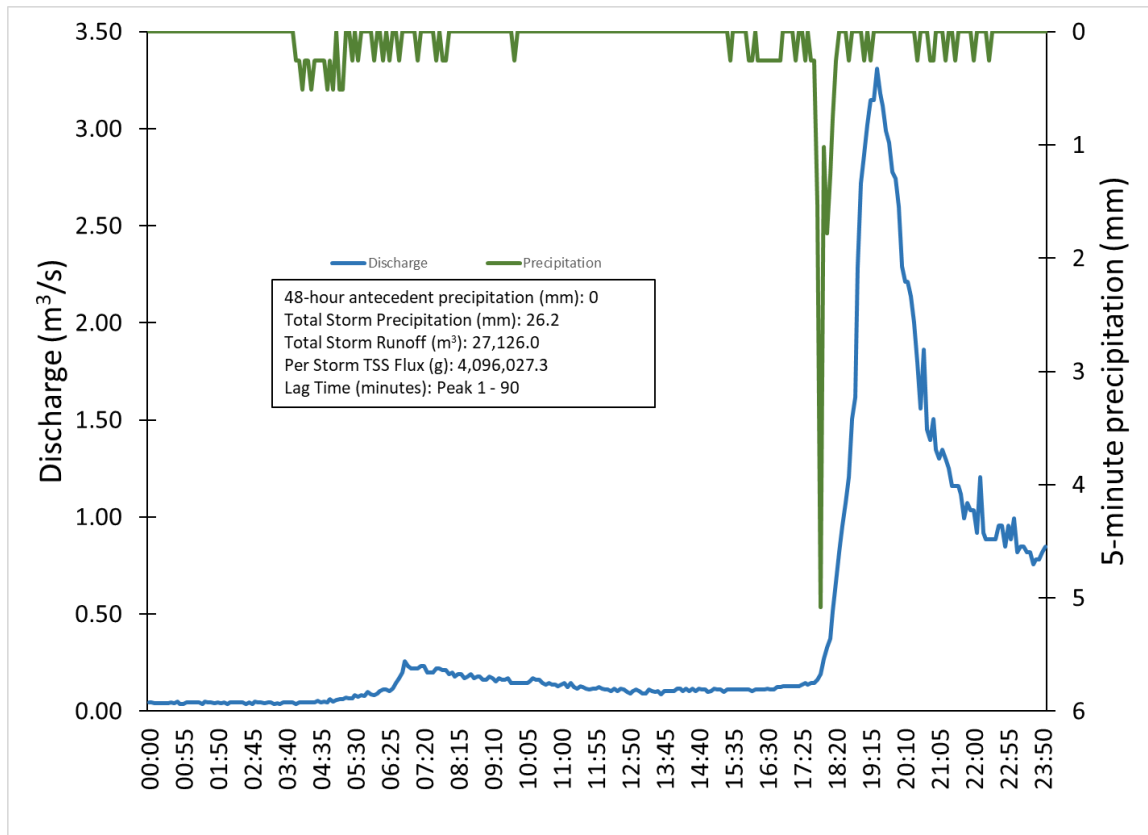


Figure 4.25 Post-restoration storm from 1/25/2020 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

