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ASSESSING DOWNSTREAM STORMWATER IMPACTS FOR URBAN WATERSHED PLANNING

by

JOHANNA MEYER PAVLOWSKY

A THESIS

Presented to the Faculty of the Graduate School of the

MISSOURI UNIVERSITY OF SCIENCE AND TECHNOLOGY

In Partial Fulfillment of the Requirements for the Degree

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Approved by

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ABSTRACT

The urbanization of watersheds has caused debilitating effects to downstream aquatic ecosystems in catchments and streams. The implementation of green infrastructure (GI), such as permeable pavements and bioretention facilities, has been shown to alleviate these effects by both reducing runoff and mitigating pollutants; however, the implements are often not designed with a specific goal of water improvement. This study targets understanding a small, impaired urban watershed, and the benefits green infrastructure may have to provide environmental, social, and economic improvement to the watershed.

Portions of Rolla including much of the S&T campus drain into the impaired urban waterbody Frisco Lake, plagued with poor water quality, eutrophication, and a substantial fish kill that took place in 2014. Lake phosphorus (P) concentration serves as a good indicator of freshwater quality due to its pertinence to algal growth. Beginning in the fall of 2014, monitoring methods, involving sampling and laboratory analysis, were used to support the modeling of stormwater runoff flows and P loads at outfalls draining into the contaminated lake. Monitoring results showed TP yields of 17 and 31 kg/ha/yr and mean-annual concentrations of 0.43 and 0.42 mg/L at the stormwater outfalls to the lake and were used in mass balance modeling to determine a required 40% P loading reduction to improve lake quality. Recommendations for upstream stormwater management, including a proposed GI plan were developed. Stormwater improvements were projected and used in a post-GI implemented Frisco Lake mass balance model, resulting in healthy lake P levels. The project methodology and watershed improvement plan are to be utilized by city managers for watershed management planning.

ACKNOWLEDGMENTS

Though countless hours of my life were spent completing this project, it would have never been possible without the help of many people. I would like to thank Dr. Burken for advising me through every step of this project by providing me with infinite guidance and wisdom as I progressed as a graduate student and professional. He is a great mentor, and an even greater person. I must also thank Dr. Holmes for co-advising this project. He was critical in the design and implementation of this study and provided much needed hydrological insight, always with utmost enthusiasm to teach and learn. Additionally, I want to thank my former professors and committee members, Drs. Wronkiewicz and Mendoza, for doing their part in providing me with the education that allowed me to become the engineer I am today.

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My research group was great and deserves recognition. Jordan Wilson was always there to answer all my monitoring related questions with alacrity. As was Matt Limmer. Madison Gibler was also crucial in the project site equipment set up and teaching me the ways of the lab. Katie Bartels. Heart and will miss all you guys.

Most importantly, I want to thank my family (with a special shout out to Aaron and my grandparents) and friends as they are responsible for me being where I am today. Mom, Dad, and Carrie were all always there to provide guidance and encouragement through the times of thesis-completing frustration of which I am infinitely grateful. Eric. Thanks for being my number one research assistant and one-and-only. I love you all.

I consider myself the luckiest person in the world, mainly due to the wonderful people in my life...who are all saints for putting up with mah shit.

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1. INTRODUCTION

1.1. URBANIZED WATERSHEDS

As global populations rise and continue to migrate from rural to urban areas at increasing rates, land is converted from its natural state to urban environments, causing significant degradation to downstream water bodies (Carey *et al.,* 2013; Paul & Meyer 2001; Brabec *et al.,* 2002). The construction and development of urban cities has required the conversion of vegetated, pervious, "green" terrain to man-made, impervious, "gray" landscapes that disrupt the area's natural hydrology (UACDC, 2010). Runoff volumes are attenuated during wet-weather events over undeveloped landscapes as much of the precipitation can collect in natural surface depressions, be intercepted by the vegetation, or infiltrate into the soil column of pervious areas, resulting in significant precipitation losses to evapotranspiration and infiltration (Paul & Meyer, 2001). However, in urban environments comprised of increased impervious area in the form of roadways, sidewalks, and building roofs, precipitation is incapable of penetrating the impervious surfaces and, thus, reduces the precipitation losses (Sun & Lockaby, 2012).

The water cycle is short-circuited in urban areas, resulting in a larger percentage of precipitation being converted to runoff, which has been hydraulically designed to be channel water swiftly downstream through extensive stormwater collection systems (Bedient *et al.,* 2013). Additionally, these gray landscapes generate urban pollutants such as heavy metals, *E. coli,* oil and grease, nutrients, pharmaceuticals, and suspended sediments that are picked up by stormwater flows and carried downstream, without mitigation that would have naturally been provided in undisturbed landscapes by vegetation and infiltration (Steinman *et al.,* 200;, UACDC, 2010). As a result, the flow regimes of stormflows from urban watersheds are characterized by increased total runoff volumes, greater peak flows, flashier hydrographs with rapidly rising and receding flows, and poorer water quality (Walsh *et al.,* 2005; May *et al.,* 2006). A comparison showing the typical proportions of precipitation conversion to runoff, shallow and deep infiltration, and evapotranspiration in urban and natural environments is presented in Figure 1.1.

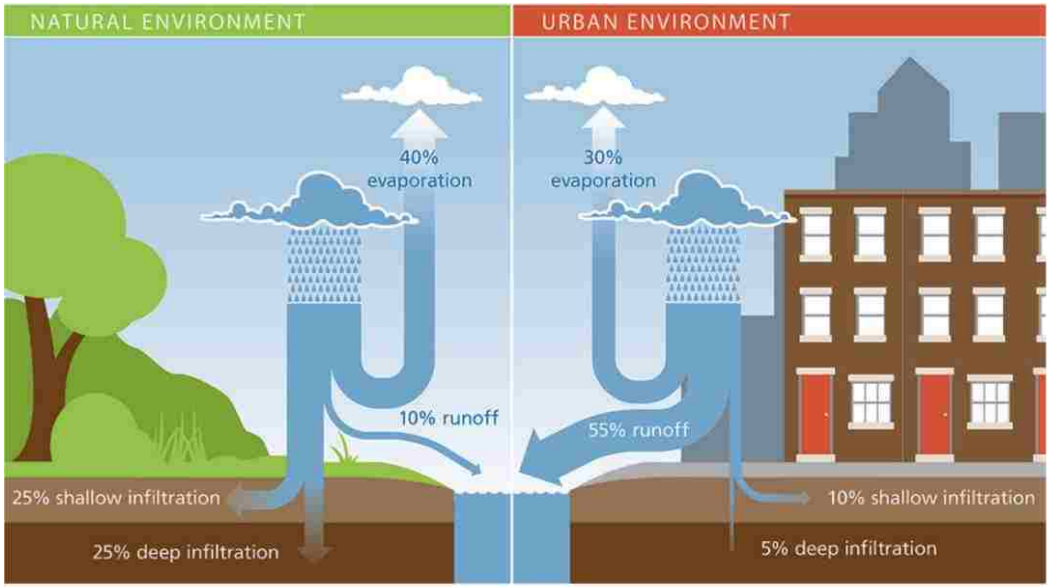

Figure 1.1. Hydrologic comparison of urban vs. natural watersheds (phila.gov/water/wu/stormwater)

1.2. DOWNSTREAM EFFECTS

1.2.1. Urban Stream Syndrome. Urbanized watersheds have produced consistent, often drastic negative effects on downstream water bodies including increased urban pollutant, nutrient, and suspended sediment loads; flashier hydrographs; reduced biotic integrity; and altered stream geomorphologies (Paul & Meyer 2001, Walsh *et al.*, 2005, UACDC, 2010). Increased runoff volume and flooding alters natural stream morphologies by uprooting trees and vegetation, promoting channel bed and bank erosion, disrupting floodplain connectivity, and ultimately degrading natural aquatic habitats (Walsh *et al.*, 2005, Bedient *et al.*, 2013). With the drastic changes in channelization, the hyporheic flow, which is the exchange of water between the groundwater and riparian waters, is reduced (UACDC, 2010). This reduction disrupts many key ecological functions necessary for fish spawning and nutrient cycling, as well as the natural filtering that reduces stream pollutant levels (May *et al.*, 2014; Dauer *et al.*, 2000). These changes in flow regimes also affect urban and suburban human developments in immediate and downstream environments. Each year in the United States, FEMA expects $2 billion to be spent on property damage as a result of flooding (NRC, 2008). Ultimately, these harmful effects of urban stormflows affect overall watershed health disturbing many watershed functions, such as energy balances, baseflow and groundwater recharge, evapotranspiration, nutrient cycling, peakflow flooding, and aquatic biodiversity of ecosystems, which are summarized in Figure 1.2.

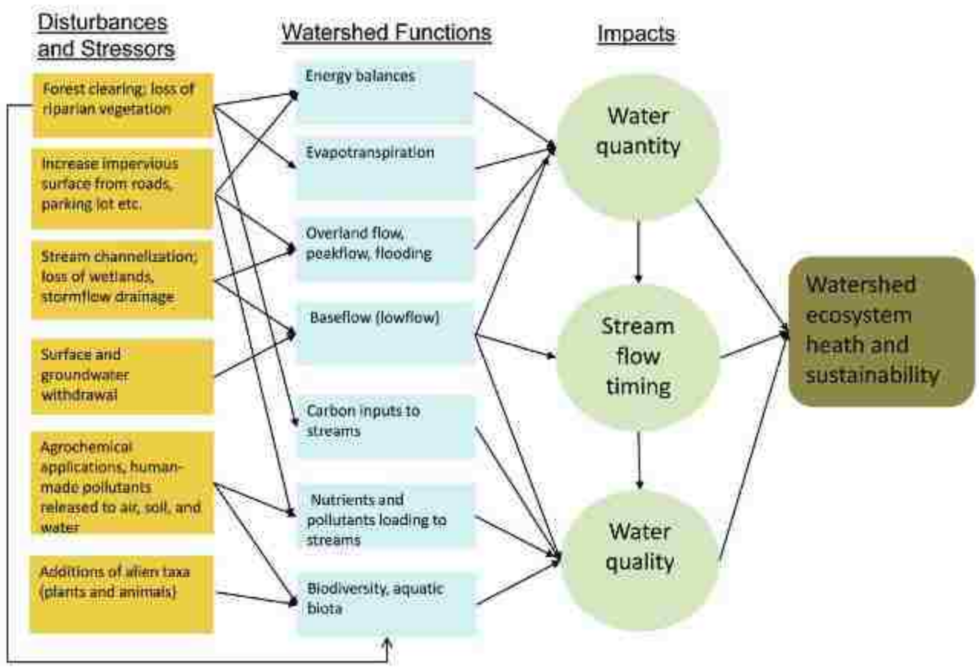

Figure 1.2. Urbanization stressors and relation to downstream watershed health (Sun & Lockaby, 2012).

1.2.2. Eutrophication. Urban stormflows generally have warmer temperatures and higher nutrient pollutant levels that often destroy downstream aquatic ecosystems through a naturally occurring process called eutrophication (Carpenter *et al.*, 1998; Hall *et al.*, 1999). As a result of increasing global urbanization and land-use alterations, this process has become accelerated beyond its natural rates, termed cultural eutrophication (Steinman *et al.*, 2009; Anderson *et al.*, 2002; Carey *et al.*, 2013). When excess amounts of nitrogen and phosphorus are carried downstream, simple photosynthetic organisms, most prominently algae and plankton, quickly uptake the nutrients and experience excess growth, called blooms (Elser, 2012; Leitz, 1999). Algal biomass adds excess biological organic matter to the water body. When the algae die, the organic material sinks to the bottom of the catchment and decomposes, exhausting large amounts of oxygen (Carpenter *et al.*, 1998). Without any oxygen left in the waters, hypoxic conditions form (DO < 2 mg/L), and other aquatic lifeforms asphyxiate (Carey *et al.,* 2013; Anderson *et al.,* 2002). Eutrophication creates hypoxic dead zones, which are areas of unproductive biological activity, in many freshwater ponds and lakes as well as many coastal estuarine environments, particularly those that drain significantly large agricultural watersheds or heavily urbanized areas (Steinman *et al.,* 2009; Rabotyagov, 2014; Dauer *et al.*, 2000; Leitz, 1999).

Eutrophic algal blooms can also prevent sunlight from reaching benthic communities, depriving underwater bottom-dwelling aquatic plants and animals, such as filter feeders, of life (Carey *et al.,* 2013). In marine environments, algal blooms create harmful toxic conditions known as brown and red tides which release toxins that can be lethal to humans and aquatic animals such as manatees and finfish (Steinman *et al.,* 2009; Carpenter *et al.*, 1998, Anderson *et al.*, 2002). In freshwater systems, algal blooms are characterized by blue-green cyanobacteria that emit foul odors and toxins that pose threats to humans, livestock, and aquatic animals (Carpenter *et al.*, 1998). Several areas of the US, such as the Chesapeake Bay, Gulf of Mexico coast, and Lake Erie are devastated by recurring algal blooms each year (Davis & Masten, 2009; Dauer *et al.*, 2000; Rabotyagov, 2014; Anderson *et al.*, 2002). The degree of severity and occurrence of aquatic hypoxia associated with algal blooms is dependent upon seasonal, geographical, chemical, and biological factors as well as the water's recent climate and flows (Sondergaard *et al.,* 2001; Steinman *et al.*, 2007).

On global, regional, and local scales, eutrophic biomass blooms and subsequent aquatic hypoxia incur economic costs from the destruction of recreational waters, impairment of drinking water, and reduction of productivity of commercial fresh and marine shellfish industries and fisheries, limiting geotourism, and increasing human health costs (Carey *et al.*, 2013; Steinman *et al.,* 2009; Mallin *et al.,* 2000). Indeed, considering the US alone, the economic toll of decreased waterfront real estate values, impaired recreational water activities, endangered species and habitat recovery for inland eutrophic water bodies was estimated at $2.7 billion (Elser, 2012). Though the magnitude of the global cost of eutrophic waters is unclear, a study estimated a possible gain of $10 billion if eutrophic conditions in the Baltic Sea, alone, were reduced (Rabotyagov, 2014). The harmful effects of eutrophication are realized not only in freshwater lakes in large scale watersheds, wetlands, and coastlines, but also in smaller, local urban lakes such as Frisco Lake in Rolla, MO. Stormwater runoff from smaller urban drainage areas can cause eutrophication, creating the unaesthetically pleasing, toxic algal blooms as depicted in Figure 1.3. Such urban eutrophication occurrences impact the cultural and ecological services provided by urban water bodies.

Figure 1.3. Floating dead fish in Frisco Lake in Rolla, MO July 2014 as a result of aquatic hypoxia.

1.3. URBAN STORMWATER MANAGEMENT

Traditional urban stormwater management involves the use of widespread collection and transportation systems, comprised of hydraulically designed channels based on natural topography and underground sewer systems, to remove the runoff from the urban area as quickly as possible (EPA 1993; Roy *et al.*, 2007). The stormwater runoff is released into a downstream river, a designed runoff catchment, or a wastewater treatment plant for cases in which the storm and municipal sewers are not separated (NRC, 2008). Many of these sewer systems were initially designed to carry both sewage and stormwater, creating potential overflow situations, termed combined sewer overflows, during storm events in which sewage water will bypass treatment plants and discharge directly into surface waters (Roy *et al.,* 2007).

Due to the water quality problems associated with traditional urban stormwater management, the US Environmental Protection Agency (EPA) expanded legislation to its original water pollution control plan. In the 1970s, the EPA passed the Clean Water Act, which used the National Pollution Discharge Elimination System (NPDES) permit program to regulate the stormwater discharges of cities and other point sources into surface waters (EPA, 1993; Tsihrintzis & Hamid, 1997). Since then, the EPA has expanded regulation on stormwater handling by implementing phased approaches to gradually separate combined sewers into sanitary sewer systems and municipal separate storm sewer systems (Davis & Masten, 2009; EPA, 1993). In many places, separating in-place combined sewer systems is not economically feasible, so cities are responsible for designing stormwater management plans that incorporate Best Management Practices (BMPs) to reduce runoff volumes and CSO events (Steinman *et al.*, 2009;, UACDC, 2010; Tsihrintzis & Hamid, 1997).

1.4. URBAN STORMWATER MITIGATION

1.4.1. Best Management Practices (BMPs). In order to begin solving the complex issue of urban stormwater pollution, cities are building stormwater management plans that are unique to each location. In many cases, systemic changes from current city infrastructures to designs including more BMPs are being implemented. BMPs include conserving natural areas and vegetation, reducing hard and/or impervious surface cover, and retrofitting urban areas with Low Impact Development (LID) features that effectively hold and treat stormwater instead of conveying it downstream with no treatment (NRC, 2008). An increasingly popular BMP technique is Green Infrastructure (GI), which mimics the natural water cycle by harvesting, infiltrating, and evapotranspiring stormwater and promotes climate regulation and ecological functioning, such as sediment retention and nutrient cycling, that is lost in urban "gray" landscapes (UACDC, 2010). As such, city planners have worked to incorporate GI implements, such as green roofs, bioretention facilities, and porous pavements, into city design schemes due to their effectiveness in reducing and mitigating stormwater and providing economy and aesthetic enhancements to communities and their citizens' life quality (Tzoulas *et al.*, 2007).

1.4.2. Stormwater Monitoring and Planning. An understanding of urban watershed hydrology and water quality, including the quantity and movement of stormwater runoff, sources and locations of contaminants, and the degree of downstream water body impairment is required to effectively design and plan GI implements (Carey *et al.*, 2013; Lathrop *et al.*, 1998; UADAC, 2010). In the US, many governmental agencies such as the EPA, city public works, and USGS, work to provide these understandings by conducting field hydrologic and water quality monitoring studies that model and quantify downstream pollutant levels and stormflow volumes to estimate overall contaminant loads across land-areas and regions (Leitz, 1999). After an assessment of human and natural factors, such as city climate and location and urbanization rates, land-use, and the observed water quality conditions, stormwater pollution reduction goals and more effective stormwater management strategies can be created (EPA, 1993; NRC, 2008). In order for implemented GI to provide solutions to urban runoff in cities, particular care must be taken on a community-specific basis in designing management strategies. For example, in densely populated urban environments, incorporating retention strategies involving the infiltration of large volumes of runoff may not be economic compared to other alternatives, such as water storage (Bedient *et al.*, 2013). Once an effective management plan is derived, an interdisciplinary collaboration between city planners, community leaders, engineers, scientists, and builders must be completed to implement GI measures as many developed cities include pre-existing gray infrastructure.

2. GOALS AND OBJECTIVES

2.1. GOALS

The primary goal of this study was to evaluate the beneficial environmental impacts associated with upstream green infrastructure implementation by understanding the urban hydrology of an impaired watershed through monitoring downstream impacts of stormwater runoff. To achieve this goal, specific research objectives were set as follows:

2.2. OBJECTIVES

- Objective 1: Develop, implement, and conduct a year-long, fine-scale urban hydrologic stormwater quality monitoring plan using site specific methodologies
    - Hypothesis: Stormwater discharges and related nutrient loading trends will show elevated P levels that can be used as baseline data for future studies.
- Objective 2: Model annual discharges and P loads into Frisco Lake using collected hydrologic and water quality data in order to estimate the P reduction required for water quality improvement
    - Hypothesis: P loads into Frisco Lake will need significant reduction in order to reduce eutrophic activity in the future.
- Objective 3: Assess watershed land cover characteristics and observed stormwater quality to plan GI implements and stormwater management strategies for lake water quality improvement
    - Hypothesis: Green infrastructure implements, such as bioretention facilities, will reduce the runoff volumes and improve the quality of urban stormwater to levels capable of preventing Frisco Lake’s eutrophication.

Successfully accomplishing these objectives will provide valuable information and data on urban water quality and the role of green infrastructure in improving watershed functioning. If the resulting findings support these core hypotheses, the overarching goal should be met. However, if these hypotheses are determined to be incorrect, the resulting research is still useful for scientific understanding and purposes.

3. LITERATURE REVIEW

3.1. URBAN STORMWATER

3.1.1. Stormflows. Stormwater is any precipitation that is unable to infiltrate into the ground and therefore runs over the surface, collecting and carrying debris and pollutants (Carey *et al.*, 2013). Perennial rivers and streams receive waters from the shallow subsurface and groundwater and therefore consistently discharge throughout the year, termed baseflow (Moix & Galloway, 2004). During storm events, stormwater runoff concentrates into flows causing stream discharges greater than the baseflow, termed stormflow. Stormflow waters are often more polluted with suspended solids, phosphates, surfactants, BOD, and fecal coliform bacteria than those of baseflow (Mallin *et al.*, 2009). Additionally, numerous ephemeral streams receive water solely from surface runoff sources and, therefore, appear during storm events and subside once the runoff is channeled downstream (Leopold & Miller, 1956). Due to the drastic land-use changes associated with urbanized environments, the resulting stormflow runoff hydrographs of both perennial and ephemeral streams generally have higher peak discharges and flashier curves than those of less developed watersheds (Paul and Meyer 2001; Bedient *et al.*, 2013). Post-urbanization hydrographs are characterized by steeper rising and receding limbs with shorter lengths of time to peakflow and return to baseflow conditions (EPA, 1993; Walsh *et al.*, 2005) as exemplified in Figure 3.1.

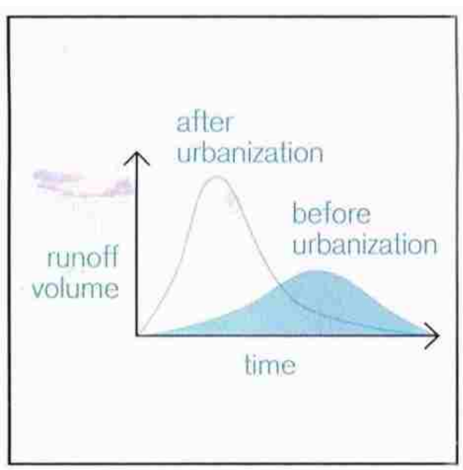

Figure 3.1. Runoff hydrograph from post and pre-developed watershed (UACDC, 2010)

3.1.2. Quality. Stormwater carries a variety of dissolved constituents and particulate matter from both anthropogenic and natural sources. For monitoring purposes, pollutants are categorized into physical, chemical, and biological subgroups (Caltrans, 2013). Physical characteristics are based upon physical properties of the stormwater itself, such as temperature, turbidity, conductivity, total suspended solids, pH, and biological oxygen demand. Healthy Ozark streams have temperatures below 30°C, pHs within the range of 6 to 9, turbidities less than 5 NTU, conductivities ranging from 150 – 500 μS/cm, and saturated DO levels of 80% with DO levels less than 5 mg/L (Hutchison, 2010). Chemical characteristics are based upon chemical constituents of the water that can be specifically measured (EPA, 1993). Typically, they can be split into dissolved and suspended fractions via filtering methods in lab analyses. Examples of common chemical stormwater pollutants include metals and nutrients (NRC, 2008). Biological characteristics of stormwater include the properties that relate to any living materials within the water and its toxicity to living organisms (Caltrans, 2013). Stormwater samples are typically tested for indicator bacteria such as total coliforms, fecal coliforms, and *E. coli* that correlate the degree of pathogenic activity and toxic effects from the stormwater (EPA, 1993).

3.1.3. Pollution. Though most of the Earth's lands are rural, the drastic land use changes associated with urbanization, such as increases in vegetation clearing and industry, disproportionately affect downstream water quality in rivers, lakes, estuaries, and streams (Brabec *et al.*, 2002; Sun & Lockaby, 2012). In the U.S. and all across the globe, urban stormwater is the foremost source of contamination to fresh and estuarine water bodies (Mallin, 2009; EPA, 1993). In the United States alone, there are a documented 38,114 miles of impaired streams and rivers, 948,420 acres of impaired lakes, 2,742 square miles of impaired bays and estuaries, and 79,582 acres of impaired wetlands of which urban stormwater is labelled as responsible (NRC, 2008). Urban stormwater flows are typically seasonal, dependent upon climates and precipitation patterns, and can be variable in quantity and quality depending on the degree of land-use alterations (Walsh *et al.*, 2005). For both large and small urban watersheds, stormwater flows vastly disrupt baseflow conditions and potentially degrade the natural hydrology, water quality, soils, and aquatic ecosystems of downstream wetlands (Walsh *et al.*, 2005;

EPA, 1993). In some heavily urbanized areas, the concentration of metals and pollution index of stormwater runoff from the initial portion of the storm event can be greater than that of raw sewage. (Sansalone & Cristina, 2004; UACDC, 2010).

3.1.4. Pollutants. Urban pollutants are generated by an assortment of human activities and strongly associated with increases in impervious land cover, which occur as a result of urbanization (EPA, 1993; Brabec *et al.*, 2002; Sun & Lockaby, 2012). Common urban pollutants and their primary sources include sediment, from erosion and construction sites; oil, grease, and toxic chemicals from industrial uses and motor vehicles; pesticides and nutrients from lawns and gardens; detergents from laundromats and car washing; viruses, bacteria and nutrients from pet waste and sewage; road salts from highways and transportation operations; heavy metals from roof shingles, motor vehicles, and industry; and also thermal pollution from dark impervious surfaces such as streets and rooftops (Carpenter *et al.*, 1998; EPA, 1993). Pollutants come from one of two types of sources: point and non-point. Point sources are those that discharge waste, such as sewage or stormwater, from a single point and are regulated by the EPA in the NPDES program (EPA, 1993). Non-point sources have non-discrete or multiple points discharging pollutants as a result of sheetflow flowing over urban surfaces during storm events. Non-point sources generally require more robust, BMP solutions to mitigate as they are more complex to contain (Hoos *et al.*, 2000; Davis & Masten, 2009; Carpenter *et al.*, 1998).