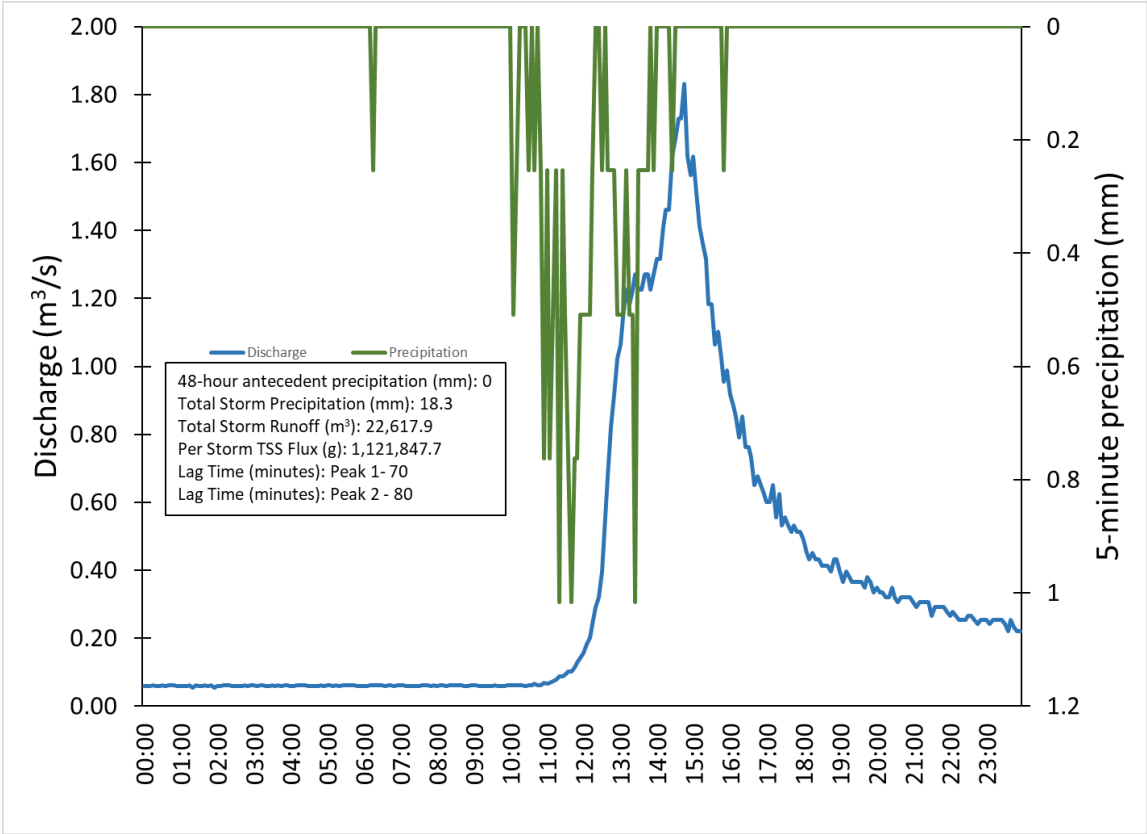


Figure 4.26 Post-restoration storm from 3/4/2020 showing precipitation (mm) and discharge (m³/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m³), storm TSS flux (g), and lag times (minutes), are presented on figure.

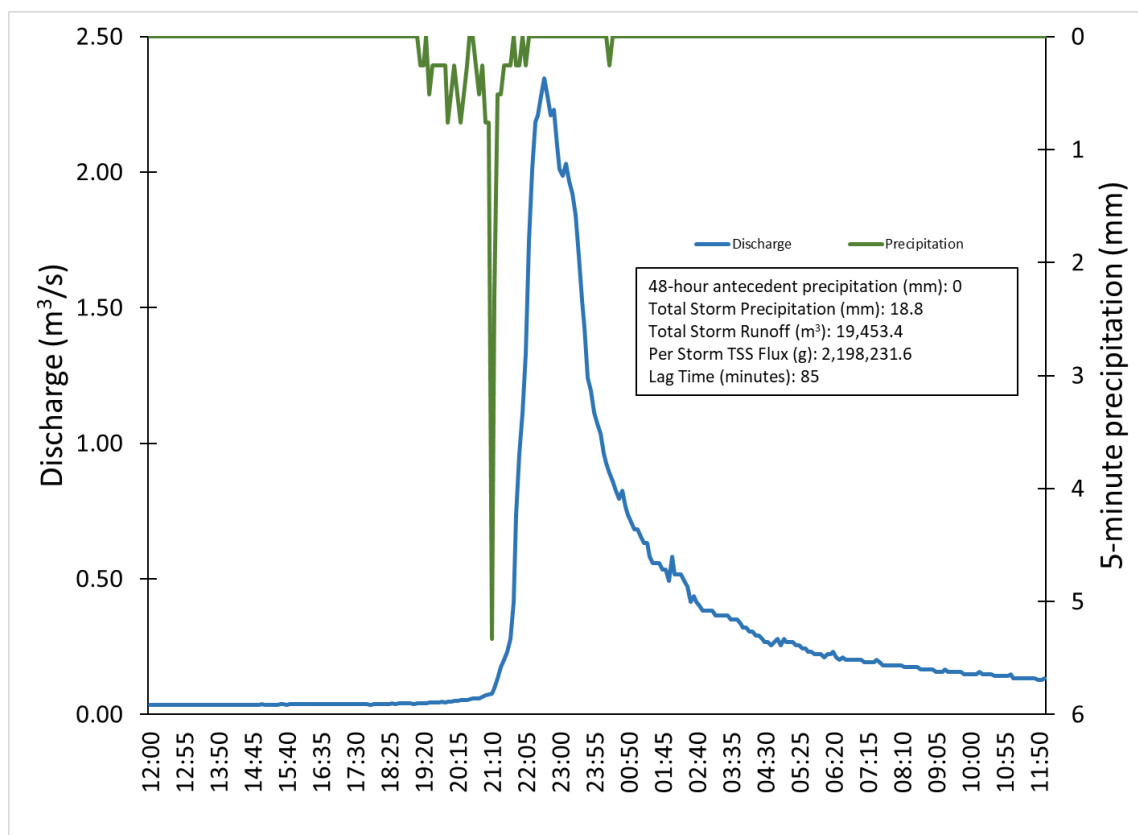


Figure 4.27 Post-restoration storm from 12/06/2020 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