3.2. NUTRIENT POLLUTANTS

In natural quantities, nitrogen and phosphorus provide necessary growth to all living plants and animals (Elser, 2012; Davis & Masten, 2009). However, within the last fifty years, the industrial and anthropogenic use of nitrogen and phosphorus has increased to two to three times the natural levels (Rabotyagov, 2014), allowing these vital nutrients to become stormwater pollutants that heavily influence the level of eutrophication in downstream waterbodies when large quantities are flushed and collected downstream (Carpenter *et al.*, 1998; Hall *et al.*, 1999; Dauer *et al.*, 2000; Leitz, 1999). Typically, phosphorus is the limiting nutrient that leads to eutrophication in freshwater lakes, rivers, streams, and inland waters, while nitrogen is generally the limiting nutrient in brackish waters along the coasts and marine systems. A water body is deemed phosphorus limited if the N:P ratio is greater than 15:1 (Vitousek *et al.,* 1997; Davis & Masten 2009). Phosphorus often adheres to sediment within the stormwater runoff and settles in catchments where it remains in the environment for indefinite periods of time. Therefore once the nutrients are deposited, algal blooms are likely to reoccur each year (Carpenter *et al.*, 1998; Sondergaard *et al.*, 2001).

3.2.1. Nitrogen. N exists primarily in the environment as non-reative N_2 gas in the atmosphere, where it can be fixed naturally by leguminous crops or during lightning strikes and converted into biologically available N in the forms of organic N, ammonium, nitrate, or nitrate (Carpenter *et al.,* 1998; Khwanboonbumpen 2006). In the 1950s, with the discovery of the Haber-Bosch process, nitrogen was capable of being artificially fixed by humans (Elser, 2012). Additionally, N can be fixed inadvertently during combustion by vehicles and industrial processes where it will form NO_x gases (Vitousek *et al.,* 1997). Since then the amount of industrially created nitrogen, in the form of ammonium nitrate, ammonia, or urea, in the environment has exponentially grown. The current rates are at 450 million tons a year due to its global use as an agricultural fertilizer and efficacy in increasing crop productions (Elser, 2012; Vitousek *et al.,* 1997). Once atmospheric N is fixed and introduced into the terrestrial environment, it is converted back and forth into ammonia, nitrite, and nitrate by biological interactions with plants, animals, and microbes (Leitz, 1999). Organic N is not available to plants until it has been converted to a soluble form such as nitrate or ammonia (Barth, 1995).

With excessive quantities of industrialized fixed N introduced into the environment, much of it is mobilized and flushed away by overland flows during storm events into rivers and deposited into downstream water bodies where harmful effects take place (Carpenter *et al.,* 1997; Groffman *et al.*, 2004; Leitz 1999). For one, excess concentrations of nitrates and ammonia cause severe aquatic ecosystem degradation via eutrophication, particularly in estuarine environments (Vitousek *et al.,* 1997). Also, nitrates are highly dissolvable in groundwater where they can pose harmful threats to livestock and humans causing methemoglobinema, a disease commonly known as blue baby syndrome, if used as a drinking water sources (Carpenter *et al.,* 1998). In higher concentrations, ammonia has also been known to be toxic to fish (Khwanboonbumpen

2006). Lastly, quantities of ammonia in wastewater treatment processes can react with disinfectants during disinfection to form harmful chloramines (Davis & Masten 2009).

3.2.2. Phosphorus. P, like carbon and nitrogen, is one of the essential constituents in biological tissues and, therefore, sustains growth rates and life (Elser, 2012; Khwanboonbumpen, 2006). Naturally derived in water from slow rock weathering processes, P is not often readily available for plants as it is attracted to organic portions of soil, so it remains a limiting factor in the growth of primary producers in many terrestrial ecosystems (Leitz, 1999; Elser, 2012). However, similar to industrial N fixation, since the industrial revolution, the rate at which fossil phosphorus has been mined for human use, as a fertilizer to increase crop production, has increased to more than 400% (Elser 2012). With unnaturally high levels of P newly available in the environment, P is collected in surface runoff, collects in freshwater bodies, and causes devastating effects via eutrophication (Hall *et al.,* 1999; Carey *et al.,* 2012).

P exists in surface waters in either a particulate form, where it may be directly deposited into channel or lake beds, or in a dissolved inorganic form, generally an orthophosphate or polyphosphate that is easily taken up by aquatic plants and primary producers (Barth, 1995; Khwanboonbumpen, 2006). Soluble inorganic orthophosphate is both stable and readily available to plants. Therefore, it is the most hazardous form to introduce to aquatic environments (Sondergaard *et al.,* 2001). Additionally, in nature P has a tendency to adsorb to fine sediments, such as silts, clays or organic soils or react with minerals within soils, such as calcium carbonate and iron, and thus becomes immobilized by the sediment and not readily available to plants (Leitz, 1999; Sondergaard *et al.,* 2001).

3.2.3. Urban Nutrient Sources. Major sources of N and P in stormwater runoff include residential fertilizers, atmospheric deposition, vehicle emissions, point source discharges of municipal and industrial waste, laundry or cleaning detergents, pet waste, yard litter, and suspended sediment and particulate matter (Carpenter *et al.,* 1998, Davis and Masten 2009, EPA 1993).

3.2.3.1. Fertilizers. Urban residential areas that have been stripped of natural soils during development are landscaped with non-native plants, turfgrass lawns, and gardens of that frequently require fertilizer application and irrigation in order to maintain growth (Carey *et al.,* 2012). These fertilizers are comprised of N, P and potassium (which has little environmental concern) that are often flushed away in urban stormwater runoff (Davis & Masten, 2009). Many factors dictate the proportion of N and P taken up by plants versus the amount that leaches into runoff including: length of time between application and irrigation, timing of fertilizer application relative to plant growth period, placement of fertilizer, type of grass or plant, type of fertilizer (dissolved vs. slow reactive), type of soil, and the degree of turf establishment (*i.e*., root and grass density) (Carey *et al.*, 2012). Research indicates that less than 5% of applied N will be leached from lawns if optimal fertilizer application patterns are used with moderate fertilizer rates, though losses is significantly higher using poor practices (Carey *et al.*, 2012).

In a study by Barth (1995), a compilation of residential fertilizer use survey results across several states concluded that an average of 70% of residents fertilize with roughly one third hiring a commercial lawn service. In a study of lawn care fertilizer use in five North Carolina urban communities, 54 to 80% of residents applied fertilizer an average 1.5 times a year, yielding an average annual fertilizer application of 227 tons for 80,000 residents. N fertilizer rates ranged from 24 to 151 kg N per hectare per year (Osmond & Hardy, 2004). Another study by Groffman *et al*. (2004) on suburban watersheds estimated home lawn fertilizer use at 14.4 kg N per hectare per year and found retention rates of 75%. Additionally, in a similar study by Carey *et al.,* (2012) a moderate rate of residential fertilizer was considered 200-300 kg per ha per yr. Estimates for commercial lawn service application rates in the U.S. include 194 to 258 lbs/acre/yr of N, and estimates for homeowners include anywhere from 44 to 261 lbs/acre/yr and 4 to 26 lbs/acre/yr of N and P, respectively (Barth, 1995). Ultimately, fertilizer rates are not easily estimated and are generally compiled using survey methods and are, therefore, variable.

3.2.3.2. Municipal and industrial waste. As N and P are present in human excrement, another contributor of nutrient pollution to water bodies includes inputs from municipal sewage and industrial waste discharges as well as any leaky sewage connections. Depending on the locality of the point source in relation to the catchment, the proportion of nutrient pollution from discharges varies (Khwanboonbumpen, 2006). In point sources under the NPDES permitting system, N and P effluent rates are measured routinely, and, therefore, specific rates at different locations can generally be calculated. In a study determining N contributions to a Tennessee watershed, municipal wastewater effluents without data were given an average value of 15 mg/L and average P concentrations at 3.5 mg/L (Hoos *et al.*, 2000). Industry wastewater effluent concentrations of N vary by facility type, level of treatment, and size of operation, and average values can be looked up in government agency tables provided by the National Oceanic and Atmospheric Administration or EPA (Hoos *et al.*, 2000, Davis & Masten, 2009).

3.2.3.3. Atmospheric deposition. Ammonia, nitrous oxide, nitric oxide, nitrogen dioxide, and nitrate exist in the atmosphere as fine particulates, liquid aerosols, or gases and are derived from various sources such as lightning strikes, fossil fuel combustion, vehicle emissions, plant volatilization and decomposition processes (Carey *et al.*, 2013; Khwanboonbumpen, 2006). These fixed forms of N in the atmosphere can be deposited during wet-weather by being dissolved in rain water or during dry weather as particulate solids (Carpenter *et al.*, 1998). In the US alone, mostly due in part to ubiquitous fossil fuel combustion, an estimated 3.2 million tons of N is deposited from the atmosphere with studies showing annual deposition rates in suburban watersheds of 11.2 kg N per hectare (NRC, 2008; Groffman *et al.*, 2004). In a study by Hoos *et al.*, in the Tennessee River basin, wet nitrate rates ranged from 0.33 to 0.68 kg/ha for ammonia and 0.53 to 0.73 kg/ha for nitrates (2000). Additionally, a study within the Washington, DC metro area estimated atmospheric deposition rates at 17 lbs/yr and 0.7 lbs/yr for N and P respectively (Barth, 1995). Atmospheric P inputs to watersheds are considerably less than those of N, although, in areas with excess dust or sediment deposition, estimated P rates could be higher (Hoos *et al.*, 2000; Carey *et al.*, 2013). Depending on certain watershed characteristics, such as geology, soils, vegetation, and slope, atmospheric deposition inputs from N will vary (Davis and Masten 2009).

3.2.3.4. Yard and pet waste. Though yard and pet waste are considered minor sources of N and P, they are part of the estimated 12% of nutrient pollution that results from non-point sources (Khwanboonbumpen, 2006). N and P are major constituents of biological matter and, therefore, are present as organic material in animal excrement and yard waste. Limited studies exist on the inputs of N and P from pet waste, but an N source study in Baltimore, MD, showed pet waste inputs of N at 17 kg/ha/yr, exceeding inputs from fertilizers and atmospheric deposition (Carey *et al.*, 2013). In areas with P restrictions in fertilizer, pet waste can be a predominate source of P into surface runoff (Carey *et al.*, 2013). In a study by the Northern Virginia Soil and Water Conservation District in Fairfax County, VA, dogs were estimated to excrete and introduce into the environment 180,000 pounds of waste daily (NRC, 2008). In different urban areas with varying densities of people and pets, N and P inputs from pets are likely to vary accordingly, but remain significant.

As much as 25 to 60% of N fertilizer that is applied to lawns and gardens is taken up stored by plants (Carey *et al.*, 2012). In a study by the Rodale Institute Research Center, an acre of yard clippings provides an average of 235 pounds of nitrogen, 210 pounds of potassium, and 77 pounds of phosphorus (Barth, 1995). Leaves, comprised of approximately 0.2% P and 1.0% N dry weight, flowers, weeds, and grass clippings can be flushed downstream during wet-weather events (Khwanboonbumpen, 2006). Yards that recycle grass clippings are prone to additional N and P leaching unless fertilizer usage rates decrease accordingly (Carey *et al.*, 2013).

3.2.3.5. Detergents. Before the 1980s, when regulatory measures banned sodium phosphate-containing cleaning agents, detergents were a major source of reactive phosphorus into downstream environments (Davis & Masten, 2009). Still today, many commercial cleaning agents, such as those used for washing cars or cleaning laundry, are comprised of orthophosphates and polyphosphates that can be directly introduced into surface waters (Sondergaard, 2001). Current rates relating population and expected laundry and household detergents discharged downstream have been estimated at 0.3 kg P /capita and 0.1 kg P/capita, respectively, in the U.S. (Khwanboonbumpen, 2006). Car washing, in particular, allows detergents to be directly introduced to stormwater catchments as residents hose down their cars on their driveways (EPA, 1993; NRC, 2008).

3.2.3.6. Sediment erosion. Of the two major pollutant nutrients, P has a tendency to adsorb to sediment particles and can therefore be transported into water bodies in storm runoff from areas experiencing soil erosion (Carpenter *et al.*, 1998). Soil erosion rates are orders of magnitude higher in construction sites of urban areas compared to agricultural or forested land areas because the natural landscape is being altered, thus disturbing the protective vegetation or surface holding the sediment in place (Carpenter *et al.,* 1998). Therefore, increased rates of eroded material increase P inputs downstream as the P becomes dissolved into the surface water (Elser, 2012). A study in North Carolina reported that sediment export from a phase I construction site was ten times greater than other land uses, and average N and P inputs from phase II construction sites have been found to be 36.3 kg/ha/yr and 1.3 kg/ha/yr, respectively, (Carey *et al.*, 2013). Soil impacts from construction areas can be expected years after development and eventually stabilized to expected release rates (Carey *et al.,* 2013).

3.3. STORMWATER QUALITY MONITORING

Stormwater runoff quality varies temporally and spatially. Therefore consideration of project goals is pertinent in designing a sampling plan. Holistic and project specific approaches can be taken depending on the extent of each monitoring site and project goal (Caltrans, 2013).

3.3.1. Planning. When building a stormwater monitoring plan, regardless of scale, the initial consideration is to determine the project objectives by defining what data or knowledge is required by the end-users. Knowing the project goals allows one to properly determine the relevant hydrological and water quality parameters to monitor (Hamilton, 2012). Once the objectives and data output goals have been developed, the plan's geographical boundaries and temporal time frame should be established to sufficiently accomplish the project goals (Caltrans, 2013). Traditionally, water quality data has been collected daily and then accumulated annually and published for practical use. However, as technology advances and more data streaming options become available, monitoring plans are incorporating real-time data collection methods (Hamilton 2012). Next, the monitoring data and information must be determined by deciding the types of data required (such as site, event, and sample data), identifying any project constraints and determining availability of sampling methods, and understanding the performance criteria requirements (Caltrans, 2013). Next, the data analysis approach including any statistical procedures, should be identified. Then, data quality objectives specific to each parameter should be determined. Data quality must be assured by abiding by standard operating procedures. Several recognized industry standards for hydrologic monitoring include those in USGS Techniques & Methods, USGS Techniques of Water Resources Investigations, ISO Technical Committee 113, and World Meteorological Organization Operational Hydrology Reports (Hamilton, 2012). Finally, the plan should be developed and improved upon as needed (Caltrans, 2013).

3.3.2. Sampling Schemes. Because it is impossible to sample an entire stormflow volume, many sampling schemes exist in order to capture the variability of contaminant concentrations throughout storm events that use random sampling to provide load estimates for the entire event (Lurry & Kolbe 2000; Holmes *et al.*, 2001). Hydrographic sampling requires periodic sampling throughout a single rain event in order to capture the changing concentrations during the rise, peak, and receding parts of the storm hydrograph (Caltrans, 2013). A mixture of hydrographic sampling and random sampling of storm events can effectively approximate nutrient concentrations and loading during all storm events and is resource and time efficient. Given the limited resources for the current project, this method was selected.

3.3.3. Sampling Methods. Sampling methods are dependent upon the project goals and constraints. Stormwater quality samples are collected using automatic composite sampling or manual grab sampling (Lurry & Kolbe, 2000). Grab sampling advantages include reduced capital cost, less required training of personnel, and storm event flexibility. However, greater personnel presence and labor expenses can accrue for larger projects, and the flow measurement and loading data may be less reliable. Electronic composite samplers will generally have peristaltic pumps and a collection tank so they can take flow-weighted samples throughout the storm event (Lurry & Kolbe, 2000). Advantages to composite sampling include: is more reliable for monitoring the first stages of runoff, can better simplify volumetric loading trends, is generally safer, and requires less personnel and labor (Caltrans, 2013). Disadvantages to composite sampling include greater equipment costs, increased maintenance, incapability for certain pollutant constituent monitoring, and requiring more intense training of personnel (Caltrans, 2013). For this project, grab sampling techniques were used to characterize pollutant loads during storm events.

Grab sampling protocols and techniques vary depending on the water quality parameter being analyzed. Grab sampling protocols for nutrient pollutants TP, TN, and TOC require collection in cleaned, polypropylene plastic bottles with holding times of 28 days, before analysis (Lurry & Kolbe, 2000). In-place field stream water quality sampling methods for flowing water vary, including methods requiring multiple samples across a stream profile or taking a single measurement per profile of a well-mixed stream. Measurements are taken by wading to the center of the channel at the centroid of flow along the vertical axis and, holding the open bottle parallel to the flow to collect the sample (Lurry & Kolbe, 2000).

3.3.4. Laboratory Analyses. Collected stormwater samples are taken to a laboratory to conduct an analysis to determine the concentration.

Total Phosphorus is the measure of all phosphorus species (i.e. orthophosphate and organic phosphorus), within a sample (Caltrans, 2013). Standard methods used are Hach Methods 8190 and 8040 using a spectrophotometer with EPA Method 365.2 for freshwater samples. Each sample undergoes an acid digestion, boiling, and persulfate addition. The acid digestion and heat additions cause the inorganic phosphorus to hydrolyze. Organic forms of phosphorus react with the persulfate and form orthophosphate. After these chemical conversions take place, ascorbic acid is added to the samples as well as molybdate. The phosphates then react with the molybdate to form a blue compound that increases in hue with rising concentrations. The spectrophotometer then corresponds the hue to a known, pre-calibrated concentration (Harper, 2013).

Total Nitrogen measures all species of nitrogen present within a sample such as nitrate, nitrite, and organic nitrogen. Catalytic thermal decomposition and chemiluminesence can be used to determine the concentration from liquid samples. Prepared, filtered stormwater samples are loaded into the analyzer where they are combusted using oxygen and ozone until stable nitrogen dioxide is produced. Ultimately, light from this final product is emitted, quantified, and correlated to the total nitrogen concentration in the original sample (Harper, 2013).

3.4. HYDROLOGIC FIELD MONITORING METHODS

3.4.1. Stream Gauging. Each year, thousands of streams and rivers in the US are monitored by the US Geological Survey (USGS) through the use of a streamgauge for the purposes of collecting water quality data (Lurry & Kolbe, 2000; Bedient *et al.*, 2012). Stage data is collected by manual field measurements using a staff gauge (or other instruments such as a wire weight) as well as continuous measurements using a nearby installation that houses and protects a continuous data recorder, and stage sensor generally in the form of a float system, gas bubble system, or a submersible pressure system (Sauer, 2002; Holmes *et al.*, 2001). Generally the data recorder continuously logs the stage data using an electronic data loggers and can be accessed via telemetry methods (Sauer, 2002; Davis & Masten, 2009). Stream gauges provide measurements that are accurate to the nearest hundredth of a foot and are referenced to a constant elevation, termed a datum. Nearby features with steady elevations, termed benchmarks, are surveyed and related to the elevation of the stilling well and staff gage measurement in order to ensure observations are accurate (Bedient *et al.*, 2012; Sauer, 2002). A picture of a USGS staff gage and a basic monitoring well holding a continuous depth recorder is shown in Figure 3.2.

Figure 3.2. Stream gage and staff gage within a channel.

Considerations must be made when installing a stage monitoring station or choosing a field streamflow monitoring site. The particular location should be located along a straight reach a suggested 300 feet upstream and downstream (Lurry & Kolbe, 2000; Holmes *et al.*, 2001). Flow bypass at the site should be limited, the streambed and banks should be permanent and stable with a limited chance of scour and vegetative growth or disturbance at all levels of the recorded stage. The site should be easily accessible and maneuverable for good discharge measurements. It is ideal to have an upstream pool to control turbulence and flooding and enough channel length downstream without contributing flow to prevent backwater issues (Holmes *et al.*, 2001).

3.4.2. Stream Velocity Measurement. Stream discharge measurements calculate the volume of water moving through a cross sectional piece per unit of time. This is accomplished by determining the stream's cross sectional area and taking velocity measurements along the profile in subareas of the overall cross section (Lurry & Kolbe, 2000, Caltrans, 2013). These cross sectional subareas should be sized so that comprise less than 5 or 10% of total stream cross sectional area (Holmes *et al.*, 2001) to provide a representative estimate of the flow velocity through that sub-section of channel. A measuring tape is extended perpendicular to flow from bank to bank along the cross section and used to determine the spacing and observation point locations (Lurry & Kolbe, 2000).

There is a variety of methods used to determine stream velocity. The principal instruments for measuring stream velocity are the conventional current meters, electromagnetic velocity meters, Acoustic Doppler Current Profilers (ADCP) (Sauer *et al.*, 2002; Holmes *et al.*, 2001). Each method has advantages and disadvantages that must be weighed per project site basis such as level of instrument maintenance required, potential installation costs, and measurement accuracy (Caltrans, 2013).

3.4.2.1. Conventional current meters. Common vertical axis velocity meters include the Price AA meter, Price pygmy meter, Vane Ice meter, and Price OAA meter (Sauer, 2002). The Price pygmy meter and Price AA meter were used to measure streamflow in this monitoring project and are further detailed. These devices are comprised of cups that are affixed to a wading rod or cable and then submerged into a stream and are pushed by the strength of the streamflow, analogous to the motion of a anemometer in its measurement of wind speed. A picture of the pygmy meter used to measure the streamflow velocity for this project is shown in its traveling case in Figure 3.3.

Figure 3.3. Pygmy meter used during flow monitoring.

The number of times the bucket wheel revolves per unit time is related to the linear velocity of the stream using a pre-calibrated equation. The Price AA current meters are either attached to a top-setting wading rod or to a weight and cable for lowering into water. A tail fin is used to stabilize the device parallel to the flow (Lurry & Kolbe 2000). The pygmy meter is two-fifths the scale of the Price AA meter and does not require use of a tailfin (Holmes *et al.*, 2001). The Price AA meters are used for stream depths greater than 1.5 feet or velocities between 0.2 and 12 feet per second and the pygmy meters for depths greater than 0.25 feet or 0.2 feet per sec (Holmes *et al.*, 2001). The revolutions are counted manually using a fiber optic counter that transfers the resounding click of each rotation up the wading rod to a headset, or they can be electronically counted using an automated counter (Holmes *et al.*, 2001). To ensure the current meters are operating at their pre-calibrated capacities, spin tests are routinely conducted. The devices are taken apart for cleaning and oiling, re-assembled and then a spin test is conducted by spinning the bucket wheel. If the wheel spins uninterrupted for a specified amount of time, the device measures properly (Holmes *et al.*, 2001). A picture of an instrument box with a Price AA meter, pygmy meter, headset, and tail fin is shown in Figure 3.4. All of these attachments are fixed to a top-setting wading rod where flow is measured by recording ticks are produced by the bucket wheel.

Figure 3.4. A field instrument box storing Price AA and pygmy current meters, headset, stopwatch, and linear to radial velocity conversion chart.

3.4.2.2. Other velocity determination methods. Though not used in this project, many other velocity determination methods and instrumentations exist. Common horizontal axis meters include the Ott meter (developed for use in boating), the Hoff meter (generally used for measuring pipe flow), and the Haskell meter (for swiftly moving streams). Marsh McBirney probes use electromagnetics to determine streamflow velocity. There are also Acoustic Velocity Meters that use sound waves reflections across stream channels and stream cross sectional areas to determine stream discharge and are generally used when conventional methods are not possible (Bedient *et al.*, 2012; Holmes *et al.*, 2001). A common method used by the USGS for determining stream discharges is the use of Acoustic Doppler Current Profilers, which use sound waves to detect the movement of particles within streamflow and relate those speeds with the streamflow linear velocity and multiply the velocities by the channel cross sectional area (Sauer, 2002; Caltrans, 2013).

Depending on the site conditions, it may not be possible to monitor using direct velocity measurements. Other less invasive, mathematically-based methods may be more applicable, such as using Manning's equation or indirect measurements such as the slope area method (Sauer, 2002). Surface velocity measurement includes the ball and float method. The ball and float method uses a float that is placed on top of the water surface and records the time it takes to travel a certain distance is recorded and used as a streamflow velocity measurement (Caltrans, 2013). A similar method implores tracer dyes and dilution methods (Sauer, 2002). Additionally, controlled flow structures such as calibrated flumes or weir structures can be used, and, for smaller flows, volumetric capture methods are useful as well (Caltrans, 2013; Sauer, 2002).

3.4.2.3. Average velocity determination. Open channel streamflow velocities along the vertical axis are not constant with depth, so they require the use of an averaging method to be used with stream velocities measurements across the channel. Methods include: the Vertical-Velocity method, Two-Tenths method, Six-Tenths method, Sub-surface Velocity method, and the Two and Three Point methods, with the Six-Tenths and Two-Point methods being used most commonly (Holmes *et al.,* 2001). The Two-Point method is regarded as the most accurate in calculating average flow, but is not to be used in depths less than 2.5 feet (Lurry & Kolbe, 2000). In the Two-Point method, velocity measurements are taken at 0.2 and 0.8 of the stream depth at that point and then averaged to yield a measurement representative of the mean flow represented by the velocity profile (Bedient *et al.,* 2012, Holmes *et al.,* 2001). The Six-Tenths method is used for shallower channels, in which the stream velocity measurement is taken at 0.6 of the channel depth and used as the average flow for that vertical segment of the stream (Holmes *et al.,* 2001). Due to the shallow channel depths, the Six-Tenths method was used in this monitoring study. A picture demonstrating the proper set up of a top-setting wading rod and pygmy meter to conduct field streamflow velocity measurements is shown in Figure 3.5.

3.4.3. Stream Discharge Calculation. The stream cross section is divided into a number of subsections. Discharge is calculated in each subsection by multiplying the measured average streamflow velocity and the known cross sectional area of the subsection using the continuity principle (Bedient *et al.*, 2012). The total stream discharge is the sum of the subsection discharges. The time series of continuous discharge data can then be computed from a combination of a rating curve (created from the discharge measurements) and the time series of collected stage data. This time series is used in hydrologic and water quality analyses of the watershed.

Figure 3.5. Streamflow velocity measurements are taken using Six-Tenths method.

3.4.3.1. Rating curve development. The direct, continuous onsite measurements of discharge is not feasible, so the continuous time series of discharge is determined from other surrogate data, such as stage, which can be easily collected continuously. The surrogate of stage requires the use of a rating curve to compute the discharge from the stage. The rating curve is applied with appropriate adjustments for any shifting of the hydraulic controls which would skew the relationship from the observed measurements

(Holmes *et al.*, 2001; Sauer, 2002). Rating curves are developed by plotting the measured discharges on the abscissa and corresponding stage reading on the ordinate on logarithmic paper so a linear relationship can be drawn through the points (Bedient *et al.*, 2012). Low flows can be extrapolated using a linear plot with rating curve section beginning at the stage at which zero flow occurs to the first point on the curve (Holmes *et al.*, 2001).

3.4.3.2. Stage-Slope-Discharge rating. For some monitoring sites with variable backwater conditions, more complex computational water resources investigation methods are used to calculate discharge that additionally use the drop in water level elevation between two gage locations to provide an adjusted or corrected discharge (Holmes *et al.*, 2001, Sauer 2002). These ratings use observed channel stage heights and water elevation differences between base and auxiliary gages during time of measurement to create a stage-fall rating curve. The rating curve along with the stage-discharge curve created for the channel uses concurrent measurements of the water surface elevation falls and stage levels to produce an adjusted discharge using the correction equation shown (Sauer 2002):

$$Qadj = Qr\sqrt{\frac{Fm}{Fr}} \tag{3.1}$$

where Q_{adj} is the corrected discharge in m^3/s, Q_r is the discharge rating in m^3/s, F_m is the observed water elevation fall between up and downstream gauges in m, and F_r is the water elevation fall rating in m. This stage-slope-discharge rating method was incorporated at both monitoring sites to take into account backwater effects from the nearby receiving lake that filled the channels during larger precipitation events.

3.5. NUTRIENTS IN URBAN STORMWATER

3.5.1. Nutrient Pollutant Concentrations. Nutrients in stormwater, such as phosphorus, nitrogen, and organic carbon, are expressed in terms of concentration, mass loads within a discharge, or yields over a specific drainage area and are often highly variable due to the complexity of watershed systems and a variety of factors including seasonality, land-use characteristics, climate, topography, and many others (Khwanboonbumpen, 2006; Roberts & Prince, 2010). During intense storm events, the surface is agitated by the rain and collects and carries particles resulting in greater concentrations of suspended particles. As P tends to affix to suspended sediment, it can be directly related to suspended sediment concentrations as well as streamflow discharge (Leitz, 2009; Mallin *et al.*, 2009).

3.5.2. First Flush Effects. Depending on a large number of site conditions such as land use type, climate, soils types, topography, stormwater management practices, and rain event characteristics, pollutants are not available at uniform rates throughout the sampling period, creating temporal variations of stormwater quality (Tsihrintzis & Hamid, 1997; Khwanboonbumpen, 2006). Generally, the initial portion of the storm event has elevated levels of contaminants, termed the first flush effect (Sansalone & Cristina, 2004; UACDC, 2010). Additionally, in many studies seasonal first flush effects have been noticed in places such as California and Perth, Australia, where during dry seasons without rains to wash contaminants downstream, pollutants will accumulate throughout the season and runoff in the initial storm events of the wet season (Caltrans, 2013; Khwanboonbumpen, 2006). The first flush effect is not well-defined or easily monitored, with many studies reporting no correlation between antecedent dry periods and an observable increase in contaminant concentrations (Khwanboonbumpen, 2006).

3.5.3. Modeling Pollutant Loads. A common technique to estimate stream water quality nutrient loading involves the creation of a linear regression model. This assumes that a relationship between the log constituent concentration and log stream discharge exists (Cohn *et al.*, 1989). The model is, in most general form, as shown in Equation 3.2:

$$\ln (C) = B_0 + B_1 \ln (Q) \tag{3.2}$$

where C is the constituent concentration, Q is the discharge, and B_0 and B_1 are coefficients that can be determined with an appropriate sample data set. Once a model trend is formed, constituent concentrations can be determined for any discharge value. The corresponding load can then be calculated by multiplying concentration, discharge, and an appropriate conversion factor (Cohn *et al.*, 1989). Using this method, the total required samples is reduced and still yields a substantial degree of precision in estimating loads. Additionally, since entire data sets are used to create the model instead of event observations, individual sampling event error and bias are reduced (Leitz, 1999). This relationship can be used to model nutrient concentrations at all flows if a discharge time series record exists. These simplistic loading models are used to create flow duration curves that can be used to estimate annual loading for use in TDML determinations (EPA, 1997).

Rating curve methods are statically biased to underestimate loads (Cohn *et al.*, 1989) and more detailed water resource investigation techniques exist to more accurately calculate fluvial sediment event loading (Porterfield, 1972). Graphical methods relating the ratio of maximum discharge to discharge to the ratio of maximum concentration to concentration can be used to create a concentration curve estimate that follows the shape of the storm event hydrograph. Loads can be determined by integrating the area under the concentration curve (Porterfield, 1972).

Ultimately, the results from nutrient loading analyses are used in many comprehensive models to quantify urban stormwater runoff pollution, including the most popular Storm Water Management Model (SWMM), Storage Treatment Overflow Runoff Model (STORM), and models used by the Federal Highway Administration that use GIS information, watershed properties, rainfall rates, water quality parameter information, and various other inputs to spatially and temporally assess stormwater runoff quality from urban environments (Tsihrintizis & Hamid, 1997).

3.5.4. P Loading into Catchments. The quality of freshwater lakes is heavily dependent upon external and internal P loading (Elser, 2013; EPA, 1993). Indeed, from lake P concentrations alone, the trophic state of a lake can often be determined as being hypereutrophic, eutrophic, mesotrophic, or oligotrophic with oligotrophic lakes having concentrations less than 0.01 mg/L and eutrophic lakes greater than 0.025 mg/L (EPA 1993; Davis & Masten, 2009). External P loading exists in either dissolved or particulate form with dissolved loads available to primary producers and particulate forms settling to the bottom of the catchment (Steinman *et al.,* 2007). Once P reaches the sediment, various chemical and biological processes take place, including reactions with calcium carbonate, adsorbing to iron hydroxides, clays, alum, or calcite, where is stays biologically unavailable in the sediment until additional processes cause its release (Sondergaard *et al.,* 2001; Khwanboonbumpen, 2006).

3.5.4.1. Internal loading processes. Internal sediment release mechanisms are intricate and not easily modelled, but P is often transported back and forth between the water column and sediment depending on biological factors, such as mineralization and bacterial activity; chemical factors, such as redox conditions, pH, iron to P ratios; and physical factors such as sediment perturbation by wind and mixing (Sondergaard *et al.*, 2001). Inorganic P forms typically bind to sediment, iron such as iron (III) hydroxides, strengite, and vivianite, aluminum as alum or variscite, or calcium compounds such as hydroxyapatite, monetite, and calcite. The probable mobile P are the fractions that are loosely sorbed, iron-bound, or redox-sensitive (Sondergaard *et al*. 2001). Studies have shown oxidative conditions to prevent internal P loading from sediment and reductive conditions with higher pHs has shown to increase P fluxes from the sediment (Christophoridis & Fytianos 2006). Organic forms are generally immobilized and buried within the sediment. In shallow lakes, the sediment and water surface area to volume ratio is larger, and therefore has been noticed that P interactions between the two layers increase (Sondergaard *et al.*, 2001; Lung *et al.*, 1976). For this same reason, effects of nutrient loading can be attenuated in deeper lakes (Abell *et al.*, 2011). Seasonal trends of internal P loading include a negative flux of P being released from the sediment and into the water column during summer and a positive flux during winter times. This reaction is believed to be controlled by temperature and biological activity within the lake (Sondergaard *et al.*, 2001; Lung *et al.*, 1976). A schematic showing P transport within lake systems is shown in Figure 3.6.

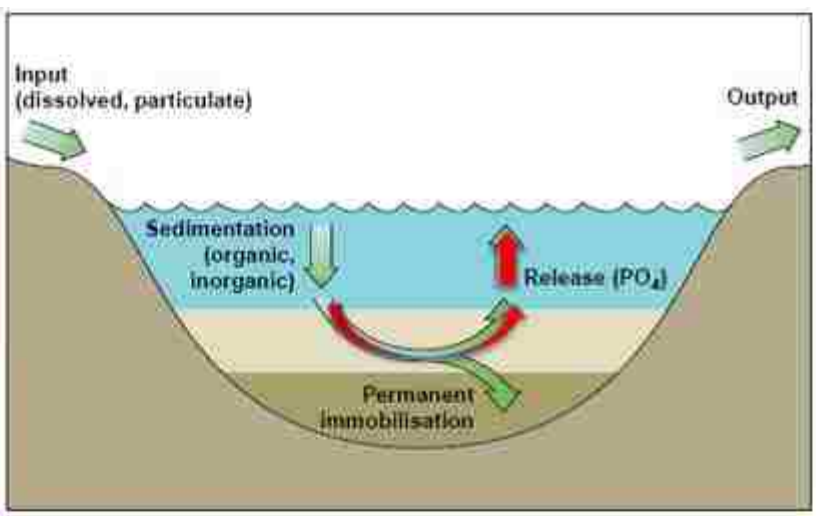

Figure 3.6. P movement within a lake system (after Sondergaard *et al.*, 2001).

3.5.4.2. Trophic response modeling. Lake models predicting trophic state based upon P loading began in the late 1960s with Vollenweider's simple model using P concentrations and lake hydraulic retention times to predict observed trophic states in various lakes (Lung *et al.,* 1976). Today, many deterministic models exist involving known Secchi depths, chlorophyll content, algal concentrations, and P concentrations in order to predict reductions in P loading that vary in complexity (Lathrop *et al.,* 1997; Steinman *et al.,* 2007). Additionally, lake water quality managers have successfully used mass balance models involving known input parameters such as P settling rates, desired lake concentrations, lake volume, and discharges to predict acceptable loading rates to avoid eutrophication (Davis & Masten, 2009).

3.5.4.3. Trophic improvements. Due to predominate internal P loading processes within shallow lakes, reducing P inputs into the lake may not immediately improve water quality (Lathrop *et al.,* 1998, Lung *et al.,* 1976). It therefore becomes important to understand both internal and external loading rates when developing a remedial plan. Dredging or physically removing the sediment from P ridden lakes and adding alum or iron to precipitate sediment P are two widely used methods to reduce internal P loading into eutrophic lakes (Sondergaard *et al.*, 2001, Davis and Masten 2009) and the implementation of upstream BMPs can effectively reduce external P loading to lakes (Steinman *et al.*, 2007, Tsihrintzis and Hamid 1997, UACDC 2010). The prediction of future trophic level and water quality within lakes after disturbances to equilibrium can be difficult as many processes affect the length of the recovery period (Sondergaard *et al*. 2001).

3.6. URBANIZATION AND LAND-USE

3.6.1. Impervious Surfaces. Human driven land-use alteration from natural to industrialized landscapes is the defining factor of urbanized environments (Paul & Meyer 2001; Steinman *et al.,* 2007; Carey *et al.,* 2013). Methods to determine the degree of environmental urbanization include mapping roadway density or human population density (NRC, 2008). Additionally, for purposes of quantifying impacts related to stormwater, the percentage of land covered in impervious surfaces or a ratio of impervious to pervious land area is most commonly used (Brabec *et al*., 2002; Singh &

Chang, 2014). This ratio of imperviousness can be relatively easily calculated through the use of aerial mapping and has therefore become a key parameter in city and watershed planning (NRC, 2008). Many studies have proven the biological and physical health of water bodies to be directly related to upstream watershed percent impervious cover as a result of the changes in stormwater quality and quantity (Tsihrintzis and Hamid 1997, Sun and Lockaby 2012). As the percentage of contributing watershed impervious surface increases, downstream waterways and bodies must absorb more runoff that becomes increasingly more contaminated with urban pollutants (Paul & Meyer 2001). Natural streams can only handle certain deviations from natural flows until degradation begins to occur. Generally downstream impairment can be seen in contributing drainage areas of 10% imperviousness and conditions worsen as the degree of urbanization increases (Mallin *et al.,* 2009; Brabec *et al.,* 2002). Additionally, studies have shown that watersheds with forested areas of at least 15% see downstream impacts mitigated (Brabec *et al.*, 2002). However, specific thresholds may vary depending on the natural watershed characteristics such as size, vegetation, geology, and soils (Sun & Lockaby, 2012; Brabec *et al.,* 2002; Carey *et al.,* 2013). Ultimately, the stormwater quality and characteristics of urban watersheds sharply contrasts to those of forested or undeveloped land areas (Sun and Lockaby 2012).

Impervious surfaces have been effectively categorized further into being directly connected to the water transport system or disconnected (Brabec *et al.*, 2002). Surfaces that are directly connecting (DCIA) contribute surface runoff into receiving waters, whereas some impervious surfaces, such as roofs, drain onto pervious areas. It was noticed in a study in Miami, Florida that runoff from DCIA comprised 44% of the watershed but contributed 72% of the stormwater runoff (Carey *et al.*, 2013). Additionally, some surfaces such as bare compacted soil and gravel driveways have shown to yield the same runoff volumes as impervious surfaces (Brabec *et al.*, 2002). These distinctions between effective and non-effective impervious surfaces add further complexity to land managing and stormwater runoff modeling as readily available data sets and land use maps often do not distinguish between surfaces (Tsihrintzis & Hamid, 1997).

3.6.2. Effects on Nutrient Loads. Many urban land covers generate N and P, and the re-configuration of these land areas from their hydrologic natural state enhances the levels of nutrient inputs into surface flows (Carey *et al.,* 2013; Roberts & Prince 2010). These relationships between land-use type and stormwater runoff quantity and quality have allowed land managers to develop modeling tools that project the hydrologic impacts and stormwater quality of urban environments (EPA, 1993; Roberts *et al.,* 2009). Water quality models, such as the aforementioned SWMM, Source Loading and Management Model (SLAMM), Hydrologic Simulation Program-FORTRAN (HSPF), and the SPAtially Referenced Regressions on Watershed attributes (SPARROW) that are used by watershed management groups and regulatory agencies such as the EPA, rely on land cover and land use information from geospatial data in order to predict nutrient loads in stormwater runoff (Tsihrintzis & Hamid, 1997, Roberts & Prince, 2010). Additionally, spatial scale has also shown to affect the correlation of runoff water qualities with land cover with better correlations increasing with finer sub-basin scales (Singh & Chang, 2014). It has been shown that further breaking down land use areas and looking at landscape characteristics such as arrangement, position, and orientation of various landscape elements improve accuracy in predicting runoff quality (Roberts and Prince 2010). For example, implementing vegetation near riparian areas where natural filtration processes can reduce pollutants and debris directly contacting urban streams and/or increasing the areas with connected tree cover throughout watershed are two methods shown to produce better stormwater water qualities (Carey *et al.,* 2013; Brabec *et al.,* 2002). In fact, 100-300 feet riparian buffers, strips of hydric soil and facultative vegetation, can filter between 50 and 85% of urban pollutants from stormwater (Carpenter *et al.,* 1998; UACDC, 2010).

It is difficult to determine the contributions from particular sources and land use covers, though many studies have attempted to quantify using intense monitoring or modeling methods (Waschbusch *et al.,* 1999; Khwanboonbumpen, 2006; Carey *et al.,* 2013). Additionally, there seems to be no single land-use alteration or BMP strategy that can ensure certain improvements in water quality as each watershed is a complex system (EPA, 1993; Brabec *et al.,* 2002). Expected nutrient concentrations in US urban stormwater runoff average 0.26 mg/L for TP and 2.0 mg/L for TN. However, increasingly elevated levels can be expected during flows with greater discharges (Carey *et al.*, 2013). These runoff concentrations represent a composite of inputs from various sources (Barth, 1995). Studies show roadways and parking lots that accumulate automobile derived pollutants contribute the highest levels of pollutant concentrations to runoff loads of all impervious surfaces (NRC, 2008). In a detailed study by Waschbusch *et al.*, in an urban residential watershed in Madison, WI, it was observed that the lawns and streets alone contributed to 80% of the dissolved phosphorus load in stormwater draining into lakes Wingra and Mendota (1999). Pervious turfgrass lawns allow infiltration of surface runoff and, therefore, reduce runoff volumes; however, they are sources of fertilizers and organic matter can be released into stormwater flows (Carey *et al.*, 2013).

3.7. STORMWATER BMPS

Unlike point sources of urban stormwater pollution that are mitigated via regulatory measures, non-point source pollution is controlled using Best Management Practices (BMPs), which can either be structural or non-structural (Tsihrintzis and Hamid 1997, EPA 1993).

3.7.1. Non-Structural BMPs. Many stormwater management techniques are non-structural measures and are based upon housekeeping practices that control sources (Tsihrintzis & Hamid, 1997). Controlling fertilizer use, by introducing no P fertilizer use policies (Barth, 1995); educating the public on healthy lawn care practices, such as fertilizing during the correct time of year and using correct application rates (Carey *et al.*, 2012); promoting the use of residential rain barrel and rainwater storage by offering free rain barrels to interested citizens; urging the public to correctly handle and dispose of pet waste via public outreach; organizing appropriate and regular street sweeping programs, spreading public awareness concerning environmentally friendly leaf litter re-use practices, introducing zoning policies that restrict development densities and land area configurations; and limiting the amount of road grit applied to streets during winter are all examples of practical, effective stormwater BMP strategies (Carpenter *et al.*, 1995; Carey *et al.*, 2013; NRC, 2008).

3.7.2. Structural BMPs. Urban planners are challenged with redesigning built city infrastructures to incorporate structural BMPs (commonly termed "green infrastructure") to more effectively reduce downstream impacts from stormwater flows (Palmer *et al.*, 2015). Green infrastructure uses a network of ecologically-based, natural features that mimics the same functions as built gray infrastructures while preserving pre-development hydrological and ecological conditions (Palmer *et al.*, 2015, Deitz 2007). A variety of GI implements aids in different components of a BMP network including using techniques that infiltrate, filter, store, and evaporate stormwater runoff near its origin (UACDC, 2010). Runoff conveyance features that transport stormwater without exacerbating flows, for example swales and level spreaders; pre-treatment implements incorporating filtration buffers such as grass filter strips, filter cloth barriers, and stilling basins, (Tsihrintzis & Hamid 1997) and retention and infiltration facilities bioretention gardens and constructed wetlands, use both are all examples of structural BMPs used to mitigate and reduce flows (EPA, 1993; UACDC, 2010).

3.7.2.1. Planning and design. For effective implementation of GI, an understanding of the watershed hydrology and stormwater quality characteristics must be established (EPA, 1993). Designs are generally based upon the area's climate and precipitation trends, taking into consideration the runoff volumes expected from varying design storm events (Davis *et al.*, 2009; Bedient *et al.*, 2013). In GI design, emphasis is placed upon connectivity of green spaces and implements, incorporating redundancy, resiliency, and distribution (Dietz, 2007; UACDC, 2010). With redundancy of implements, performance is enhanced and chances of failure are reduced. Increases in resiliency arise from using multiple implements to fully realize the benefits GI has to offer (UACDC, 2010). Dispersing GI spatially throughout a watershed will increase optimal retention capacities and prevent potential concentrations of pollutants (Brabec *et al.*, 2002, UACDC, 2010).

For GI implements that use infiltration, depth to ground water and soil properties are also relevant to design (EPA, 1993). Soils with naturally high hydraulic conductivities are more conducive to stormwater reduction within GI implements involving filtration (Davis *et al*., 2009). Additional site considerations may include feasibility of retrofitting a BMP structure over existing stormwater control structures (UACDC, 2010).

3.7.2.2. Green roofs. Green roofs are vegetated gardens built on top of buildings that collect rainwater and atmospheric pollutants, attenuate flows, and reduce stormwater volumes utilizing evapotranspiration from plants (Carey *et al.,* 2013; Deitz, 2007). Green roofs have also shown the ability to regulate building temperatures, by providing a layer of insulation, and to mitigate urban heat island effect through evapotranspirative cooling (Carey *et al.,* 2013; Gibler, 2015). Green roofs provide the best stormwater retention benefits during intense, short-duration storms in areas prone to flash flooding events, and in temperate climates (UACDC, 2010). Green roofs have been shown to reduce stormwater volumes by 50% (UACDC, 2010) with some studies showing consistent retention capacities between 60 and 70% (Deitz, 2007). However, rainfall intensity, antecedent soil moisture conditions, roof gradients, and other weather conditions can all effect retention capabilities (Harper, 2013). In a study by Harper, a vegetated green roof was capable of reducing stormwater runoff by 60% over an eight month study, though the media leached significant concentrations of TP and TN, 30 mg/L and 60 mg/L respectively, into the runoff that eventually stabilized to reduced concentrations (2013). Green roofs are comprised of several media layers with different infiltration capacities and purposes including a vegetated surface layer, growing media, and filter and drainage layers (UACDC, 2010). A diagram of these layers is shown in Figure 3.7.

3.7.2.3. Pervious paving. Porous, permeable, or pervious paving allows the functionality of a rigid surface for transportation use, but also allows the vertical flow of water through concrete, asphalt, or interlocking pavers (UACDC, 2010). Generally the pavements are comprised of an underground geotextile-lined course drainage stone base overlain by a specially designed or mixed asphalt in which the finest aggregates are intentionally removed (Tsihrintzis & Hamid, 1997). A schematic showing the basic designs and pertinent layers to a pervious pavement are depicted in Figure 3.8. The amount of runoff and drainage area contributing to the pavement determines the design depth at which the varying levels of porous material must be built to ensure proper infiltration or runoff (EPA, 1993). Pavements can have varying degrees of porosity with types such as modular precast pavers, poured in place systems, porous asphalt, porous concrete, and gravel (Dietz, 2007, UACDC, 2010). In a study located in Washington, properly designed and maintained permeable pavements retained 100% or nearly all runoff during a six year period (Dietz, 2007). It is recommended to use pervious pavements along light traffic areas such as parking lots or largely foot traffic streets (EPA, 1997; UACDC, 2010) and within soils with naturally high infiltration rates, though studies have shown porous pavements to remain effective in clayey soils with lower hydraulic conductivities (Dreelin *et al.*, 2006). Large vacuums and high pressure jets are required to maintain the original porosity and stormwater removal efficiencies of pervious pavements (UACDC, 2010, EPA 1993; Dreelin *et al.*, 2006).

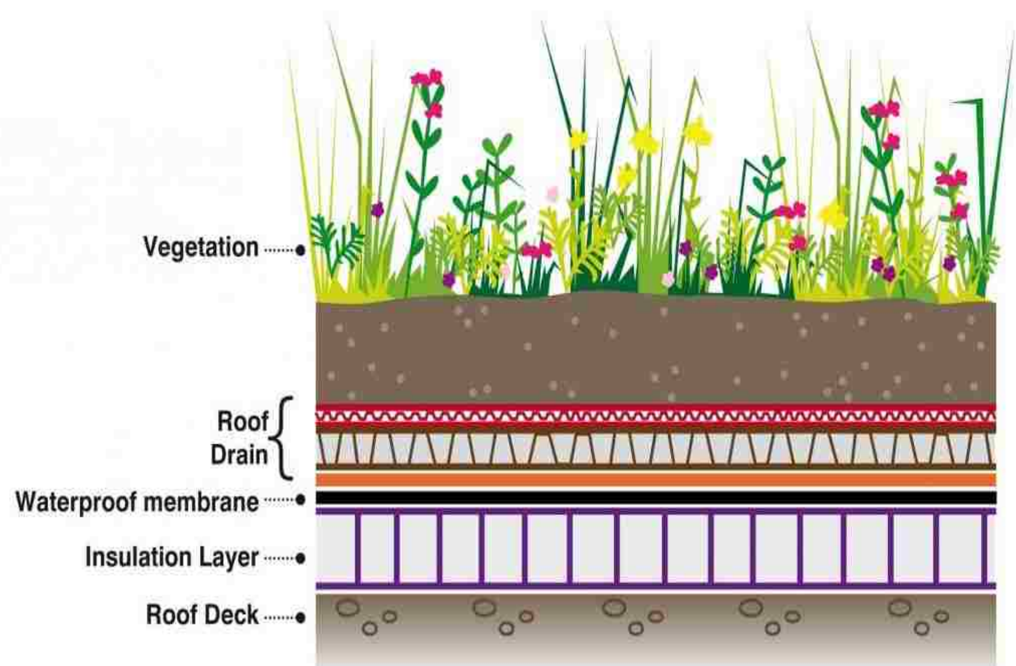

Figure 3.7. Schematic of green roof design (lindumgreenroofs.co.uk).

Permeable pavement systems are effective tools for removing suspended solids and nitrogen and decreasing the levels of urban stormwater pollutants that are generated by impervious surfaces (Scholz & Grabowiecki, 2007). For example, runoff from permeable pavers used in Connecticut driveways showed significant reductions in measured pollutant concentrations compared to runoff from asphalt driveways (Dietz, 2007). A Villanova study concluded that pervious concretes could effectively remove water quality constituents such as chloride and copper without contaminating the groundwater beneath (Dietz, 2007). Additionally, permeable pavements show the ability to degrade oil and diesel fuel contaminants by operating as hydro-carbon traps and powerful bio-reactors using naturally occurring microbial communities that develop within the pavement matrix (Scholz & Grabowiecki, 2007).