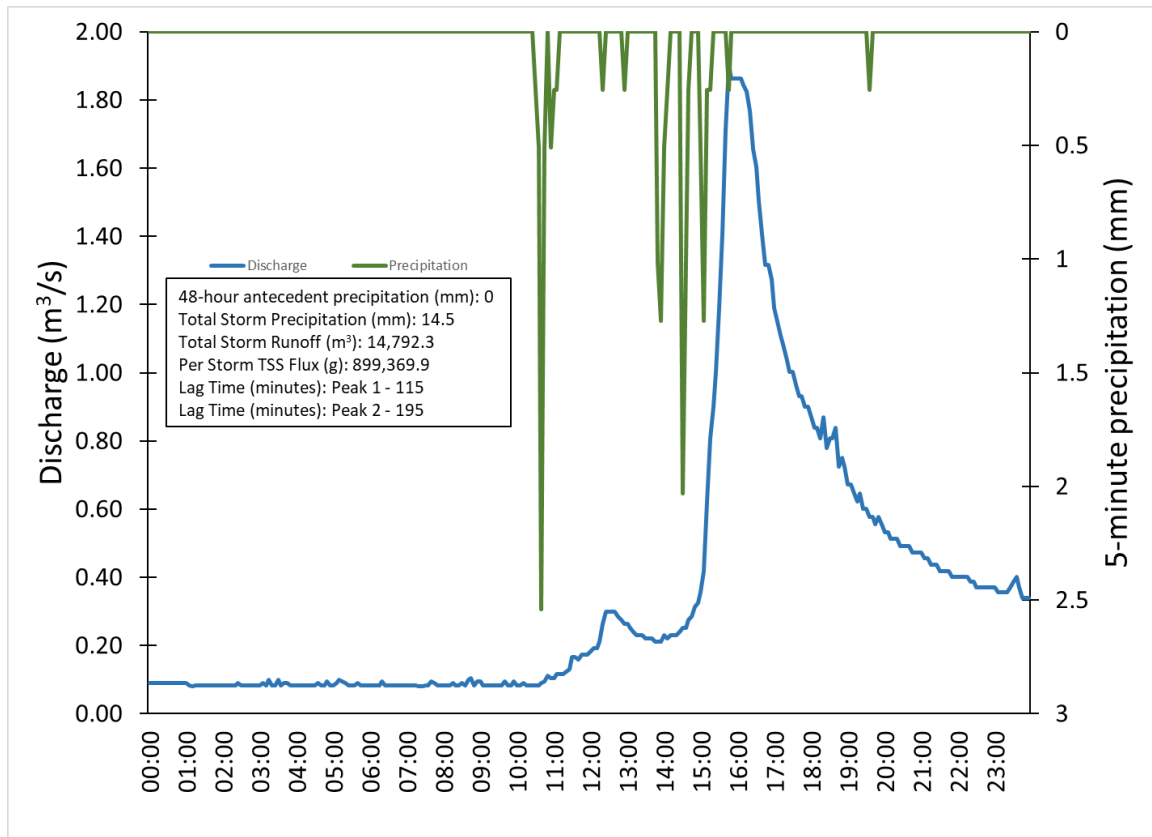


Figure 4.28 Post-restoration storm from 4/01/2021 showing precipitation (mm) and discharge (m^3/s). Storm characteristics, antecedent precipitation (mm), total precipitation (mm), total storm runoff (m^3), storm TSS flux (g), and lag times (minutes), are presented on figure.

5 SYNTHESIS

This dissertation research is one of the first studies to evaluate the efficacy of whole-watershed stream restoration. Here, I have assessed whether whole-watershed stream restoration has the potential to improve water quality and quantity within the Reedy Creek watershed by examining groundwater levels, baseflow N, and baseflow and stormflow sediment dynamics. This research is scientifically important as it addresses the importance of watershed-scale restoration which is needed as many streams in the United States are impaired (U.S. EPA, 2017). Furthermore, stream restoration is a growing field used around the globe to offset human alterations of the biosphere in a variety of land-uses (Bernhardt et al., 2005; Jenkinson et al., 2006; Brooks & Lake, 2007; Szalkiewicz et al., 2018; dos Reis Oliveira et al., 2020). Addressing stream alteration and incision at the watershed level, unlike at small reach scales, is needed to confirm long-term success of such projects, however, little research exists on large-scale restoration projects. Using a four-year pre-post analysis (2016-2021), this research has the goal of investigating the efficacy of using whole-watershed stream restoration approaches for ensuring that ecosystem functions are reestablished and maintained.