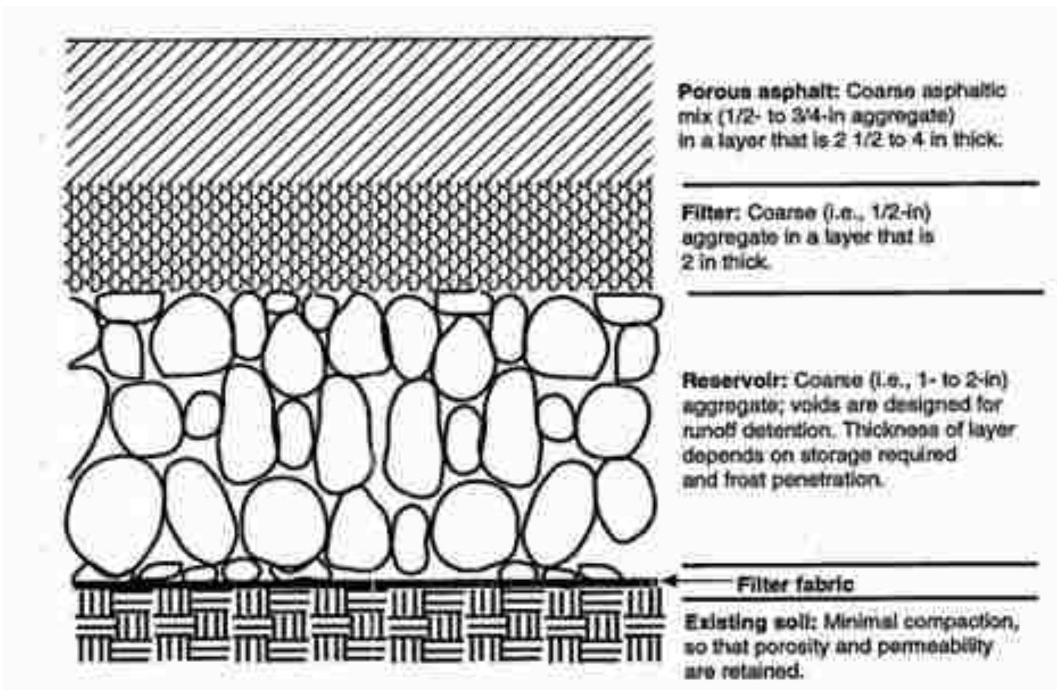

Figure 3.8. Schematic of pervious paver design (EPA, 1993).

3.7.2.4. Bioretention facilities. Bioretention facilities, such as rain gardens and bioswales, are depressions designed to mitigate pollutants through the utilization of pervious soils and vegetation, termed bioinfiltration (UACDC, 2010), and have become the most preferred green infrastructure implement for LEED certification (Davis *et al.*, 2009). Detaining stormwater runoff volumes on-site rather than channelizing and moving water away allows for the filtration and capture of pollutants before they are introduced to receiving water bodies (Steinman *et al.*, 2007). Bioretention facility stormwater retention capabilities are strong, though dependent upon storm intensities, antecedent moisture conditions, and season (Davis *et al.,* 2009). Hunt *et al.,* showed annual outflow volumes being reduced to less than half the inflow volumes (2006), and a study in Villanova showed a bioretention facility designed to retain 1 in of runoff, removing 80% of stormwater volume into the watershed (Davis *et al.,* 2009). Due to the volume decreases, bioretention facilities can help restore natural hydrologic conditions to urban watersheds by significantly reducing peak flows and increasing time of concentrations (UACDC, 2010; Davis *et al.,* 2009). Additionally, bioretention remains an effective tool to improve water quality through employing sedimentation, filtration, chemical sorption, biological activity, and heat transfer processes to remove pollutants (UACDC, 2010).

Implementing infiltration basins and GI implements that incorporate bioretention allows for utilizing soil and vegetation abilities to provide nutrient sinks for urban stormwater (UACDC, 2010). For planning purposes, it is assumed that if retention is correctly designed to retain 0.5 to 2 in of stormwater required in watershed management designs, the resulting water quality improvement will also satisfy the less specific pollutant mitigation goals required (Davis *et al.,* 2009). However in some circumstances, biological implements can serve as nutrient sources that elevate N and P levels (Carey *et al.,* 2013). For example, in field studies of various bioretention facilities by Hunt *et al.,* (2006), P removal efficiencies ranged from 65% to additions of 240% with varying influent concentrations playing a role in performance. In a study by Brabec *et al.,* (2002), P removal capacities in detention ponds across Washington, Florida, and North Carolina were found to vary from 13-66% with most BMPs seeing less than 50% reduction rates. Other field studies of BMPs in Maryland by Davis *et al.,* (2009) showed P removal rates of 77% and 79%. In a study by Hunt *et al.,* (2006) the annual mass nitrate-nitrogen removal rate for bioretention basins was 40%, outflow to inflow runoff volumes ranged from 0.07 to 0.54 depending on seasonal conditions, and P removal rates were also evident though greatly depended on the P index of the fill media used. Ultimately, bioretention nutrient removal efficiency is dependent upon influent stormwater quality pollutant concentrations, as biorentention media will leach some nutrients into the effluent stormwater (Dietz, 2007). For example TP effluent concentrations from field and lab bioretention facilities were found to range from 0.05 to 0.18 mg/L (Davis *et al.*, 2009). Therefore, stormwater with elevated TP concentrations beyond 0.18 mg/L will have greater TP removal efficiencies, and influent concentrations below this observed effluent threshold will see negative removal efficiencies as the "clean" stormwater collects nutrients from the BMP media.

Little research has been conducted in understanding bioretention benefits in mitigating temperature effects and E. coli (Dietz, 2007; Davis *et al.*, 2009). Due to the general speed at which water percolates through the BMP, the temperature effects may not be significant (Dietz, 2007). In limited field and laboratory studies in North Carolina, significant E. coli removal efficiencies of approximately 70 and 91.6% were observed (Davis *et al.*, 2009). Ultimately, more definitive research must be completed for a better understanding of the impacts bioretention has on the mitigation of pathogens and increased temperature from urban heat island effects before they can be standardized as stormwater management tools.

Designing bioinfiltration systems is difficult as it requires the integration of principles from surface and subsurface hydrology and hydraulics, horticulture, soil science, and landscape architecture in order to provide benefits to baseflow and groundwater recharge, pollutant mitigation, erosion reduction, and peak flow attenuation (Davis *et al.*, 2009). Two driving factors in biofiltration design are basin size and soil permeability, which determine the runoff retention and mitigation capacities of the implements (EPA, 1993). Additionally, considerations regarding drainage area, pre-treatment requirements and pollutant removal rates, surface area sizing, media depth and composition, vegetation, maintenance, and overflow and underdrain design must be standardized according to the quantity and quality of runoff it is expected to receive (Davis *et al.*, 2009). Widespread use of bioretention is difficult due to groundwater contamination concerns, lack of design guidance, and long-term maintenance and upkeep (Brabec *et al.*, 2002, Tsihrintzis & Hamid, 1997). Careful attention should be taken to provide a porous soil medium that has low P levels as P is known to leach into effluent flows (Dietz, 2007).

Unlike detention ponds and bioswales, rain gardens are not designed to store and hold water for longer periods of time (Tsihrintzis & Hamid, 1997). Rain gardens are generally designed on smaller scales (500 square feet or less) and provide adequate stormwater collection for smaller rain events. (EPA, 1997). Rain gardens are comprised of organic sandy soils to allow permeation of water and promote soil ecologies, may require an underdrain system for poorly draining soils, may include a permeable geotextile membrane, and have native flowers, shrubs, and grasses that use phytoremediation processes to mitigate first flush pollutants (UACDC, 2010). A schematic of a typical rain garden with an amended soil layer is shown in Figure 3.9.

Figure 3.9. Schematic of a typical rain garden set up (after OSU 2010).

3.7.3. Benefits. In addition to those observed in stormwater management, GI provides many ecosystem benefits including increasing biodiversity by providing refugia and habitat, aiding in food production, providing better air quality and climate regulation, allowing for sustainable energy production, cycling nutrients, and promoting clean water and healthy, less-eroded soils (Palmer *et al.*, 2015, UACDC, 2010).

Green areas in cities, particularly those with larger trees, have been proven to mitigate harmful temperature increases in urban areas, known as the urban heat island effect, by providing shade and removing heat through evapotranspiration (UACDC, 2010). In a study estimating the monetary benefits of reduced and mitigated stormwater runoff volumes of Georgian wetland forests, each hectare of wetland was valued at $11,588 to $20,490, depending on its proximity to urban environments (Sun & Lockaby, 2012). GI not only provides economic benefits in stormwater management, but also yields anthropocentric benefits by promoting public health and increasing social aesthetics (Tzoulas *et al.*, 2007, UACDC, 2010). Natural areas and the ecosystem services associated with GI increased mental health and wellness in citizens in a multitude of experimental studies, ultimately providing increases in socio-economic activity at community-wide scales (Tzoulas *et al.*, 2007). Though capital costs implementing GI are often greater than traditional methods, ecosystem benefits appear far reaching and increase over time (UACDC, 2010). Indeed, the non-monetary benefits of GI are too often underestimated as the realized benefits of improving environmental, health, and social conditions are not easily quantified (Palmer *et al.*, 2015).

PAPER

I. Assessing an Impaired Urban Watershed to Project Upstream Stormwater Best Management Practices

Abstract: Over the past century, the routine mismanagement of the earth's hydrologic cycle has resulted in the deterioration of much of the world's surface waters through the conversion of natural landscapes into urban, impervious areas that produce destructive stormflows. To begin addressing this problem, the urban watershed must be redesigned from its current focus of stormwater conveyance to emphasize pre-development hydrologic conditions using Best Management Practice (BMP) watershed planning strategies that include Green Infrastructure (GI).

This study assessed the hydrology of a specific impaired urban watershed and the benefits GI may have to improve its quality; however, the study's methods can be applied to any urban watershed. Portions of Rolla, Missouri including much of the Missouri S&T campus are channeled into the impaired urban waterbody Frisco Lake. The lake is plagued with poor water quality, eutrophication, and a substantial fish kill in 2014. Lake phosphorus (P) concentration served as a water quality indicator due to its pertinence to algal growth. Since the fall of 2014-15, traditional lab, field sampling, and hydrologic monitoring methods were used to continuously model stormwater runoff flows and corresponding P loads at inlets to the lake. TP yields of 17 and 31 kg/ha/yr and mean-annual concentrations of 0.43 and 0.46 mg/L at the stormwater outfalls were used in mass balance modeling to determine a required 40% loading reduction to improve lake quality. Stormwater modeling results were incorporated into a watershed improvement plan.

The City of Rolla has plans to finance a temporary, lake-based solution to improve lake water quality; however, this study provides additional site-specific upstream GI implement recommendations more capable of providing lasting stormwater management and watershed improvement. Effectively realizing the full hydrological, ecological, social, and aesthetic value of GI implements requires watershed-specific design and planning that includes an assessment of stormwater quality and downstream impacts.

1. Introduction

As global populations rise and continue to migrate from rural to urban areas, land is converted from its natural state to human-dominated, urbanized environments [32, 5]. The construction and development of cities has required the conversion of vegetated, pervious, green terrain to man-made, impervious, gray landscapes that disturb the area's natural hydrology by increasing stormwater runoff [31]. Traditional urban stormwater management emphasizes the rapid removal of runoff to avoid harm to human and property by channelizing stormflows into downstream environments [23]. These stormflows are characterized by less attenuated peak flows and discharges that contain harmful urban pollutants, and thereby disrupt the geomorphology, water chemistry, temperature, nutrient cycling, and biotic integrities of downstream environments [21, 23, 32].

Widespread destructive pollutants discharged from urban watersheds include nutrients nitrogen (N) and phosphorus (P), sediments, and potential pathogens as indicated by *E. coli*. Nutrients N and P are among the most problematic and are derived from sources such as residential fertilizers, atmospheric deposition, municipal and industrial discharges, pet waste, yard litter, and eroded sediment [8, 26]. Excessive nutrient loads in stormwater runoff collect in downstream catchments and cause eutrophication [1, 16]. For many urban freshwater lakes, P inputs cause eutrophic algal growth that is responsible for creating the hypoxic conditions that lead to fish kills and poor water quality [8, 13]. Extensive, large-scale watershed monitoring programs have been carried out through collaborations of groups and agencies that provide hydrologic stormwater quality information for local, state, and federal decision makers and other interested parties to use in developing management solutions [15, 27, 28]. As a result, common water flow and quality evaluation tools are designed for larger scales, making smaller urban watershed assessments cost prohibitive. A limited number of studies have employed subcatchment-scale urban water quality monitoring methodologies and their usefulness in understanding stormwater characteristics to provide practical management strategies. Overall, better assessment tools are needed to understand urban water quality impacts at all scales to assuage the ongoing degradation to the world's surface waters.

Stormwater Best Management Practices (BMPs) are used to mitigate urban non-point source pollution responsible for degrading downstream water bodies [8]. Non-structural BMPs include practices such as fertilizer application control, street sweeping, and public outreach encouraging proper yard and pet waste management are used to improve urban watersheds [7, 28]. Green infrastructure (GI) is an increasingly popular structural BMP that reconnects urban landscapes to their natural hydrologic functioning, making it an effective watershed management tool [23]. GI implements, such as bioretention facilities and pervious pavements, are increasingly popular structural BMPs that utilize natural ecologic functioning to manage stormwater by reducing runoff volumes and removing water quality contaminants [28, 31]. Society places great value on urban green and blue spaces, as their aesthetics increase human physical and emotional health, mitigate urban heat island effects, and provide ecological benefits [29]; however, these benefits are not easily valuated, making the implementation process more challenging.

Worldwide, thousands of lakes, wetlands, and estuaries are eutrophic as a result urbanization [20]. One such example is Frisco Lake in Rolla, Missouri, a Midwestern town of nearly 20,000 people. Frisco Lake is a small, shallow man-made impoundment, 0.02 km^2 in size with an average depth of 1.5 m, residing within a community park surrounded by urban residential areas. The lake was built in the 1860s by the Frisco Railroad to be used as a reservoir for watering stream locomotives. In 1982, the city of Rolla partially drained and excavated a portion of deposited sediment of the lake after severe flooding occurred [10]. The lake is currently listed in the EPAs 303(d) Impaired Waters for containing elevated concentrations of mercury attributed to inputs from atmospheric deposition [19]. The lake experiences seasonal algal blooms, including a fish kill in 2014, ultimately reducing its aesthetic and recreational functioning. There are ongoing plans by the city to dredge the lake to reduce the algal blooms. However, there is need for a long-term mitigation strategy.

In this article, we provide (1) a sub-catchment scale field monitoring study to assess and model urban stormwater nutrient loads into a eutrophic freshwater lake, providing a basis for (2) stormwater management recommendations incorporating GI implements that will provide long-term lake water quality benefits.

2. Methods and Materials

2.1 Study Area

Frisco Lake is located within the Ozark Plateau physiographic region. This drainage area is located within the larger Dry Fork sub-basin, part of the Meramec watershed in which waters eventually drain into the Mississippi River [3]. The study area is located in a humid continental zone characterized by cool to cold winters and long, hot summers. The total annual precipitation is approximately 107 cm with two thirds of the rain occurring in April through September. The average seasonal snowfall is about 43 cm [30].

Contributing catchment areas totaling 0.31 km^2 drain into Frisco Lake and were categorized into land use categories: urban residential, institutional, and commercial/industrial. Geospatial analyses and mapping were completed using ArcMap 10 software [14]. Catchment A was 0.22 km^2 in size comprised of 80% residential, 8% institutional, and 12% commercial land use areas, with 56% having impervious cover. Catchment B was 0.16 km^2 in area, comprised of 41% residential and 59% intuitional land use area of which 59% is impervious. Study area catchments and land-use characteristics are summarized in Fig. 1 and Table 1.

Table 1 Geospatial land-use characterization of Catchments A and B.

Land-Use Type	Catchment A		Catchment B	
	Area (km^2)	% of Basin	Area (km^2)	% of Basin
Impervious Surface	0.123	56	0.093	58
Residential	0.176	80	0.066	41
Commercial	0.027	12	0	0
Institutional	0.017	8	0.094	59
Total	0.22	100	0.16	100

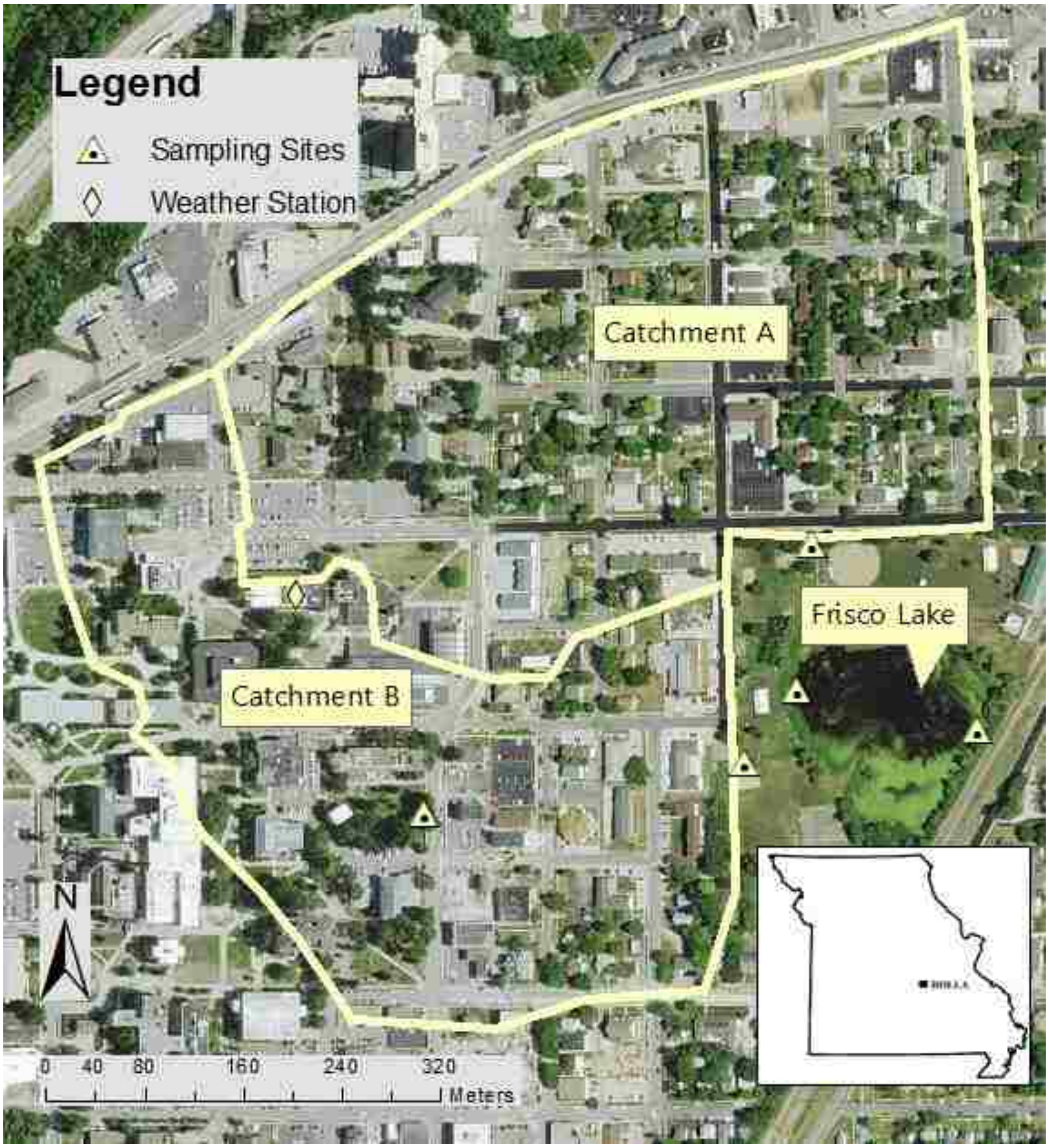

Fig. 1 Contributing catchments to Frisco Lake in Rolla, MO, with project water quality sampling locations marked.

2.2 Data Collection

Four sampling sites were monitored along both inlet and single outlet stormflow paths through Frisco Lake as shown in Fig. 2. Three sampling sites were located at stormwater drainage outfalls of Catchments A and B and of a portion of campus located in Catchment B. The catchment outfalls at channel inlets to Frisco Lake that only discharged during wet-weather events. Two sampling sites were representative of lake water quality and located at the lake's western bank and emergency spillway effluent. Stormflow grab sampling using conventional techniques [18] began in September of 2014 and continued until June 2015. A total of 17 rain events were sampled and during four separate storm events more detailed hydrographic sampling, 5 to 11 samples per event, was conducted at Channels A and B. A total of 101 samples were analyzed in the nearby Environmental Research Center laboratory for Total Phosphorus (TP) using a Hach DR/2400 Spectrophotometer® following EPA procedure 365.2 for freshwater samples and Total Nitrogen (TN) was tested using a Shimadzu TOC-L analyzer using a 720°C catalytic thermal decomposition/chemiluminescence method. Mean TP concentrations from each sampling site are presented in Table 2 in Section 3.1.

Fig. 2 Project study area with lake influents and effluent and streamflow monitoring equipment locations denoted.

2.3 Hydrologic Monitoring

A year's worth of streamflow data was collected beginning in September 2014 using four continuous recording gauges within the channel inlets and lake positioned such that each inlet had an upstream and downstream gauge that measured the water elevation drop between them, as depicted in Fig. 2. To gauge streamflow, USGS staff plates and perforated metal monitoring wells that housed submerged Levelogger sensors were affixed to the channel sides. A nearby Barologger sensor recorded site atmospheric pressure to compensate the stage data from the submerged Leveloggers. To capture hydrograph peaks within the narrow (approx. 2 m wide and 1 m tall) channels with adequate detail, the Levelogger sensors recorded stage levels at 15 second intervals. The compensated the stage data from the Leveloggers showed erroneous diurnal fluctuations (0-2 cm) during periods where stage level was constant, caused by the instrument's failure to compensate for ambient temperature changes creating a temperature effect [17]. Though slight, these errors were corrected by manually adjusting the stage levels within each data record to observed or known stages to reduce distortion of the discharge calculation.

Field streamflow measurements used conventional current meter techniques [6]; however, modifications to the method were necessitated due to the rapid rise and fall of stage within the channels in which the number of measurements taken along the channel profile were reduced. Rather than following the conventional technique where cross-sections are broken into at least 20 sub-sections, the cross-sectional area of Channel A was broken into three sub-sections and Channel B used a single sub-section, inducing error in discharge calculation. Each channel experienced backwater effects due to the rising water surface elevation from the lake during wet-weather events. This more complicated hydraulic situation required the use of the stage-fall rating curve technique [24] for each channel and the use of the following adjustment equation.

Eqn. 1. $$Qadj = Qr\sqrt{\frac{Fm}{Fr}}$$

where $Qadj$ was backwater corrected discharge, Q_r was the stage-discharge rating, F_m was the observed water elevation fall between up- and downstream gauges, and F_r was the stage-fall rating. Rating curves were fit to collected data so that nearly all adjusted discharges fell within 10% error of observed discharges for quality assurance. During the infrequent occurrence where auxiliary stage data was not collected, specifically at Channel B during October and November 2014, Q_r instead of Q_{adj} was used to estimate discharges, overestimating runoff volumes during monitored rain events within those months. The rating curves used for both channels are presented in Fig. 3 in Section 3.1.

Rainfall-runoff (RF-RO) hydrograph analyses using standard methods [2] were performed to give a final quality assurance to the flow estimations. Most RF-RO ratios for storm events fell within expected ranges (0.5 to 0.8), though runoff volumes from Channel B were overestimated (ratios >1) during infrequent periods when backwater adjusted discharge information was unavailable. The precipitation data was taken from a Missouri S&T weather station located within Catchment B and approximately 0.4 km from the monitoring sites. The discharge information was compiled into an annual time series, with the winter record omitted due to limited instrument recording capabilities during freezing temperatures.

2.4 Nutrient Loading Analysis

Mean-annual estimates of TP-flux were estimated at each channel using the wet-weather discharge time series and collected water quality data using linear regression techniques [9]. Each channel satisfied minimum data requirements with sample sizes of 40 and 32 that were taken throughout the year. The simplistic linear regression models are known to have statistical biases that underestimate load calculations. Therefore, a more detailed event loading model using calculation methods [22] was conducted on four individual storms, three at Channel A and one at Channel B. TP loads of these four events were calculated using each nutrient concentration model and determined the average percent change in load increase between the loading models. When estimating the annual nutrient load and catchment yields, the loads from each channel were calculated using the linear regression model and then adjusted by the mean percent change. The regression models used for each channel and an event concentration curve model are presented in Fig. 4, and the event load comparisons between nutrient loading models are presented in Table 3 in Section 3.1.

2.5 Lake Eutrophication Modeling

A mass balance model was used to determine TP concentrations of Frisco Lake and using Eqn. 2 from Davis and Masten (2009).

Eqn. 2 $$V\frac{dC}{dt} = QC_{in} - QC - kCV$$

where V is lake volume (m^3) determined by multiplying the average depth (m) and surface area (m^2) of Frisco Lake; Q is the sum of flows at Channel A and B, the mean-annual flow into the lake (m^3/s); C_{in} and C are TP influent and effluent concentrations (mg/L), and a reaction coefficient represented by settling rate k (s^{-1}). The model was assumed to be at steady state and used to estimate lake TP concentration and resulting lake quality, as lakes with TP concentrations below 0.015 mg/L will avoid eutrophic conditions [12], thus achieving a target concentration of 0.015 mg/L is the goal.

The observed mean-annual flows and TP concentrations at each channel from the monitoring analysis were used for the Q and C_{in} parameters. The lake volume of 30,350 m^3 was calculated by multiplying the surface area of the lake, determined by a geospatial analyzer ArcMap 10 [14], and the average depth, based upon historical knowledge and observation. The estimated observed nutrient loads and discharges from the prior analyses were used to determine the mass rates. The settling rate k was estimated by using the Frisco Lake mass balance model to solve for k on three different storms events with measured influent and effluent mass rates. The most conservative k value from the three analyses was 0.00001 s^{-1} which was comparable to literature values [12] and therefore, used in the model analyses. A sensitivity analysis was conducted using model input parameters reflecting realistic variances in values, for example the upper and lower values of modeled k values, depth of Frisco Lake, and mean-annual P fluxes, the results of which are presented in Section 3.5.

A conceptual model diagraming the data inputs and modeling process used during this study to estimate the parameters required to run the Frisco Lake mass balance model is shown in Fig. 3.

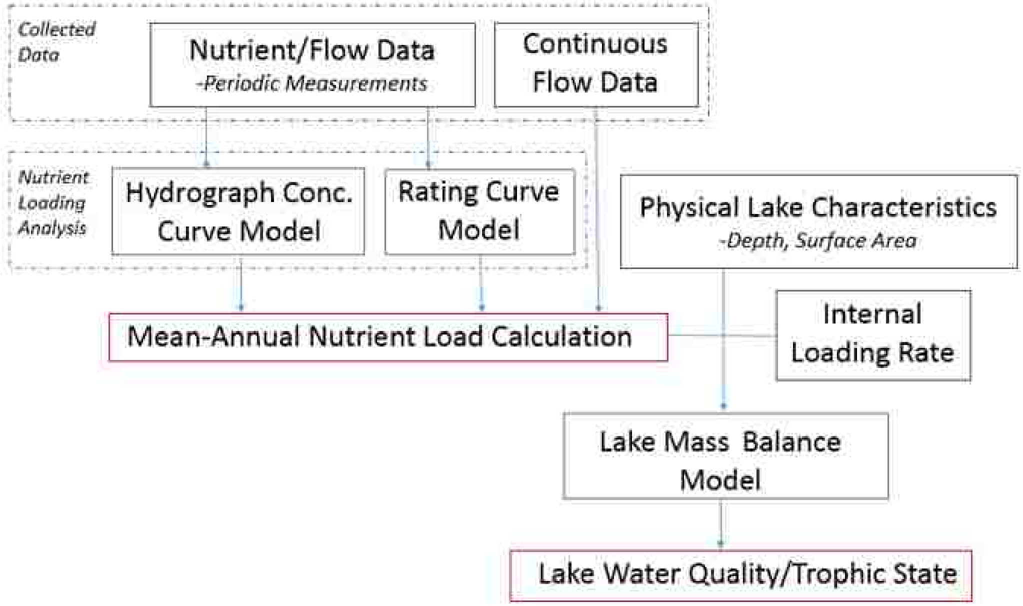

Fig. 3 Conceptual diagram showing collected data and modeling processes used to estimate Frisco Lake water quality.

2.6 Stormwater Quality Modeling

Recommendations for BMPs within the urban watershed, including an upstream GI plan with bioretention and pervious pavement implements, were developed after assessing the stormwater monitoring information and used to model potential lake trophic state improvement. Nutrient loading improvements from pervious pavement installations involved a catchment specific approach that modeled TP loading using the standard Simple Method [25] technique. Lake water quality improvements from the proposed bioretention facilities were modeled using estimated catchment yield reductions based on stormwater and nutrient removal rates from recent efficiency studies and urban watershed planning models [20].

Infiltrating stormwater leaches nutrients from the media of bioretention facilities, producing typical effluent TP concentrations of 0.05 to 0.18 mg/L [11] regardless of influent TP concentrations being greater or less than that amount. Therefore, removal efficiencies of bioretention facilities vary greatly depending on the influent concentration. For this reason, to determine the influent concentration the expected decreases were modeled by reducing both catchment observed mean-annual TP concentrations to 0.18 mg/L at bioretention effluents.

A similar method was used to estimate the reduction in stormwater quantity, as properly designed and maintained bioretention facilities reduce annual runoff volumes by 40 to 90% [11]. Modelled retention rates for each catchment are discussed in Section 3.4. Mean-annual stormwater discharges and TP concentration reductions for each catchment were used to determine improved mass rates and are presented in the results section.

The modeled mass rate reductions from each catchment after proposed GI modifications were then applied to the Frisco Lake balance model, with adjusted physical parameters to account for the proposed lake dredging to project lake trophic state post GI implementation. The results are presented in the following section.

3. Results and Discussion

3.1 Stormwater Monitoring

The TP and TN water quality data collected at each sampling site were categorized into hydrographic position and then averaged to estimate influent stormwater quality for GI design. The results of the TP water quality analysis are presented in Table 2. Additionally, the TN data was analyzed and compared to the TP in order to reassure that P was the limiting nutrient causing the eutrophication of Frisco Lake. The greatest N:P ratio observed in an assessment all water quality samples was 5:1, which is less than the approximately 15:1 ratio required for TN to be the limiting nutrient.

Table 2 TP concentrations at each sampling site presented as mean +/- standard deviation.

	Mean TP Concentration (mg/L)			
	Channel		Effluent	
	A	B	Campus	Lake
First Flush	1.07 +/- 0.87	0.94 +/- 0.48	1.40 +/- 0.31	N/A
Peak Flow	0.90 +/- 0.47	0.88 +/- 0.63	0.55 +/- 0.27	N/A
Receding Flow	0.52 +/- 0.29	0.50 +/- 0.37	N/A	0.18 +/- 0.11

The complex rating curves used to determine the annual discharge records of Channels A and B are presented in Fig. 4.

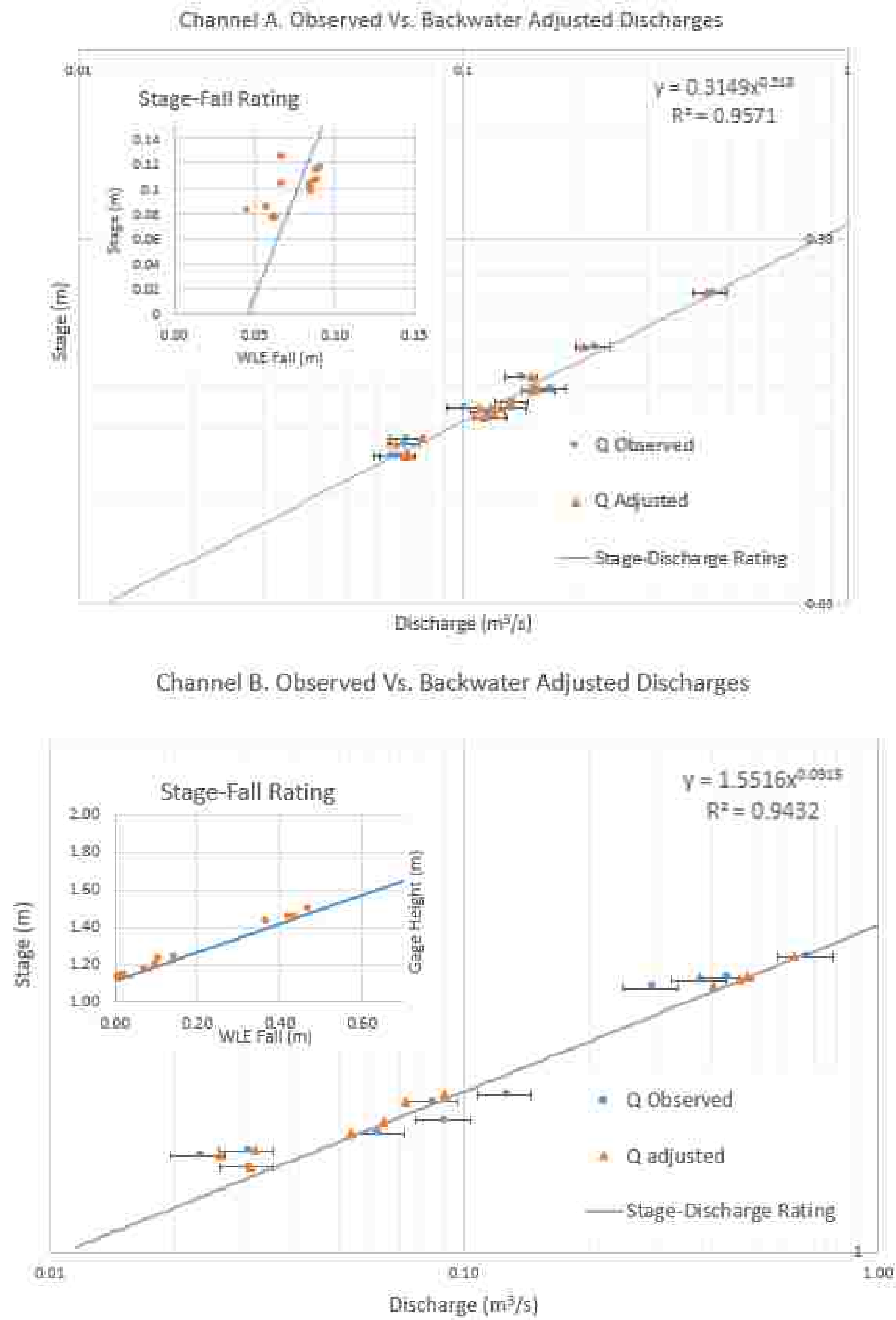

Fig. 4 Stage-Discharge rating curves are plotted with backwater-corrected and observed discharges for Channel A (top) and Channel B (bottom). Stage-Fall rating curves and Stage-Discharge rating equations used in the backwater corrected discharge calculation are pictured at the upper left and right of charts.

The linear regression and concentration curve nutrient loading models used to estimate annual TP loading into Frisco Lake are presented in Figure 5, and the event load comparisons between nutrient loading models are presented in Table 3.

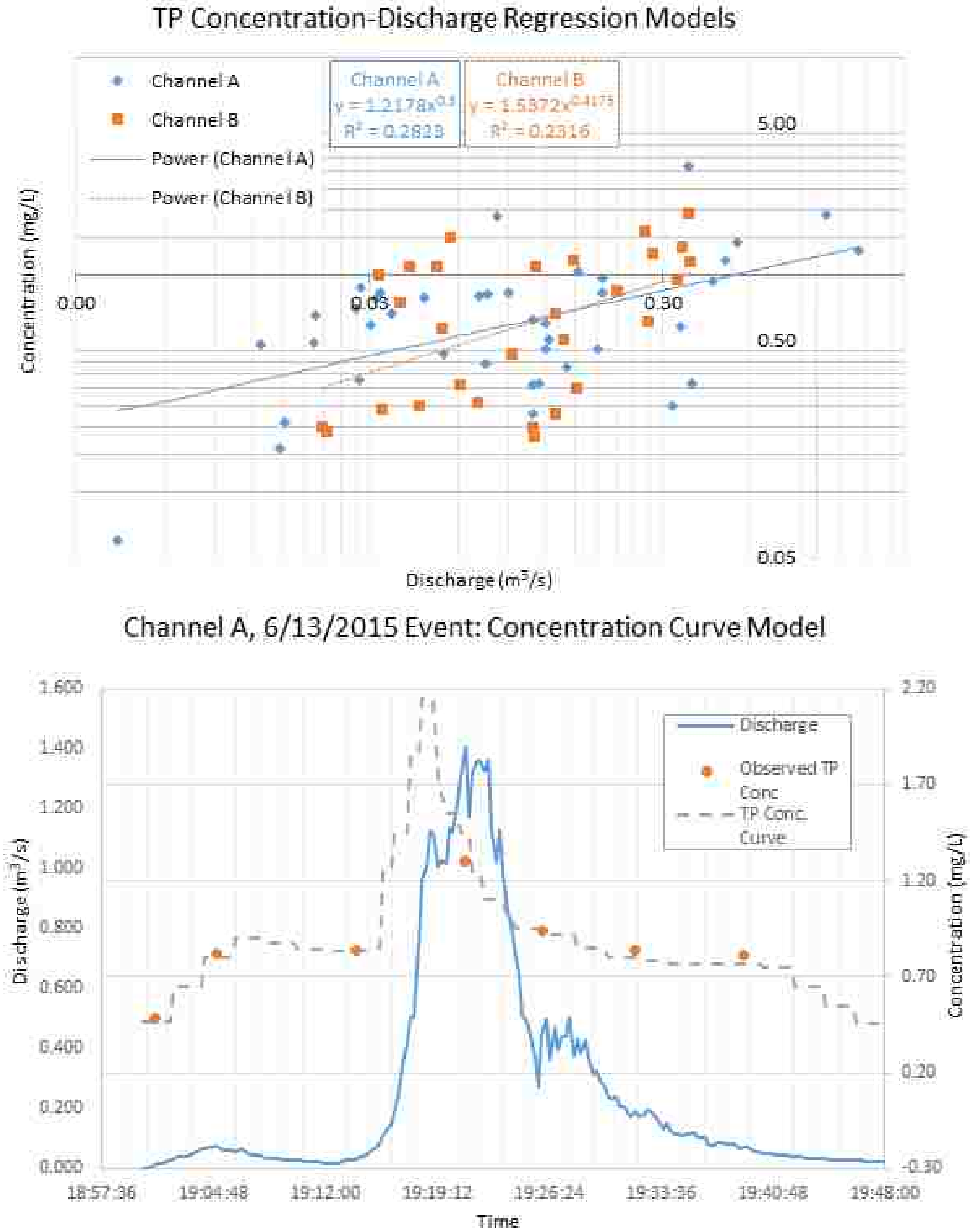

Fig. 5 Top: Observed TP concentration plotted against instantaneous discharge and resulting regressions used in nutrient loading analysis at each channel. Bottom: An event concentration curve analysis at Channel A used to determine correction factor in mean-annual load estimate.

The estimated annual TP loads, concentrations, yields, and flow rates from each catchment are presented in Table 4. The catchment yields for each channel are both higher than typical urban land-use rates from the literature. Typical land area yields that are used for non-point source TP pollution coefficients in watershed models employed by the USGS can range over five orders of magnitude from 0.001 to 7.2 kg/ha/yr, with accepted urban land use rates at the higher end of the spectrum [34]. This increase can be attributed to the fact that the measured water quality averages at each sampling location were higher than national averages [20]. Additionally, average annual precipitation in Rolla is recorded at 107 cm [30], and the yearly cumulative rainfall during this study interval was 97 cm [33], suggesting that during a typical, slightly wetter year catchment yield rates may be proportionately higher.

Table 3 Event nutrient loading model comparison between each channel.

	Event Loads (kg)			
	Channel A			Channel B
Model	4/9/2015	6/13/2015	6/12/2015	4/9/2015
Conc. Curve	0.47	0.92	0.18	0.51
Regression	0.36	0.8	0.16	0.28
% Change	31	15	13	82
Mean % Change +/- Std Dev		35 +/- 28		

Limited time and resource constraints during monitoring reduced the accuracy of the catchment yield and loading determination; however, study values are useful as best-estimates to guide watershed planning strategies. Error in the final load and yield calculations was determined by assessing expected percent errors associated with the methods and catchment properties from similar water quality monitoring studies. Small catchment sizes of <1 km^2 increase the relative error induced as increasing accuracy is required when measuring lessor volumes. In 3-yr, hydrological accuracy runoff studies [4], the total error of the calculated discharges for catchments <1 km^2 were 7-20%. The Frisco Lake continuous stream discharge monitoring study had high temporal resolution and included a backwater correction, though lacked accuracy in Channel B's discharge calculation. The regression models fit to the water quality data at each channel had relatively small R squared values, suggesting variability in the annual loading calculation. The additional nutrient loading analysis using detailed hydrographic sampling (presented in Fig. 5) aimed to reduce some of this error by calculating and comparing loads on an event by event basis. However, the loads calculated between the two methods varied and data sets with hydrographic sampling were limited. Thus, the mean-annual flowrates and concentration values used to conduct the sensitivity analysis of the lake mass balance model were varied by +/- 20% to account for potential measurement error.

Table 4 Annual TP flux information at each channel based on monitoring results.

		Annual		Mean-Annual	
	Catchment Area (ha)	Load (kg)	Yield (kg/ha/yr)	Conc. (mg/L)	Flow Rate (m^3/s)
Channel A	22	380	17	0.42	0.044
Channel B	16	560	35	0.46	0.054
Total	38	940			0.098

3.2 Watershed Improvement Plan

In order to reduce the algal blooms and restore water quality of Frisco Lake, both external and internal P sources must be reduced. Shallow lakes with significant sedimentation often internally release P into the water column from the sediment during summers in which conditions, such as increased biological activity and temperature and decreased oxygen content, promote P release [26]. Internal P recycling from the sediment within the lake bed is expected to continue elevating P levels, creating nuisance algal blooms regardless if external P loads are decreased. Therefore, the City of Rolla has plans to dredge Frisco Lake to remove the direct, detrimental internal P source. However, without additionally reducing the external sediment and P loads that would gradually refill the lake, this will serve as a temporary solution. Lake dredging is an invasive and costly procedure, making solutions that reduce future dredging economically viable and environmentally beneficial.

Long-term lake water quality improvement can be achieved by implementing upstream stormwater BMPs throughout the watershed. The Frisco Lake watershed could benefit from practical housekeeping strategies by the city including efforts to encourage the disconnection of roof drain downspouts, provide free rain barrels to interested citizens, increase public outreach highlighting proper lawn and pet waste care, or by passing a low or no P fertilizer ordinance in areas known to have naturally elevated levels of P. GI must be implemented at a watershed-scale to improve the area's disrupted hydrology [23].

3.3 Watershed GI Plan

The proposed GI plan is presented in Figure 5 and includes three bioretention facilities located at the project stormwater quality sampling locations at the outfalls of Catchments A and B, and at the campus within Catchment B. Redundancy and dispersal of facilities throughout the drainage area increases stormwater improvement capabilities [31], and were therefore utilized in this plan. For Catchment B, one upstream bioretention facility is located at a campus drainage outfall in a low-lying, open space detention area in between the Physics and Inter-Disciplinary Engineering buildings on the S&T campus. Retrofitting a bioretention facility in this location was considered feasible as the area is already serves as a stormwater detention area when flows exceed infrastructure stormwater conveyance capacity. The other areas chosen for the placement of a bioretention facilities are located at Catchment A and B outfalls at the heads of Channels A and B. The noted areas are currently open, widely unutilized, and unaesthetically pleasing as a drainage channel in a city park. These areas have the potential to benefit, aesthetically, ecologically, and functionally by retrofitting a green infrastructure implement. The most downstream portions of the proposed bioretention facilities at the lake inlet channels lie within the lake boundaries and are typically permanently inundated with water, posing potential for a constructed wetland. The specific design and construction of the GI implements are beyond the scope of this project.

For the proposed GI plan, impervious parking lot and low traffic areas within the watershed were delineated and denoted as potential "Pervious Pavement" in Figure 7. Catchment B is comprised of 17% parking lot, whereas Catchment A is comprised of

19%. Though the stormwater quality modeling in Section 3.4 only considered the improvements using bioretention, applying the Simple Method [25] to estimate TP load reductions by converting impervious parking lots to pervious areas decreased the annual catchment yields by 30%. The parking lots are located in institutional, commercial, and residential areas, and the feasibility and level of implementation would be dependent upon community-wide interest. There are many varieties of pervious pavements, and all must be correctly designed and maintained to ensure proper functioning.

3.4 Stormwater Quality Modeling

Mean observed first flush and peak flow stormwater TP concentrations at the campus and Catchment A and B outfalls were high, as shown in Table 2, making the sites viable candidates for bioretention mitigation, as the concentrations could be expected to be mitigated to 0.18 mg/L or less after infiltration. A schematic diagramming expected stormwater quality and quantity improvements at the campus outfall is shown in Figure 6.

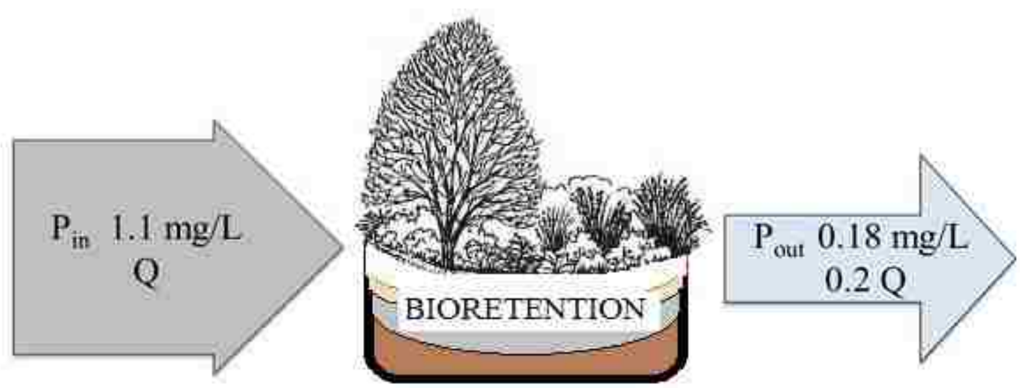

Fig. 6 Schematic showing influent and effluent stormwater characteristics at proposed upstream bioretention facility at the campus outfall.

Stormwater quality improvements from the proposed GI plan were modeled using expected stormwater and nutrient removal efficiencies from bioretention. As there are multiple areas available for biorention implementation in Catchment B, a greater stormwater retention of 80% was applied to the observed mean-annual flow rate and a more conservative rate of 20% was applied to the flow at Catchment A. Greater stormwater retention efficiencies are expected by implementing bioretention facilities throughout watersheds creating a treatment train effect, as the redundancy will mitigate the peak runoff flows that cause the reductions in stormwater retention and subsequent decreases in removal efficiency [31]. The rates used in stormwater quality modeling are listed in Table 6 in Section 3.5.

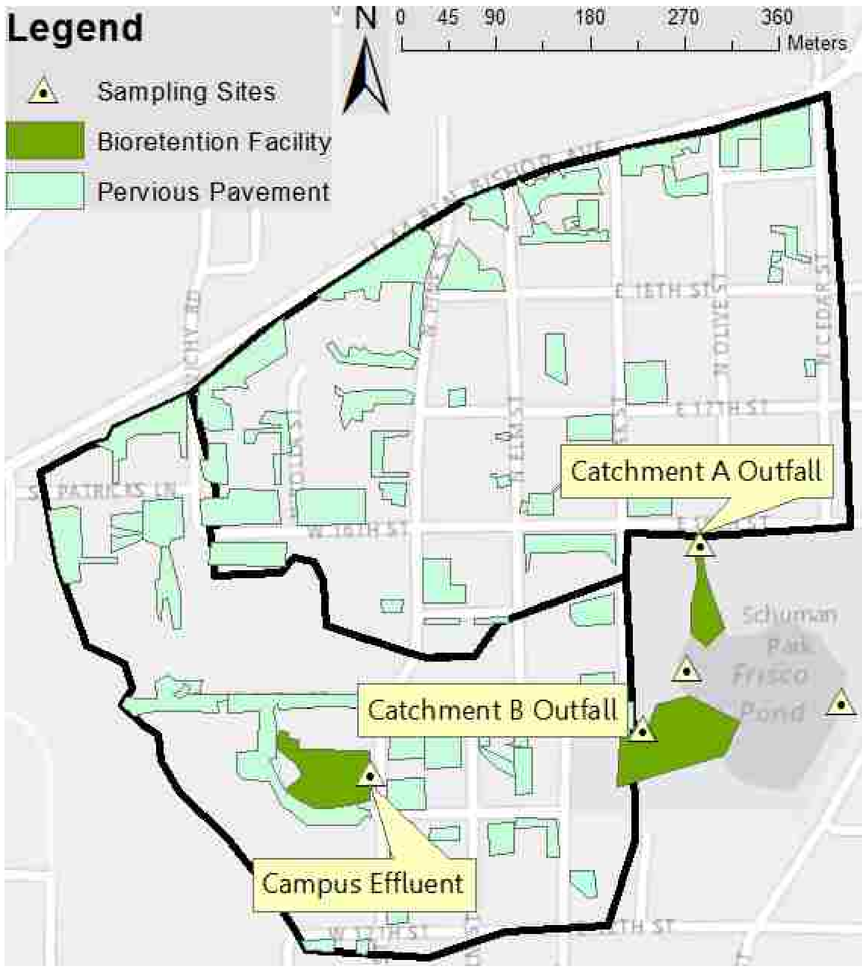

Fig. 7 Watershed improvement plan with proposed bioretention facilities and pervious pavement locations denoted.

3.5 Frisco Lake Mass Balance Modeling

The estimated catchment P fluxes and physical and chemical lake properties determined from stormwater monitoring were inputted into a current state Frisco Lake mass balance model as presented in Table 5. The output lake P concentration of 0.16 mg/L was representative (within 12% error) of the observed mean lake P from the water quality analysis. The Frisco Lake mass balance modelled stormwater improvements from the proposed GI plan using adjustments, listed in Table 6, from the observed monitoring results and the increase in mean lake depth to 3 m after dredging occurs. The post implemented GI plan model predicted lake P concentrations of 0.01 mg/L, a value that would put the lake within the upper oligotrophic range and at a healthy water quality [12]. The current and post-GI plan Frisco Lake model input and output parameters are presented in Table 5.

Table 5 Frisco Lake mass balance model comparing predicting P concentrations at current and post GI implementation.