Chapter 2) Returning streams to pre-disturbed states, such as has been done in the Reedy Creek watershed, has the potential to impact the quantity of water within the watershed. Restoring all the streams within the watershed and creating shallower channels (i.e.: raising bed 2-3 meters) provides the conditions needed to increase groundwater levels and alter hydraulic gradients within the watershed itself which leads to increased water quantity and increased stream baseflow. Similarly, findings from a priority 1 restoration in Maggie Creek, Nevada indicate that restoration has the potential

to increase groundwater levels (Rosgen, 2010). Ultimately, increases in groundwater quantity within headwater watersheds can impact flow regimes downstream and sustain flows throughout dry periods (Hornberger et al., 1998; Alexander et al., 2007; Holden, 2014; Hofmann et al., 2018).

Chapter 3) Restored streams within the Reedy Creek watershed exhibit the characteristics needed to transform nutrients with increased regions of hyporheic exchange, changes in stream morphology from channelized to pre-disturbed reference conditions, and changes in canopy cover (Valett et al., 1996; Freeman et al., 2007; Tonina & Buffington, 2009; Stonedahl et al., 2012; Hester et al., 2017; Reisinger et al., 2019). Thus, restoring all the stream length (12km) within the watershed, altering stream morphology, increasing surface area to volume ratios, and increasing access to the surrounding floodplain has provided the conditions needed to increase nutrient processing and retention (Alexander et al., 2000; Gomez-Velez et al., 2017; Bernal et al., 2019; Li et al., 2020).

Chapter 4) The shallower channels within the Reedy Creek watershed have the potential to increase stream connection with surrounding floodplains which would allow for the deposition and accumulation of sediments and nutrients needed to support robust riparian and aquatic habitat (Jansson et al., 2007; Wohl, 2017). Altering stream morphology, adding instream features, and reconnecting surface water with the surrounding floodplain has the potential to decrease erosion of stream banks and beds allowing for decreased TSS during baseflow and stormflow by slowing baseflows and reducing flashy flows typical of degraded/urban systems (Walsh et al., 2005; O'Driscoll et al., 2010).

Overall, restoration can be considered successful if the above-mentioned statements are met, and thus, findings from this research within the Reedy Creek watershed indicate that restoration at the whole-watershed stream scale can enhance both ecological and hydrological watershed functions (Lewis et al., 1996; Lake et al., 2007; Hamilton, 2012; Stoll et al., 2016). As processes such as groundwater storage, nutrient processing, and sediment dynamics occur at the watershed-scale it is imperative that these issues are addressed at the same scale instead of small-scale reach restoration. For instance, increasing groundwater storage in the Reedy Creek watershed through raising streambeds has provided a steady baseflow which helps maintain a constant and balanced movement and processing of nutrients and sediments through the systems and has allowed streams to regain access to the surrounding floodplains increasing the potential of floodplain access where nutrients and sediments can be deposited.

Here, evidence suggests that restoration changes in stream channel elevation, morphology, and bed features led to changes in groundwater levels. These changes in groundwater levels coupled with the addition of bed features and alterations of watershed characteristics (i.e: canopy cover and AFDm) created the conditions needed for nutrient processing, retention, and transformation. Furthermore, the changes in channel morphology, the addition of bed features, and a shift in bed materials altered sediment dynamics by creating slower flows and reducing shear stress on the stream bank and beds. Findings from this research indicate that alteration of streams at the watershed level is able to provide positive benefits to the ecological and hydrological components essential to watershed function.

Finally, this research offers societal benefits as well. **Chapter 2)** With increasing populations and the potential for future drought, it is imperative to restore degraded watersheds in a way that can aid hydrologic retention. Ensuring the availability of water is important with climate change causing varied precipitation and drought conditions. Increasing groundwater storage is multifaceted in its benefits for both human use and natural processes. Benefits range from supporting streamflow, providing water for human consumption, and acting as a natural reserve for when drought intensifies, to providing a pathway for soil nutrients to enter streams. **Chapter 3)** Urban, suburban, and agricultural areas are known to alter stream nutrient dynamics. Understanding how stream restoration can impact nutrient dynamics can provide a way to neutralize nutrient issues from these land-use types. Being able to regulate nutrient dynamics is essential and can be the deciding factor on whether or not water is of high quality for human use or ecological function. **Chapter 4)** Changes in land-use have been shown to alter stream suspended solid loads through increasing terrestrial and stream erosion. Understanding how restoration can affect these loads is important as TSS impacts all aspects of the stream ecosystem including nutrient processing, food chains, vegetation/light interactions, and water quality. **Overall**, stream restoration at the watershed level is important to ensuring high quality water for humans and animals/plants alike. To do so, watershed managers and practitioners should identify and prioritize impaired waterways in headwater systems for restoration, create restoration goals and approaches that align with those goals, implement various restoration approaches known to benefit those goals, and monitor water quality, using testable hypotheses, for an extended period of time after restoration is completed to determine if restoration was successful. Restoration of stream ecosystems

is imperative if we are to protect water quality and quantity for current and future generations.

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