Model Inputs: Current State					Post Implemented GI Plan		
		Channel			Channel		
Observed Mean-Annual Value		A	B	Total	A	B	Total
Mass Rate (g/s)	QC_{in}	0.038	0.044	0.082	0.006	0.002	0.008
Discharge (m^3/s)	Q	0.044	0.054	0.098	0.035	0.011	0.046
		Frisco Lake			Frisco Lake		
Volume (m^3)	V	30350			60,700		
Area (m^2)	A	20235			20,200		
Depth (m)	d	1.5			3.0		
Settling Rate (s^{-1})	k	1.3E-05			1.3E-05		

Model Outputs: Current State			Post Implemented GI Plan		
Lake Conc. (mg/L)	C	0.16	Lake Conc. (mg/L)	C	0.01

A sensitivity analysis was performed on each input parameter of the mass balance model to test for model accuracy and error. The percentage varied and resulting upper and lower limits analyzed were based upon realistic margins of error in expected from data collection and determination. The model was most sensitive to lake depth (and resulting volume) variance, suggesting that model accuracy can be refined by completing a precise lake bottom depth survey. The minimum and maximum output values were calculated by choosing the most and least conservative conditions, using either all of the upper or lower limits for each measured input parameter. Outputs for the maximum and minimum lake P concentration to be expected are 0.09 and 0.30 mg/L, values within the range of observed lake water quality monitoring.

Table 6 Adjustments to current state mass balance model input parameters used in post-GI implemented plan model.

Input Parameter[1]		Adjustment
Influent Conc. (mg/L)	Cin	Channel A decreased from 0.42 to 0.18 mg/L Channel B decreased from 0.43 to 0.18 mg/L
Discharge (m^3/s)	Q	Channel A reduced by 20% Channel B reduced by 80%
Depth (m)	d	Increased to 3 m

[1]all parameters are mean-annual values

.

Table 7 Results of Frisco Lake mass balance model input parameter sensitivity analysis.

Min. and Max. Input Values			
Parameter	% Varied[1]	Lower	Upper
Mass Rate (g/s)	20	0.065	0.097
Discharge (m^3/s)	20	0.078	0.118
Depth (m)	40	2.1	1.0
Settling Rate (s^{-1})	20	1.56E-05	1.04E-04

Min. and Max Output Values[2]			
		Lower	Upper
Lake Conc. (mg/L)	C	0.09	0.30

[1]Percent varied based on estimate of realistic measurement error

[2]Used lower and upper limits for all input parameters

The Frisco Lake watershed monitoring assessment results were successfully used to model and plan upstream watershed management implements. Simple land-use modeling of the proposed GI plan indicated that long-term lake water quality improvement is possible; however, the performance of GI to achieve the expected modifications to stormwater quality will depend upon the correct design and implementation of the structures, as the values used in this analysis were based upon previous, similar studies. Mass balance models provide the capabilities to test multiple scenarios using varying GI performance efficiencies, making it a useful planning tool for urban watershed management.

Additionally, the restoration of Frisco Lake, like many urban impaired water bodies, will require willingness and organization by both citizens and local government to make the necessary watershed-scale improvements. With city-wide action to dredge Frisco Lake, construct and maintain properly designed bioretention facilities and pervious pavements throughout the watershed, and promote better stormwater practices through public awareness or ordinances, long-term lake and watershed restoration is expected.

3.6 Watershed Benefits of GI Implementation

In addition to the mitigation of urban stormwater pollution to downstream environments, implemented GI within the study watershed would provide ecologic, economic, and social benefits to the city, university, and citizens. Educational value to the university, specializing in science and technology, could be provided from research opportunities to study GI effectiveness, as the watershed monitoring information can be used as baseline data and compared against water qualities post implementation. Financial benefits would be realized by the city through the reduction of costs induced by stormwater handling and control, such as flood damage, stream and lake restoration projects, and gray infrastructure maintenance. GI provides increased ecosystem services in urban environments through climate regulation, habitat re-establishment, and air quality improvements. Social benefits from the improved aesthetics of GI would be provided to Rolla citizens by increasing the community livability and cultivating public connectedness to the community environment. Green spaces promote human mental and physical health by creating more desirable living spaces. The benefits from GI implementation in this particular community are not limited to this watershed; urban watersheds all across the globe would see similar increases in hydrological, ecological, and social health.

4. Conclusions

Successful restoration of nationwide urban impaired water bodies calls for rethinking and redesigning urban watersheds to mimic pre-development hydrologies of which is accomplished by assessing stormwater characteristics and downstream. Useful, widely-applicable watershed monitoring and planning methods were provided in this study. Additionally, multi-level cooperation between citizens, local government, and state and federal agencies is required to make watershed improvements, Local governments are responsible for the implementation of capital projects, such as lake restoration via sediment dredging or through the design and construction of GI implements; large-scale public awareness campaigns, such as impacts of excess fertilizer use and improper pet waste disposal, and the passing of ordinances controlling water quality pollutant sources, such as restricting use of P containing fertilizers and fining pet owners who do not pick up waste. Citizens also play a strong role by remaining involved and interested in restoring surface waters and by reducing their watershed footprint, such as responsibly fertilizing, properly disposing pet and yard waste, disconnecting downspout connections, setting up rain barrels, and building rain gardens. Federal agencies that are responsible for monitoring, characterizing, and assessing water quality can work to improve data collection methods and modeling tools to provide more widely applicable and available information to be used in planning and learning.

Many cities nationwide are often unwilling to implement BMPs as there are uncertainties in BMP performance and cost, insufficient guidelines and engineering standards, and a lack of funding and economic incentives and legislative mandates. The economic, social, and environmental benefits associated with the implementation of GI and other stormwater BMPs are not often realized. Ultimately, efforts to assess and research the improvements and outcomes of implemented GI on urban watersheds are needed to valuate the true, unrealized benefits of GI and push citizen and government engagement toward building healthy, sustainable watersheds.

Acknowledgments

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SECTION

4. SUMMARY AND CONCLUSIONS

The over-arching goal of this study was to understand the hydrology of an impaired urban watershed in order to provide an effective improvement plan. This goal was achieved by completing of each of the objectives. The first objective of designing and conducting a stormwater quality monitoring plan encountered more difficulties than initially assumed, resulting in collected data that was most useful as best-estimate information in further parts of the project. Hydrologic monitoring conditions were less than ideal: the channels were affected by backwater conditions, the contributing catchment sizes were small (< 1 km^2) that increased the need for more precise measurements, the flashiness of each channel during rain events with moderate or greater intensities required extremely rapid windows of useful sampling periods, the fineness of stream depth data required was beyond the streamflow gauging instruments could provide due to temperature effects induced within the recording device. Additionally, the water quality data collection plan was resource and time limited, leading to unreliability in the water quality trends used in nutrient loading analysis. However, the collected data, was useful in providing previously unavailable baseline watershed stormwater quality information and therefore supported the initial hypothesis.

The second objective of modeling annual discharges and P loads into Frisco Lake in order to estimate the P reduction required for water quality improvement using a mass balance model was also accomplished. As previously mentioned, the loading rates were not easily modeled due to the variability and potential inaccuracy in the streamflow and water quality monitoring record; however, the observed measurements were useful in approximating real life water quality conditions in mass balance modeling. Additional refinements to the lake nutrient loading model, such as to the P reaction rates during different seasons and geochemical conditions, would increase its accuracy, predictive capabilities, and usefulness. However, the second hypothesis was also supported as modeled lake TP concentration estimations matched observed, elevated values.

The final objective of assessing watershed information to plan GI implements and stormwater management strategies for lake water quality improvement was also completed. Simplistic modeling techniques and stormwater improvements from GI were based upon literature values were used to model expected post-GI plan outcomes. Only through actual implementation of the proposed GI plan can the impacts be studied and lake water quality improvement verified. The hypothesis that implemented upstream GI could be used to improve Frisco Lake water quality was supported by the watershed modeling techniques used in this study.

Though this study focused upon a single urban watershed in central Missouri, its overall method to holistic watershed improvement, including formulating a site-specific monitoring scheme, modeling nutrient fluxes and downstream impacts, and selecting and planning upstream BMPs for most effective improvement can be applied in any urban watershed. To protect downstream aquatic environments in this urbanizing world, changes must be made to current watershed management practices. Emphasis should be placed upon reducing and treating stormwater at its upstream source by systemically recreating pre-development hydrologies and pollution generation patterns throughout urban watersheds. GI provides a means to achieve pre-development hydrologies; however, without economic incentives or social pressure driving the need for increased implementation, it is unlikely that the measures required to improve surface water quality will be taken. As promising as GI, and other practical housekeeping BMPs that reduce non-point pollutant source generation remain, their true efficiencies and stormwater mitigation capabilities are in need of greater understanding that can only be provided by researching, understanding, and performing watershed management projects such as this study completed on Frisco Lake.

5. RECOMMENDATIONS FOR FUTURE STUDY

A continuation of this particular study that aimed to understand an urban watershed in Rolla, MO would benefit from a refinement of the continuous hydrological monitoring of stormflows into Frisco Lake, particularly by conducting more field monitoring at Channel B to improve the rating curve, a more detailed and extensive water quality plan with automated equipment and tools to improve nutrient modeling accuracy, additional sampling and assessment of the geochemical constituents of the lake sediments and surrounding soils to better quantify and understand the internal interactions and nutrient cycling within the lake to improve the mass balance modeling, a lake bottom survey to more accurately determine lake volume, and finally additional analysis of BMP implements to the watershed that include more varieties of GI, such as green roofs, and a more in depth look at site-specific potential stormwater and nutrient removal impacts on stormwater.

On a broader scale, a research endeavors that would greatly benefit the future of watershed improvement projects, similar to this study, include developing user-friendly environmental and water quality data collection tools that would make monitoring simpler, more ubiquitous, and more cost effective. Currently, environmental and hydrologic monitoring tools are designed for large-scale watershed monitoring, making them less refined and less capable of providing useful information for sub-catchment, smaller site-specific planning. A collaboration on this campus is currently working to improve both environmental data collection capability and availability by developing cellular phone applications capable of allowing citizen scientists to monitor streamflows with substantial accuracy. Involving citizens in the data collection process promotes societal interest and motivation toward understanding and improving the nation’s surface water quality.

In addition, further research is needed on the effectiveness and impacts associated with implemented BMPs and GI as they are relatively new developments. It is clear that urbanization is responsible for much of the impairment of watersheds and BMPs such as GI implements can offset these negative effects. However, before communities, citizens, and governments will drive changes, the true value of GI must be assessed. Valuation must include non-monetized benefits realized beyond stormwater mitigation, such as social, health, and climate impacts. Green spaces have been shown to be critical in maintaining human health just as much as ecological health (Tzoulas *et al.*, 2007). This project succeeded in providing Frisco Lake watershed planning improvements; however, actual implementation of the GI plan and further evaluation on its impacts on lake water quality and citizen quality of life would be valuable. Case studies that quantify and report the positive impacts and environmental, social, and economic benefits of GI are necessary before any real change or reform to current watershed practices can be expected.

APPENDIX A

WATER QUALITY DATA

Water Quality Data Collection Methods

Sampling Scheme. Four sampling sites were chosen along the inlet and outlet flow paths through Frisco Lake. Sampling sites A and B were located at concrete channel and natural channel inlets to the lake where the upstream base gauges were located, as shown in Figure A1. Channels experienced flow during only runoff producing wet-weather events. Sampling sites C and E, pictured as yellow 'X's in Figure A1, were located at the western edge of Frisco Lake and at the lake effluent and were representative of lake water quality. Sampling sites are presented in Figure A1. An additional sampling site was located upstream from Channel B at a campus outfall where campus runoff effluent concentration data was collected.

Figure A1. Study site with water quality sampling and upstream, downstream, and barometric pressure gauge locations denoted.

Stormflow sampling began in September of 2014 and continued until June 2015. A total of 17 rain events were sampled during the sampling period. During four separate storm events at sampling sites A and B, hydrographic sampling was conducted using conventional grab sampling techniques in which multiple samples were collected along the storm event hydrograph. Samples per storm event ranged from four to fifteen depending on the intensity and duration of the storm event. A summary of number of water quality samples collected at each site is shown in Table A1.

Table A1. Summary of water quality samples collected

Sampling Site	Number Collected
Channel A	42
Channel B	30
Lake Effluent	13
Campus Effluent	16
Total	101

Grab Sampling. Grab samples were collected by one Missouri S&T graduate student. During periodic rain events, plastic, dishwasher-cleaned 125 mL Nalgene® bottles were submerged just below the water surface for shallower flows, or roughly one third of the stream depth for deeper flows, and in the center (i.e. thalweg) of each channel at each of the aforementioned sampling sites. Each bottle was labeled with a number, and data was compiled in a field notebook. Immediately after collection, the samples were transported by foot or vehicle to a nearby lab in the Environmental Research Center in the Missouri S&T campus where they were refrigerated until lab analysis. Samples were tested before exceeding their maximum holding times of 26 days. This procedure was based upon the USGS Interagency Field Manual for the Collection of Water Quality Data (Lurry & Kolbe, 2000).

Lab Analysis. Stormflow samples were tested in the Environmental Research Center laboratory for total nitrogen and total phosphorus. Total Nitrogen (TN) concentrations were tested using a Shimadzu TOC-L TOC Analyzer using the 720°C catalytic thermal decomposition/chemiluminescence method. Samples were prepared by passing through a 45 µm filter. TN standards of 50 mg/L were run each time with the samples, and blank vials of DI water were placed in between every three to five samples for quality assurance.

Total Phosphorus (TP) concentrations were determined using a Hach DR/2400 Spectrophotometer® following EPA procedure 365.2 for unfiltered, freshwater samples. Each sample set included a blank and dosed sample of 2 mg/L for quality assurance. If the dosed sample was greater than 10% error, the samples were either re-run or multiplied by the error ratio. Additionally, standard checks of 0, 0.5, 1, and 2 mg/L were used at the beginning of each test to ensure the machine was reading correctly. A Maximum Detection Limit (MDL) test following methods from EPA (1997) was performed using seven 0.5 mg/L standard checks taken during each TP sampling analysis using the spectrophotometer, resulting in an MDL of 0.13 mg/L. The data used for analysis is shown in Table A2.

Table A2. Experimental data and s and t values used in MDL testing.

0.5 mg/L Standard Readings		MDL Test	
TP (mg/L)	Date		
0.46	6/26/2015	MDL = s * t	
0.56	6/16/2015	n	7
0.53	5/8/2015	t-test[1]	3.14
0.55	4/9/2015	s, std dev	0.04
0.50	11/29/2014		
0.53	11/11/2014		
0.58	10/22/2014	MDL	0.13

[1]for sample size of 7 and 99% confidence interval, t = 3.14

Water Quality Data. Collected water quality sample lab analysis results for each sampling location can be seen in the following Tables A3-5. The date and time at which each sample was collected were used to determine the corresponding instantaneous discharge and if the sample was collected during the rising, peak, or falling limb of the hydrograph. The 24 hour and 7 day antecedent precipitation was determined by summing the amount of recorded rainfall within the respective time interval before the sample was taken. Table A6 shows the mean TP concentrations with standard errors at each sampling site. TN data was compared to TP data to ensure TP was the limiting nutrient.

Table A3. Site A Water Quality Sampling Data

Date Time	Qi (m^3/s)	Hydro graph Place-ment[1]	TP (mg/L)	TN (mg/L)	TOC (mg/L)	24 hr Antec. Precip.
9/17/2014 8:30	0.12	2	0.60	1.52	12.50	0.15
9/17/2014 9:10	0.11	3	0.23	0.74	4.90	0.13
10/2/2014 8:08	1.09	1	1.90	1.04	4.79	0.45
10/2/2014 8:15	0.54	2	1.42	1.92	10.43	0.65
10/2/2014 13:46	0.33	3	0.25	2.44	12.37	0.24
10/9/2014 14:00	0.07	1	0.39	1.20	8.44	0.07
10/9/2014 14:05	0.38	1	0.32	2.50	23.69	0.1
10/9/2014 15:45	0.11	3	0.32	1.06	4.03	0.05
10/10/2014 11:00	0.11	2	0.31	0.62	0.34	0.11
10/13/2014 9:20	0.05	1	0.79	2.49	10.06	0.06
10/13/2014 9:30	0.37	1	3.20	1.29	8.62	0.09
10/13/2014 10:30	0.18	2	0.46	0.38	1.94	0.18
10/30/2014 16:01	0.08	2	1.86	10.50	3.65	0.08
10/30/2014 16:40	0.02	3	0.65	9.30	3.23	0.03
11/4/2014 15:30	0.02	3	0.21	0.70	2.76	0.01
11/4/2014 12:30	0.05	3	0.43	2.80	2.90	0.06
11/23/2014 14:30	N/A	3	0.90	1.54	4.33	0.09
3/4/2015 11:47	0.01	3	0.16	N/A	0.00	0
4/2/2015 10:41	0.03	3	0.59	N/A	14.70	0.28
4/3/2015 9:16	0.00	3	0.06	N/A	40.00	0
4/9/2015 18:18	0.03	1	0.88	2.72	27.52	0.05
4/9/2015 18:43	0.19	2	0.97	1.12	4.67	0.32
4/9/2015 19:13	0.03	3	0.70	1.74	5.19	0.29
4/9/2015 19:28	0.02	3	0.49	1.65	5.16	0.17
4/25/2015 16:03	0.04	3	0.67	6.90	13.10	0.09
5/8/2015 8:41	0.15	1	1.04	4.66	0.56	0.49
5/8/2015 8:55	0.09	2	0.84	6.19	0.71	0.49
5/8/2015 9:05	0.11	2	0.62	5.93	0.60	0.32
5/8/2015 9:27	0.12	3	0.46	7.24	0.92	0.27
5/8/2015 9:38	0.14	3	0.38	3.91	0.62	0.26
5/8/2015 9:51	0.12	3	0.51	3.71	0.65	0.27
6/12/2015 14:49	0.35	2	0.58	1.08	11.62	0.2
6/12/2015 14:43	0.03	3	0.78	1.35	11.39	0.19
6/12/2015 14:47	0.49	3	1.18	1.62	13.54	0.2
6/13/2015 19:01	0.01	1	0.49	3.18	8.83	0
6/13/2015 19:05	0.08	1	0.83	2.10	23.46	0.01
6/13/2015 19:14	0.03	1	0.85	1.88	26.78	0.37
6/13/2015 19:21	1.41	2	1.32	0.26	1.11	0.67

6/13/2015 19:26	0.44	2	0.95	0.40	1.41	0.68
6/13/2015 19:32	0.19	3	0.85	0.65	2.98	0.69
6/13/2015 19:39	0.07	3	0.82	0.83	4.44	0.69
6/26/2015 11:40	0.03	3	0.33	N/A	10.57	0.05

[1]1 = Rising Limb, 2 = Peak, 3 = Falling Limb

Table A4. Site B Water Quality Sampling Data

Date Time	Qi (m^3/s)	Hydrograph Place-ment[1]	TP (mg/L)	TN (mg/L)	TOC (mg/L)	24 hr Antec. Precip.
9/17/2014 9:00	0.11	3	0.18	0.53	6.50	0.12
9/17/2014 10:25	0.04	1	0.25	0.79	4.26	0.04
10/2/2014 9:36	0.27	2	0.61	1.52	6.10	0.11
10/9/2014 16:21	0.26	3	1.61	1.50	2.30	0.06
10/10/2014 10:55	0.37	2	1.15	0.75	0.66	0.13
10/13/2014 0:20	0.06	2	0.31	0.63	0.96	0
10/30/2014 16:19	0.05	2	0.57	0.67	4.34	0.07
11/4/2014 9:00	0.14	2	0.51	1.30	1.50	0.07
11/4/2014 13:00	0.11	3	0.2	2.80	2.90	0.06
11/23/2014 14:24	N/A	3	0.25	0.10	1.60	0.08
4/2/2015 10:38	0.06	3	1.49	N/A	10.00	0.3
4/3/2015 9:13	0.02	3	0.19	N/A	25.00	0
4/9/2015 18:14	0.28	1	1.25	3.39	8.56	0.05
4/9/2015 18:30	0.21	1	0.85	3.68	14.16	0.34
4/9/2015 18:38	0.37	2	1.93	1.56	4.24	0.34
4/9/2015 18:43	0.35	2	1.35	1.66	4.35	0.32
4/9/2015 18:48	0.15	3	1.17	3.53	5.94	0.3
4/9/2015 18:59	0.05	3	1.1	3.83	5.65	0.3
4/9/2015 19:05	0.04	3	1.1	3.78	6.05	0.3
4/9/2015 19:10	0.04	3	0.75	3.64	5.58	0.29
4/9/2015 19:16	0.03	3	1.01	3.73	5.92	0.29
4/9/2015 19:25	0.03	3	0.24	3.61	6.00	0.17
4/9/2015 19:33	0.02	3	0.2	3.49	4.62	0
5/8/2015 9:08	0.09	3	0.43	1.10	5.40	0.32
6/12/2015 14:45	0.15	1	0.3	0.46	5.39	0.2
6/12/2015 15:00	0.34	3	0.95	1.28	6.04	0.2
6/12/2015 15:16	0.11	3	1.09	2.43	6.67	0.2
6/13/2015 19:11	0.07	1	0.26	1.35	6.35	0.37

6/13/2015 19:55	0.13	3	0.67	1.05	3.90	0.69
6/26/2015 11:33	0.13	3	0.23	N/A	12.00	0.07

[1]1 = Rising Limb, 2 = Peak, 3 = Falling Limb

Table A5. Sites C and E Water Quality

Site[1]	**Date Time**	**TP (mg/L)**	**24 hr Precip.**	**7 Day Antec. Precip.**
C	9/17/2014 8:30	0.29	0.15	0.21
E	9/17/2014 9:10	0.22	0.13	0.28
E	10/2/2014 8:30	0.09	0.66	0.66
E	10/9/2014 15:35	0.14	0.05	1.56
E	10/10/2014 11:00	0.25	0.11	1.61
E	10/13/2014 10:40	0.27	0.16	1.93
C	10/30/2014 15:10	0.16	0.01	0.36
E	10/30/2014 15:40	0.06	0.05	0.4
E	11/4/2014 13:00	0.05	0.06	0.86
E	11/4/2014 17:00	0.32	0	0.93
C	10/31/2015 12:00	0.33	0	0.51
C	10/31/2015 18:00	0.05	0	0.51
E	11/23/2014 13:30	0.14	N/A	N/A
E	6/13/2015 19:48	0.33	N/A	N/A
C	6/26/2015 11:35	0.06	N/A	N/A

[1]C = Lake edge, E = Lake Effluent

Table A6. TP concentrations at water quality sampling locations presented as mean +/- standard deviation.

	Mean TP Concentration (mg/L)			
	Channel		Effluent	
	A	B	Campus	Lake
First Flush	1.07 +/- 0.87	0.94 +/- 0.48	1.40 +/- 0.31	N/A
Peak Flow	0.90 +/- 0.47	0.88 +/- 0.63	0.55 +/- 0.27	N/A
Receding Flow	0.52 +/- 0.29	0.50 +/- 0.37	N/A	0.18 +/- 0.11

APPENDIX B

STREAMFLOW DISCHARGE CALCULATION

Hydrologic Data Collection Methods

Equipment Installation. Four metal stilling wells, permeated with small holes, and nearby USGS staff gages were installed within the stream channels and Frisco Lake. A screened PVC pipe was placed within each stilling well that held and protected a continuous water depth recorder. Equipment set up is pictured in Figure A1. Four Solinst® leveloggers and one Solinst® barologger were installed in the stilling wells at each sampling site. The barologger was located above water level at all times in a nearby monitoring well. Two leveloggers were located at Channel A, one at Channel B, and one at the lake effluent.

Figure B1. Levelogger, PVC stilling well, and metal monitoring well equipment.

Stream Gaging. All leveloggers continuously recorded the total pressure of air and water above the sensor. The barologger continuously recorded the atmospheric barometric pressure and was thus later used to compensate the levelogger data. To capture hydrograph peaks with adequate detail, the sensors continuously recorded at 15 second intervals, and the lake sensor, with less rapid water elevation increases and decreases, recorded at 45 second intervals. Every two weeks data was downloaded from the leveloggers using a direct read cable attached to a laptop computer where the data was stored. Additionally, during field data downloading, the current time and stream gauge measurement were recorded in a field notebook in order to relate the recorded water depth level by the data logger to the stage elevation in future data analysis.

Manual Flow Measurements. Field methodologies used to collect manual flow measurements at both Channel A and B were based upon USGS Techniques of Water Resources Investigations (Buchanan & Somers, 1969); however, modifications to the method were necessitated due to the rapid rise and fall of stage within the channels. Suitable measurement sites were chosen where the stream reach was straight, flow was relatively uniform, and expected flow velocities were within range of monitoring instrumentation. Pygmy and Type 2 AA current meters attached to a top setting wading rod were used during data collection. A spin test was performed before each monitoring event for instrumentation quality assurance.

Measurements were taken at designated observation distances across the channel cross section. Because the channel depths were shallow and rapidly increased and decreased during storm events, the Six-Tenths method was used to calculate average flow velocity. Using the scale on the top setting wading rod, the water depth at each observation point was recorded. This depth was also used to adjust the vertical positioning of the current meter on the wading rod to 0.6 of the total depth. Flow measurements were made by counting the number of revolutions during time intervals between 10 seconds and 50 seconds depending on the rate of change of stream stage. If the stream stage was steady, a larger time interval was used, if the channel depth was changing, shorter intervals were used. At each observation distance, the water depth, number of revolutions, and time interval was recorded, and during each measurement the gauge height was observed and noted. An example flow observation data sheet used during field monitoring can be seen in Figure B2.

Flow velocity field measurements for this project were limited at both Channels A and B. The channels were narrow and flashy, meaning that by the time measurements at each observation point along a profile were completed, the stage may have risen or fallen substantially, nullifying that measurement. Most accurate field observations were those made in the few storms in which an additional field technician was able to record measurements while a second technician took measurements.

Location: UCC Date: 6/13/15 Recorder: JMP/ESA 6/13/15

Observation Depth: 0.6 Time: 7pm Device: Pygmy or AA original

Gauge Reading	Right or Left Edge of Channel	Distance from Initial Point	Width in ft (B)	Depth in ft (D)	Revolutions	Time Interval (sec)	Velocity (V)	Area (A)	Flow (Q) in cfs
4.54	LEW	2'	1.75'	.9	40	8			
↓		3.5'		.8	37	10	[Calculated		
4.54		5.5'		.6	29	10	in Excel]		
4.34	LEW	2'		.7	27	10			
		3.5'		.6	25	10			
↓		5.5'		.5	23	10			
4.25	LEW	2'		.5	27	10			
4.25		3.5'		.4	20	8			
4.24		5.5'		.3	20	10			
4.18	LEW	2'		.4	19	8			
4.18		3.5'		.35	17	10			
4.17		5.5"	↓	.3	18	10			

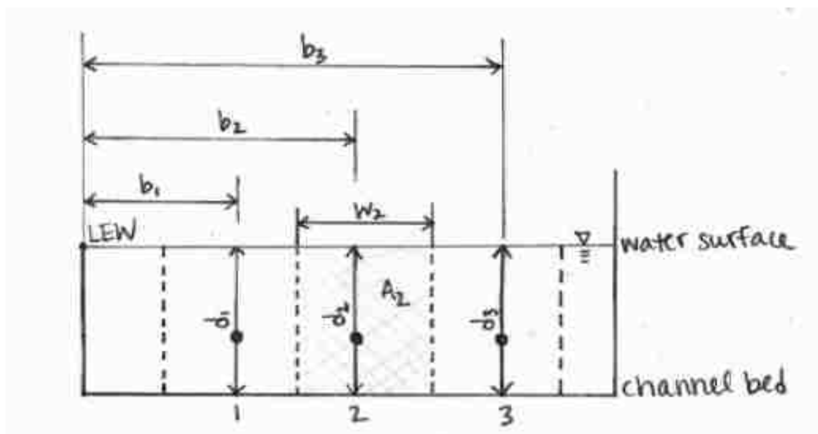
Figure B2. Field flow data sheet with observations from 6/13/15 storm event.

Discharge Calculation. Manual field measurement data were copied into Microsoft Excel® spreadsheets where the meter rating was used to convert revolutions per minute into a linear velocity in feet per second. These velocities were used as the average flow velocities across each of the sub-section areas that were calculated by multiplying sub-section width by observed depth. The discharges, in cubic feet, were calculated by multiplying the sub-section areas by the observed average flow velocity and summing. An example calculation sheet, showing columns noting gauging location, observation depth, revolutions, and time intervals recorded for each field measurement taken at Channel A can be seen in Figure B3. The additional columns calculate the flow velocity, cross sectional area, and flow. A similar calculation process was used for discharge calculation at Channel B.

Location:	UCC				Date:	5/8/2015	Recorder:		Use AA for depthsof 1.5 to 2.5 ft; V= 2.2048 + 0.0178				
							Device:	Pygmy or AA	Use Pygmy for depthsof .3 to 1.5 ft; V = 0.9604 R +				
Downstream Gauge Reading	Upstream Gauge Reading	Time	Right or Left Edge of Channel	Distance from Initial Point	Subsection Width B (ft)	Subsection Avg Depth D (ft)	Obs. Depth (0.6)	Revolutions	Time Interval (sec)	ons per Second R	Velocity V (ft/s)	Area A (sq ft)	Flow Q (cfs)
	4.18		LEW	0			0.6					0	0
				2	1.75	0.4		27	10	2.7	2.62428	0.7	1.837
				3.5	1.75	0.4		27	10	2.7	2.62428	0.7	1.837
				5.5	1.75	0.3		29	15	1.93333	1.88797	0.525	0.99119
				7									4.66518
	4.16		LEW	0									
				2	1.75	0.4		27	10	2.7	2.62428	0.7	1.837
				3.5	1.75	0.4		25	12	2.08333	2.03203	0.7	1.42242
				5.5	1.75	0.3		18	10	1.8	1.75992	0.525	0.92396
				7									4.18338
	4.17		LEW	0									
				2	1.75	0.4		33	15	2.2	2.14408	0.7	1.50086
				3.5	1.75	0.4		34	15	2.26667	2.20811	0.7	1.54567
				5.5	1.75	0.3		31	15	2.06667	2.01603	0.525	1.05841
				7									4.10494
			LEW	0									
	4.1			2	1.75	0.3		24	15	1.6	1.56784	0.525	0.82312
	4.09			3.5	1.75	0.25		26	15	1.73333	1.69589	0.4375	0.74195
	4.08			5.5	1.75	0.25		27	15	1.8	1.75992	0.4375	0.76997
	4.09			7									2.33503
			LEW	0									
	4.12			2	1.75	0.3		17	10	1.7	1.66388	0.525	0.87354
				3.5	1.75	0.275		18	10	1.8	1.75992	0.48125	0.84696
				5.5	1.75	0.25		19	10	1.9	1.85596	0.4375	0.81198
				7									2.53248
			LEW	0									
	4.2			2	1.75	0.45		34	15	2.26667	2.20811	0.7875	1.73888
	4.2			3.5	1.75	0.4		37	15	2.46667	2.40019	0.7	1.68013
	4.18			5.5	1.75	0.35		33	15	2.2	2.14408	0.6125	1.31325
	4.19333333			7									4.73226
			LEW	0									
				2	1.75	0.5		27	10	2.7	2.62428	0.875	2.29625
	4.22			3.5	1.75	0.5		27	10	2.7	2.62428	0.875	2.29625
				5.5	1.75	0.4		22	10	2.2	2.14408	0.7	1.50086
				7									6.09335

Figure B3. Discharge calculation using observed flow velocities.

Channel A. Three observation points were positioned at the midpoints of sub-section areas that were used to determine the overall stream discharge by summing all the discharges from the sub-sections. The stream channel cross section dimensions were measured during periods of dry weather using a tape measure and observation points marked with spray paint on the side of a small bridge over the channel. The channel cross section was rectangular in shape and 7 feet across the bed. The initial point, or 0 foot, measurement was located on the east edge of the channel, five feet downstream of the channel staff gauge, and noted as Left Edge of Water (LEW) in field notes. Observation locations were located at 2, 3.5, and 5.5 feet from the left edge of water. The profile was broken up into three subsections with boundaries located at the midpoints between observation points at 1', 2.75', 4.5', and 6.25' from the LEW. These boundaries can be seen in Figure B4 as the dashed lines. The sub-section widths, denoted as w, were multiplied by the observed water depths to determine the sub-section areas. The observed flow at each subsection was multiplied by the area to determine the sub-section discharges.



1,2,3	Flow Observation Points
LEW	Left Edge of Water
b_1, b_2, b_3	Distance from initial point to observation points
d_1, d_2, d_3	Depth of water at observation point
Dashed lines	Subsection area boundaries
A_2	Subsection area, determined by multiplying w_2 by d_2
w_2	Subsection width

Figure B4. Field observation points along Channel A cross section.

Channel B. Discharge was calculated from a single cross sectional area as one observation point located in the center of the channel was used during field monitoring. During a dry period, the empty channel cross section dimensions were measured using surveying tools. The channel profile at the monitoring location can be seen in Figure B5. A single observation point was located in the center of the channel at 10 feet upstream from the staff gage and stilling well in a narrower part of the channel. A formula relating cross sectional area and water depth was contrived using the measured dimensions and was used in determining the streamflow discharge. Streamflow discharge was calculated using the same process as Channel A as shown in Figure B3.

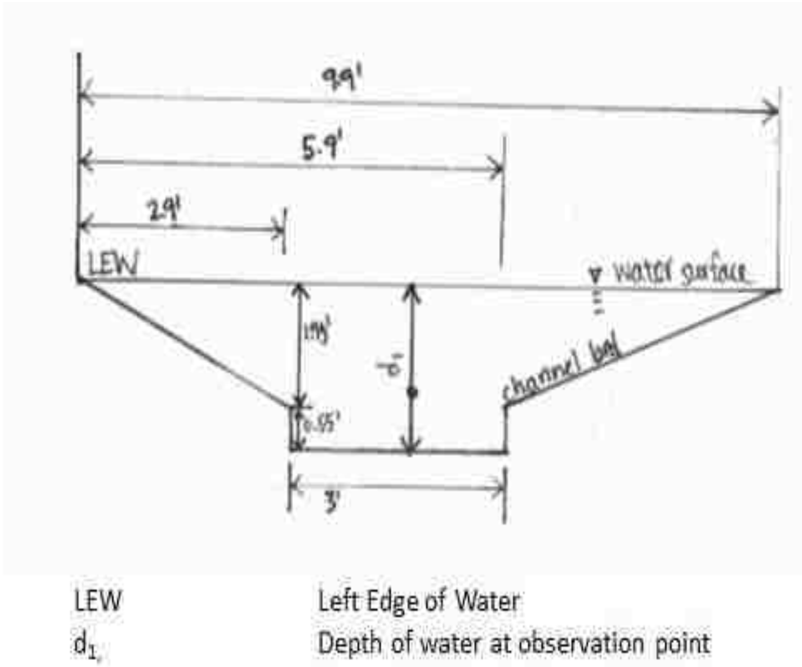


Figure B5. Field observation point along Channel B cross section.

Determining Backwater Corrected Discharges. Both channels experienced backwater effects as the lake water elevation rose during wet-weather events, so a Stage-Fall-Discharge rating for each channel was developed following standard USGS methods (Sauer, 2002). Observed stream discharges and corresponding channel stages at the time of measurement were related creating a Stage-Discharge rating. Additionally, a Stage-Fall curve using observed stage data and levelogger records between upstream and downstream gages was created for each channel and can be seen in Figure B6. As realized in Figure B6 showing a better fitting rating curve, the positioning of Channel B's upstream and downstream gauge was more favorable to monitor as the increased distance between the gauges allowed for a greater water elevation fall between the gauges, reducing the error in readings. The flow through Channel A was more turbulent and the range in water elevation fall between the gauges was 0 to 6 inches, requiring measurements to be more precise. The flow at the downstream gauge at the lake was steadier and the range in water surface elevation fall was 0 to 1.5 feet at periods of no flow to peak flows, respectively.

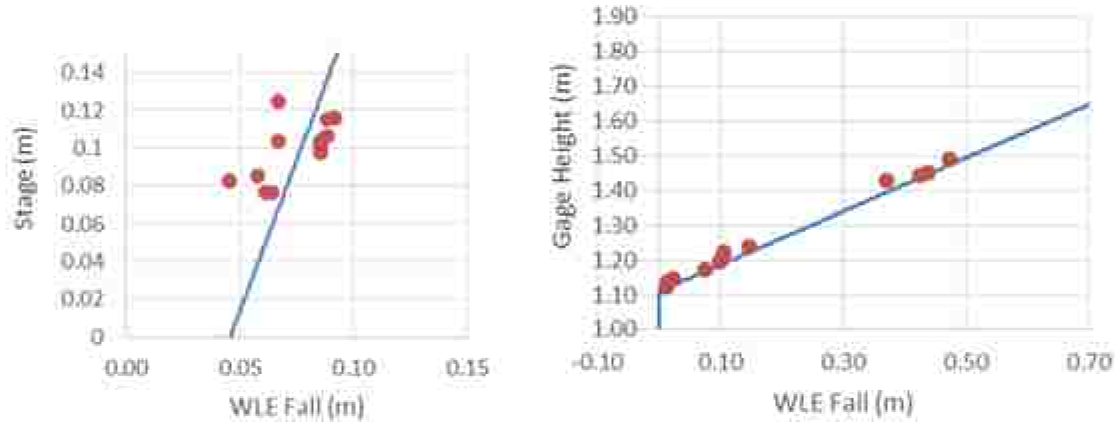


Figure B6. Stage-Fall ratings used in backwater adjusted discharge determination for Channel A (left) and Channel B (right).

An equation using four parameters: Stage-Discharge rating, Stage-Fall rating, the continuously recorded water elevation fall between up and downstream gages, and recorded stages were used to determine the continuous discharge time series. The backwater adjusted discharge equation B1 is seen below:

$$Qadj = Qr\sqrt{\frac{Fm}{Fr}} \qquad \text{(B1)}$$

where $Qadj$ is corrected discharge in m^3/s, Q_r is discharge rating in m^3/s, F_m is observed water elevation fall between up and downstream gauges in m, and F_r is the water elevation fall rating in m.

Channel A Rating Curve. The Stage-Discharge rating used the power trend-line linear regression between observed discharges and stages collected during three storm events, and is pictured in Figure B6. The rating equation closely aligned with the Manning's Equation for open channel flow using the physical dimensions for a concrete channel, providing assurance that the rating curve was appropriate for the channel. The adjusted discharges using the backwater correction Equation 1 are plotted in Figure B7. Adjusted discharges fall within 10% error of observed discharges, providing additional quality assurance that discharges were being properly calculated. Figure B7 presents the equation of the rating curve used in discharge determination with the R squared value of 0.9571, suggesting that greater than 95% of the variability of the data can be explained using this equation.

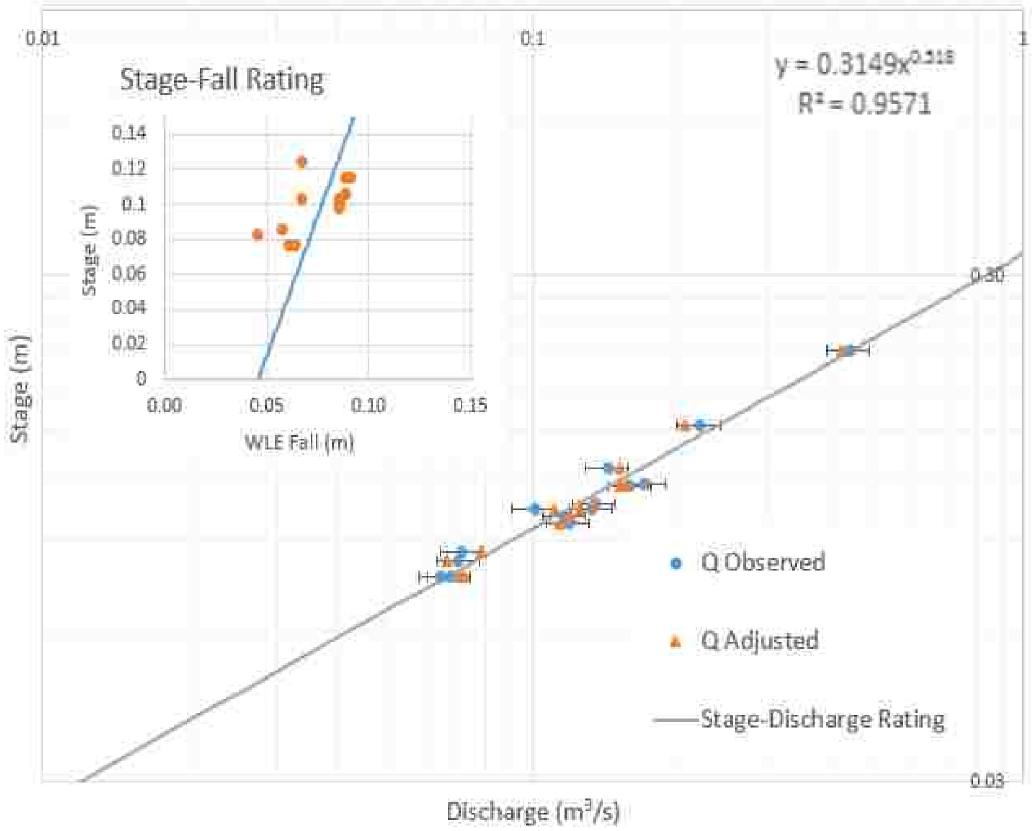


Figure B7. Channel A's rating curves and complex rating adjusted discharges, the triangles, compared to observed discharges, the circles, showing that computed discharges fall within 10% error.

Channel B Rating Curve. The Stage-Discharge rating used to create a continuous streamflow record was formed using the power trend line relationship between observed discharges and stages collected during two storm events, and is presented in Figure B8. The rating curves and the adjusted discharges using the backwater correction Equation 1 are plotted in Figure B7. Most adjusted discharges fall within 10% error of observed discharges, though others fall out of the range. This is presumably due to the rougher fit of the rating curve to the data at this channel as consistent field measurements were difficult due to varying vegetation growth and channel roughness, gradual changes in channel shape, and the instability of the streambed. However, the Stage-Fall rating curve is of greater quality than that of Channel A due to the increased distance and slope between upstream and downstream gauges as well as the less turbulent flow at the lake effluent than within the channels.

A few streamflow data sets did not have corresponding downstream gauge information to correct for backwater. These infrequent data sets used only the Stage-Discharge rating curve and, therefore, provide overestimated discharges during periods of wet-weather when the lake surface would rise.

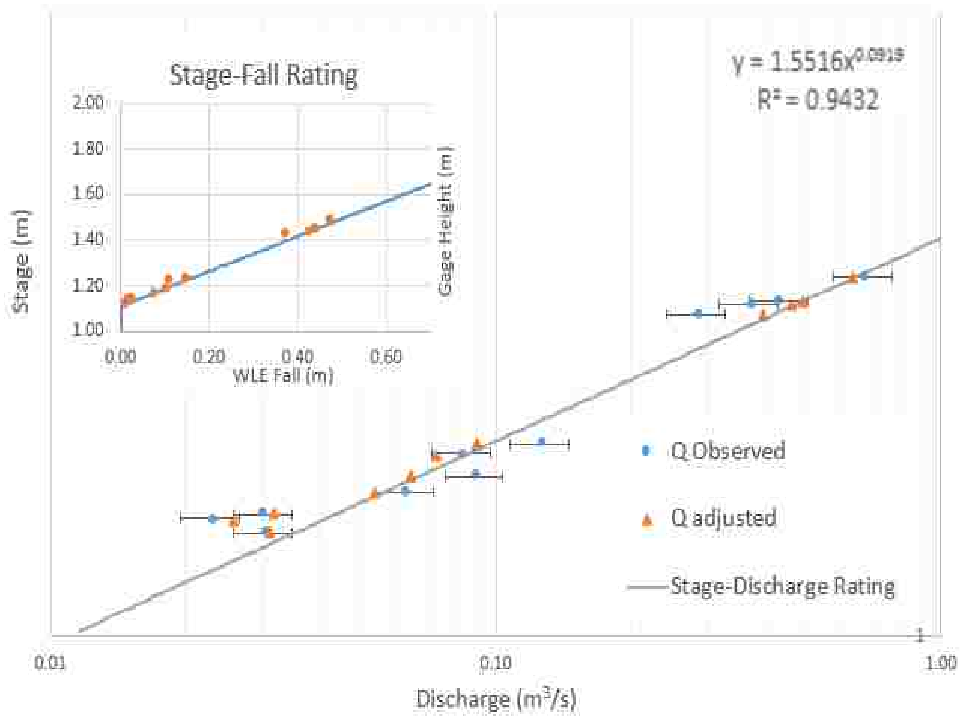


Figure B8. Channel B's discharge rating curves and complex rating adjusted discharges compared, triangles, to observed discharges, circles, with 10% error bars. The rating curve equation and R squared value is presented at the upper right corner.

Creating Discharge Time Series. Stage data from the four gages was downloaded and compensated using the barometric pressure data with the Solinst® Data Wizard software. The auxiliary gage at the lake effluent did not collect data during parts of fall and summer, and no gages collected data over the winter due to instrument restrictions. The data was further analyzed using a desktop computer in the lab using Microsoft Excel®. Data sets, consisting of a base and auxiliary gage record for each channel, were recorded in approximately 2.5 week intervals, totaling 36 records over the course of monitoring period. Stage data was collected at 15 second intervals, as necessitated by the rapid rises and falls of channel stages noticed during rain events. Manual stage recordings at noted dates and times were used to reference the levelogger reading to the water level elevation, as slight variations in the sensors' vertical positions within the monitoring wells between data intervals were noticed. An example table of manual readings used to reference the corresponding electronic data record is seen in Table B1.

Table B1. Observed stages used to reference levelogger data records.

Observed Upstream Gage Readings		
Levelogger Data Set	**Date Time**	**Gage Reading (ft)**
1	9/17/2014 9:53	4
1	9/17/2014 9:58	3.98
2	10/13/2014 8:18	3.86
2	10/2/2014 8:37	3.98
2	10/2/2014 8:42	4.03
2	10/2/2014 8:56	3.97
2	10/2/2014 8:47	4.06
2	10/2/2014 9:10	3.76

An annual discharge time series for each channel was created compiling each manually corrected data set with continuously recorded stage and water elevation drops between up and downstream gages and the appropriate Stage-Fall-Discharge or Stage-Discharge rating to convert into a discharge measurement.

Instrument Error Correction. Diurnal fluctuations in water level were noticed that correlated with ambient temperature fluctuations. These fluctuations were as a result of the instrument's failure to compensate for stresses induced from temperature effects within the probe and therefore incorrectly displayed water level elevations. For example in Data Set 11, these fluctuations were small (0-2 cm), however, due to the specific, relatively small range of water surface elevation differences (0-30 cm) between upstream and downstream gages, the error distorted the discharge record. Therefore, each data set required manual correction using two assumptions based upon logical, scientific principles, in order to correct some of the error induced by the erroneous temperature fluctuations:

1. The downstream gage water elevation reading should never be higher than the upstream water elevation reading
2. The stage reading should be as close to zero as possible during periods of dry weather

An example of a continuously recorded stage data set with erroneous water level fluctuations requiring manual correction can be seen in Figure B9.

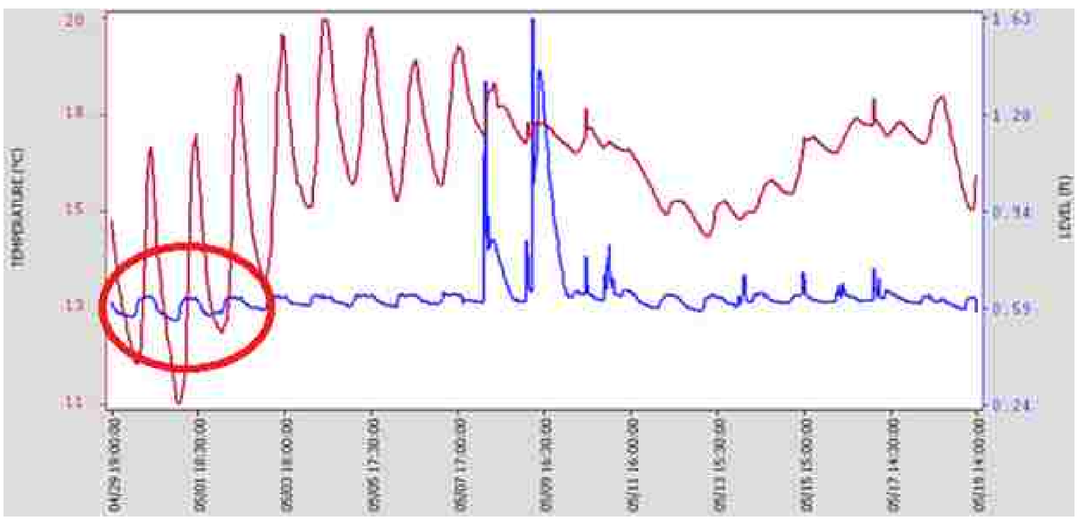


Figure B9. Upstream gage stage data set exemplifying diurnal temperature fluctuation instrument reading error.

Meteorological Data. The precipitation data used in the site hydrologic analysis was taken from the campus weather station located on top of Emerson Electric Hall on the S&T campus. This weather station was located within the study area and approximately 0.4 km from the monitoring site. Precipitation data was continuously recorded at five minute intervals using a tipping bucket gauge and downloaded every 2 weeks.

The average annual rainfall in Phelps county is 42 inches a year (USDA, 2002). The running 30 year average annual rainfall at Station 7.6 SSE Rolla (at lat./long. 37.8503, -91.7055) was 46.29 inches, and the annual rainfall for the 2015 water year, which corresponds to beginning and end of the monitoring study period, was 37.59 inches (CoCoRaHS, 2015). Figure B10 presents the monthly precipitation totals for the 2015 water year showing wetter than normal summer months and drier fall months. This figure suggests that the year of which the Frisco Lake watershed monitoring study took place was slightly drier than a typical year; and, therefore, the annual TP loads and catchment yields may have been slightly underrepresented.

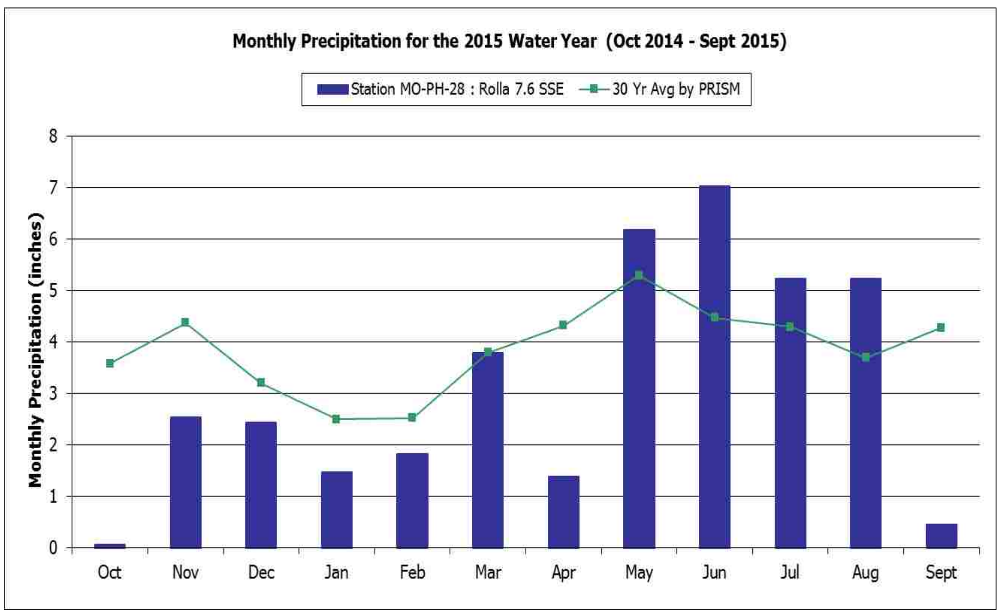


Figure B10. Monthly precipitation totals in Rolla, MO during the WQ monitoring period in blue with the 30 year running averaged superimposed.

QA/QC: Hydrograph Analysis. Using the annual discharge time series record, as well as continuously recorded precipitation data collected within the contributing catchments, individual event hydrographs were used to compare runoff volumes to rainfall volumes. Runoff volumes were calculated by integrating the area underneath the hydrograph curves, and rainfall volumes were calculated by multiplying the contributing drainage area by the measured rainfall amount from the precipitation gage located within

Catchment A on top of Emerson Electrical Hall on the Missouri S&T campus. Rainfall-Runoff ratios generally fell within realistic ranges, however some events appeared underestimated, ratios approximately 0.3, and particularly during events without downstream gage information to correct backwater effects to the stage record, some events had higher (closer to 1) rainfall-runoff ratios. A table of storm summaries of both Channel A and B with date, time, rainfall intensity, and rainfall and runoff is shown in Table B2. Rainfall was calculated by summing recorded precipitation. Storm event duration was calculated by determining the length of time between the first recorded rainfall and last during the event. Individual events were defined to have greater than 0.5 in of rain separated by six hours of no rainfall. Intensity was determined by dividing recorded rainfall by duration.

Additionally, individual hydrographs were compared using the continuous discharge time series. Hydrographs reflected realistic hydrologic patterns, for example Channel A hydrographs were flashier than those in Channel B, as the watershed was more condensed. Channel B's catchment was longer in length which provided more attenuated flows over a longer duration. The hydrograph record on October 28, 2014 for both Channel A and Channel B is shown in Figure B11.

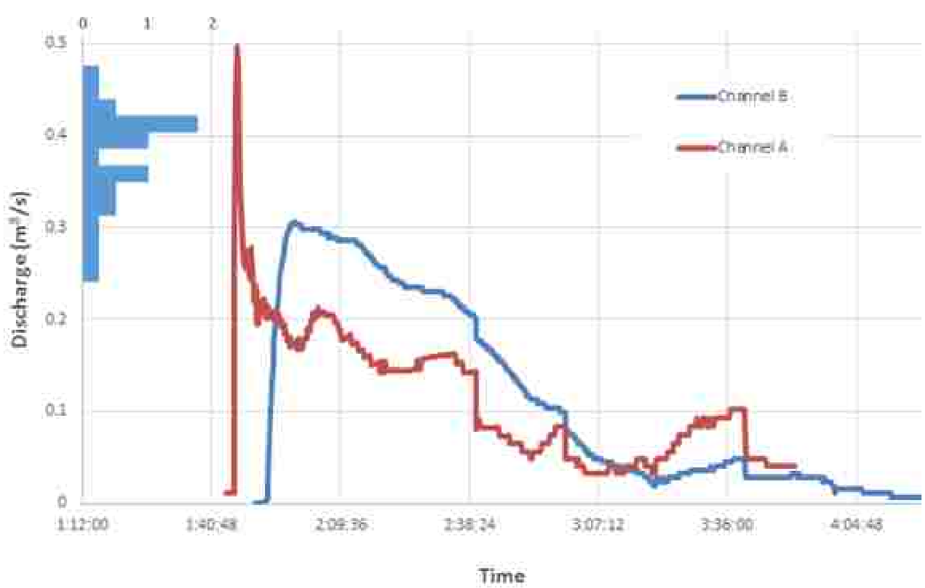


Figure B11. Event hydrograph showing rainfall pattern (top left) and peak runoff flows at Channel A and B on 10/28/2015.

Table B2. Rainfall-Runoff analyses for both channels including storm event date and time, rainfall, duration, intensity, and runoff ratio.

Rainfall Vs. Runoff Analysis Channel A

Storm Event Date and Start Time	Rainfall (mm)	Duration (hr)	Intensity (mm/hr)	Rainfall Volume (m3)	Runoff Volume (m3)	RO/RF
9/17/2014 7:40	7.87	2.25	3.50	1721	1060	0.62
9/21/2014 3:35	3.81	0.33	11.44	833	593	0.71
10/2/2014 7:47	56.64	16.70	3.39	12378	8302	0.67
10/7/2014 0:45	5.84	2.08	2.81	1277	754	0.59
10/13/2014 9:10	19.30	36.00	0.54	4219	3728.00	0.88
10/28/2014 1:25:00	8.13	2.75	2.96	1776	904.00	0.51
10/30/2014 15:10:00	2.79	4.00	0.70	611	222.00	0.36
11/4/2014 2:00:00	20.83	13.08	1.59	4552	3549.00	0.78

Rainfall Vs. Runoff Analysis Channel B

Storm Event Date and Start Time	Rainfall (mm)	Duration (hr)	Intensity (mm/hr)	Rainfall Volume (m3)	Runoff Volume (m3)	RO/RF
9/17/14 7:40	7.87	2.25	3.50	1259	869	0.69
9/21/14 3:35	3.81	0.33	11.44	609	373	0.61
2/1/15 0:30	22.61	10.25	2.21	3614	2592	0.72
3/10/15 21:40	11.68	11.00	1.06	1868	1431	0.77
3/24/15 7:48	7.87	2.33	3.38	1259	728	0.58
3/24/15 19:00	10.67	1.50	7.11	1705	525	0.31
4/9/15 17:35	8.64	0.92	9.39	1380	849	0.62
4/19/15 5:35	6.10	1.75	3.48	974	665	0.68
5/8/15 7:30	29.46	4.83	6.10	4710	1608	0.34
5/29/15 22:25	22.86	2.67	8.56	3654	2385	0.65

Continuous Discharge Record. After the runoff volumes and hydrographs from sample rain events were quality checked, each manually corrected discharge data set was compiled sequentially into a continuous discharge record for the monitoring period. The results can be seen in the following Figures B12-19. The discharge record has noticeable gaps during winter months, due to instrument incapability to monitor during freezing temperatures, mishandling of data, or inaccessibility to open wells to monitor data.

Study Period Channel A Continuous Discharge Record

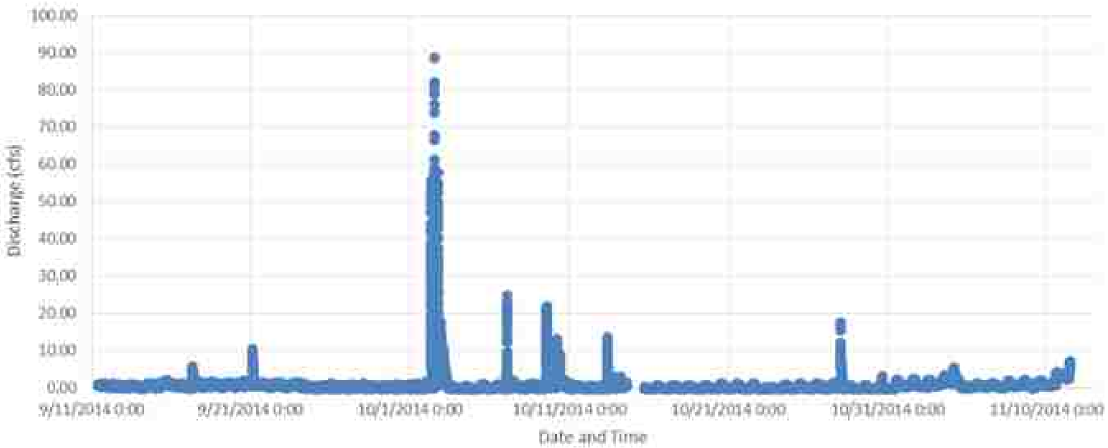


Figure B12. Continuous discharge record from 9/11/2014 to 11/11/2014

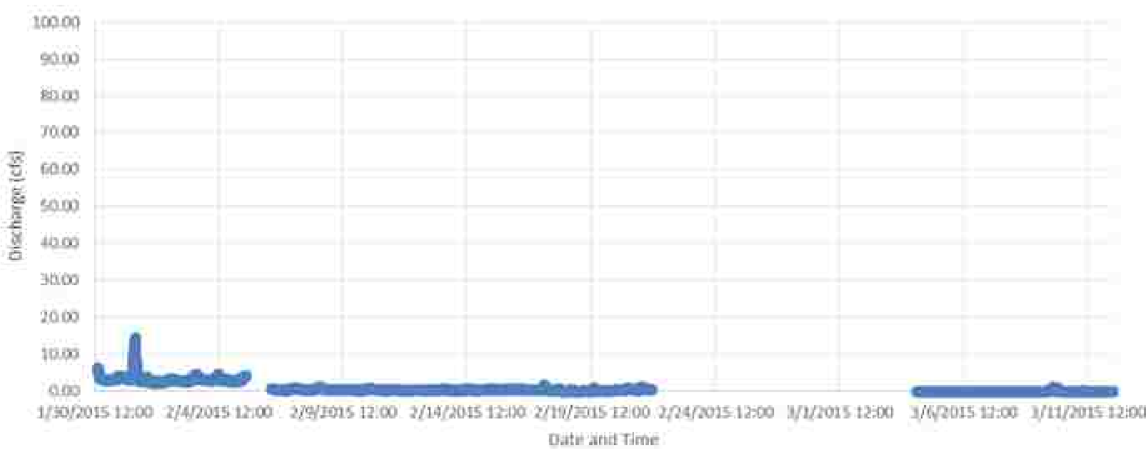


Figure B13. Continuous discharge record, with gap, from 1/30/2015 to 3/12/2015

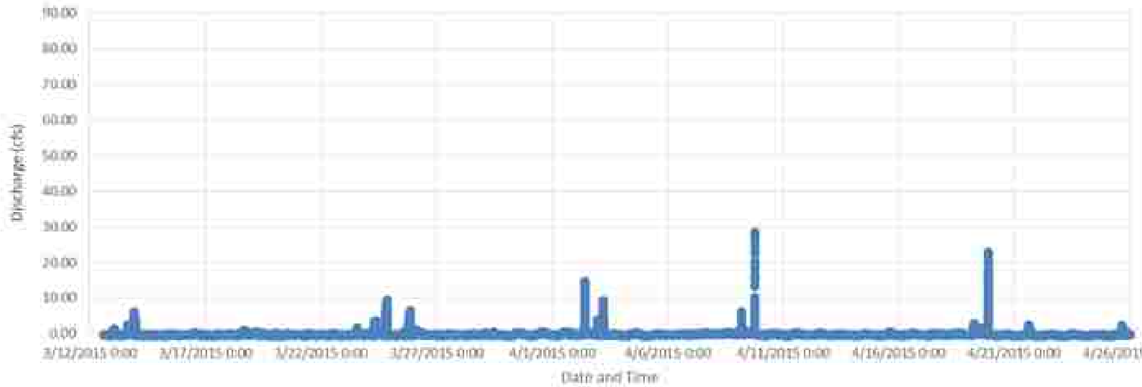


Figure B14. Continuous discharge record from 3/12/2015 to 4/26/2015

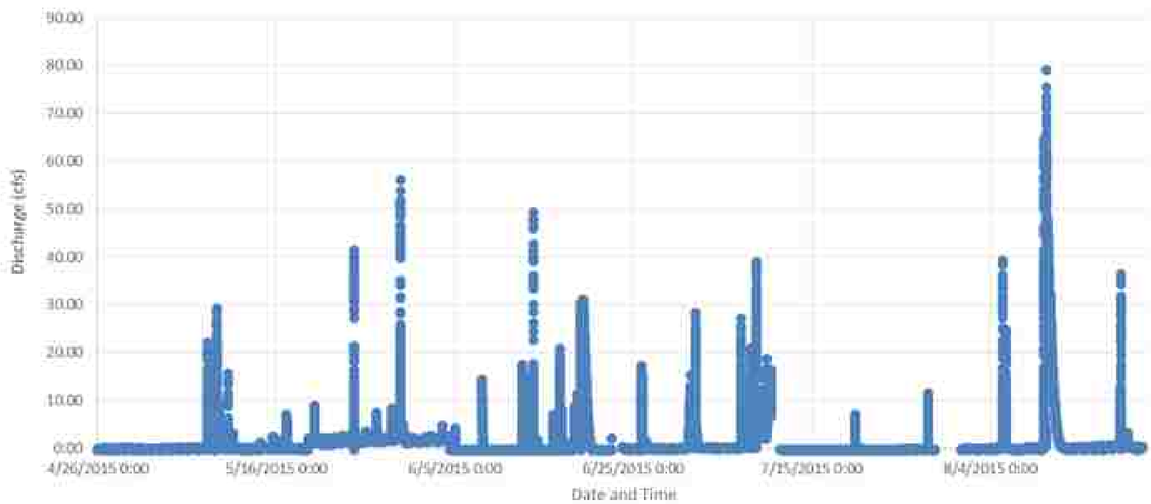


Figure B15. Continuous discharge record from 4/26/2015 to 8/21/2015

Study Period Channel B Continuous Discharge Record

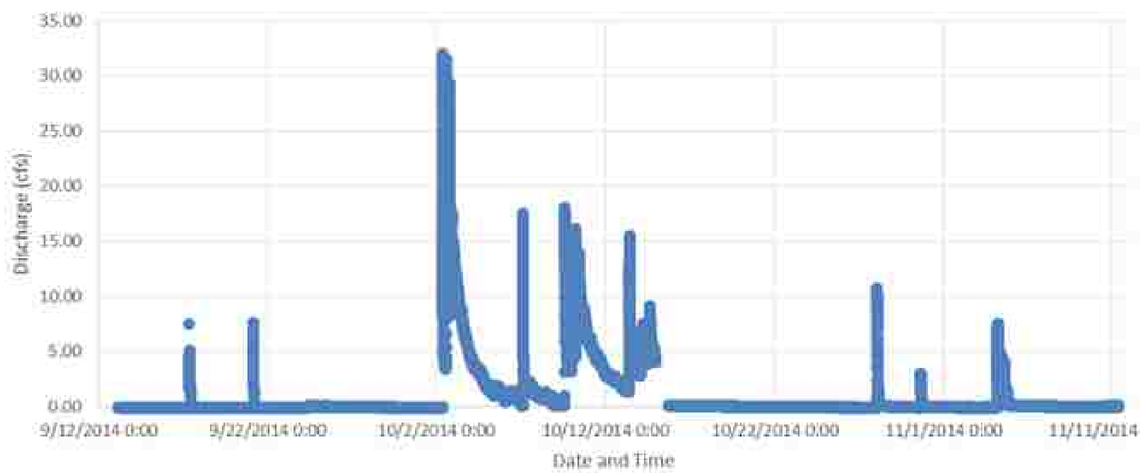


Figure B16. Continuous discharge record from 9/11/2014 to 11/11/2014

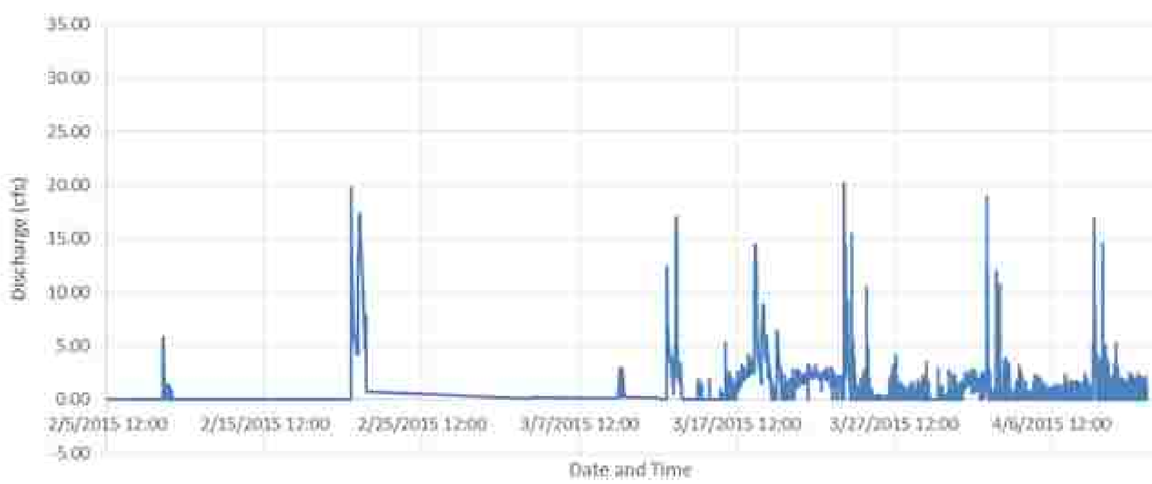


Figure B17. Continuous discharge record from 2/4/2015 to 4/12/2015

Figure B18. Continuous discharge record from 4/12/2015 to 5/16/2015

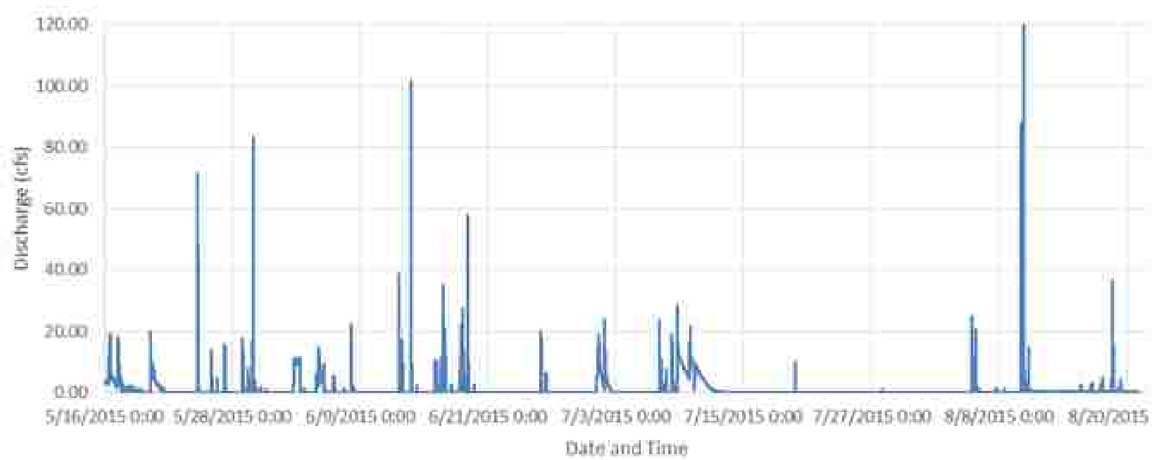


Figure B19. Continuous record from 5/16/2015 to 8/21/2015

APPENDIX C

NUTRIENT LOADING ANALYSIS

Nutrient Loading Modeling Methods

Rating Curve Model. Conventional load rating techniques from Cohn *et al.*, (1989) using the collected water quality data, were used to calculate the log-log concentration-discharge linear regression (power trend-line) relationship for each channel. The trend equations can be seen in Figure C1.

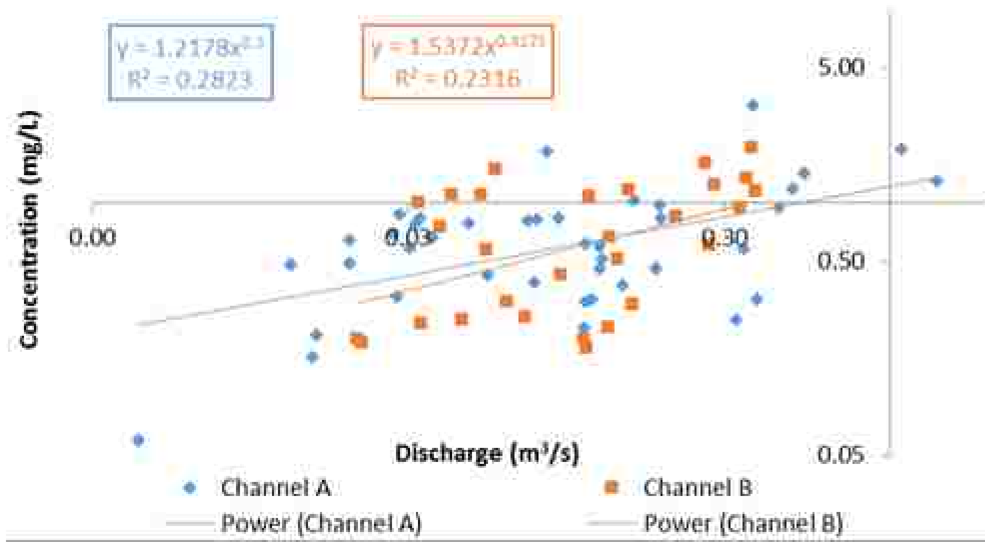


Figure C1. Channel A and B TP concentration-discharge information with linear regression (power trend line) equations fit to data and shown at upper left.

TP Load Calculation. Annual TP loads were determined using Channel A and B linear regression rating curve models, equations are shown in Figure C1, and the calculated annual discharge record from Appendix B. Discharge information used for loading analysis was limited to periods of wet-weather flows. Additionally, the discharge information during the winter season was not included, due to instrument incapability of monitoring during freezing temperatures and unreliable or missing barometric pressure data. A TP concentration time series for each channel was generated using the corresponding continuous discharge records and concentration-discharge linear regression relations. Loads were then computed using equation C1:

$$L = CQKt \quad \text{(C1)}$$

where L represented the load (kg), C was the TP concentration (mg/L) determined using the linear regression nutrient loading model, Q was the previously calculated discharge (m^3/s), K was a unit conversion factor (86.4), and t was the time interval of flow measurement (15 s).

Loads using rating curve models are often found to underestimate loads due to a statistical bias (Cohn *et al.*, 1989). Therefore, estimated annual loads using the linear regression rating curve method were then adjusted by a correction factor determined using a second, more specific and labor-intensive loading analysis. This project assumed that using the simplistic rating method to estimate annual loading and multiplying by the corresponding calculated correction factor determined at each channel for the specific storm events would provide a more realistic estimate of nutrient loads.

Concentration Curve Loading Model. A labor intensive computational Technique of Water Resources Investigations (Porterfield 1972) was used to calculate TP loads during three storm events, in which an estimated concentration loading curve that followed the shape of the hydrograph through observed concentrations along the hydrograph profile was generated. This method was used where detailed data was available, as it yields a more accurate load estimation than simpler methods (Porterfield 1972). Three storms had detailed, hydrographic sampling data that were superimposed over event hydrographs and used to complete individual storm analyses. Event loads were determined by integrating the area under the curve by multiplying the estimated instantaneous TP concentration and discharge by the 15 second time interval and a unit conversion factor. The four event analyses are presented in following Figures C2-5.

Hydrographic sampling at Channel B was conducted during several storm events; however, only one event corresponded with the rising, falling, and peak portions of the hydrograph and was used in analysis. Hydrographic sampling along the rain event hydrograph during three storm events at Channel A were captured. To increase representativeness and accuracy of this nutrient loading comparison, more events with hydrographic sampling would need to be completed, particularly at Channel B as there was a wide variance between the percent changes between the events.

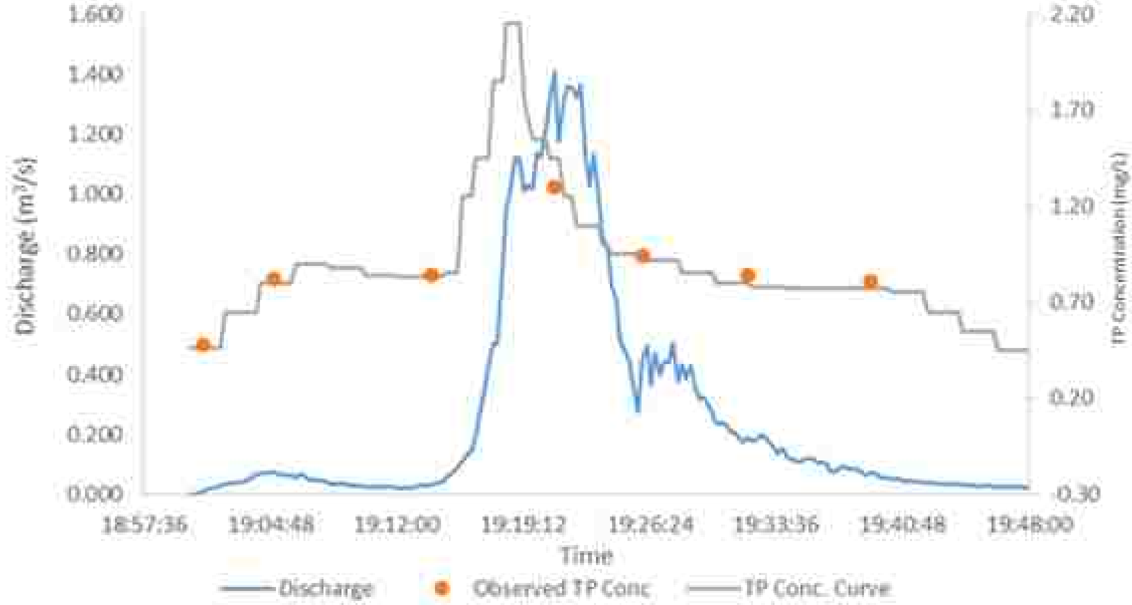


Figure C2. Concentration Curve Method used to estimate 6/13/15 event load.

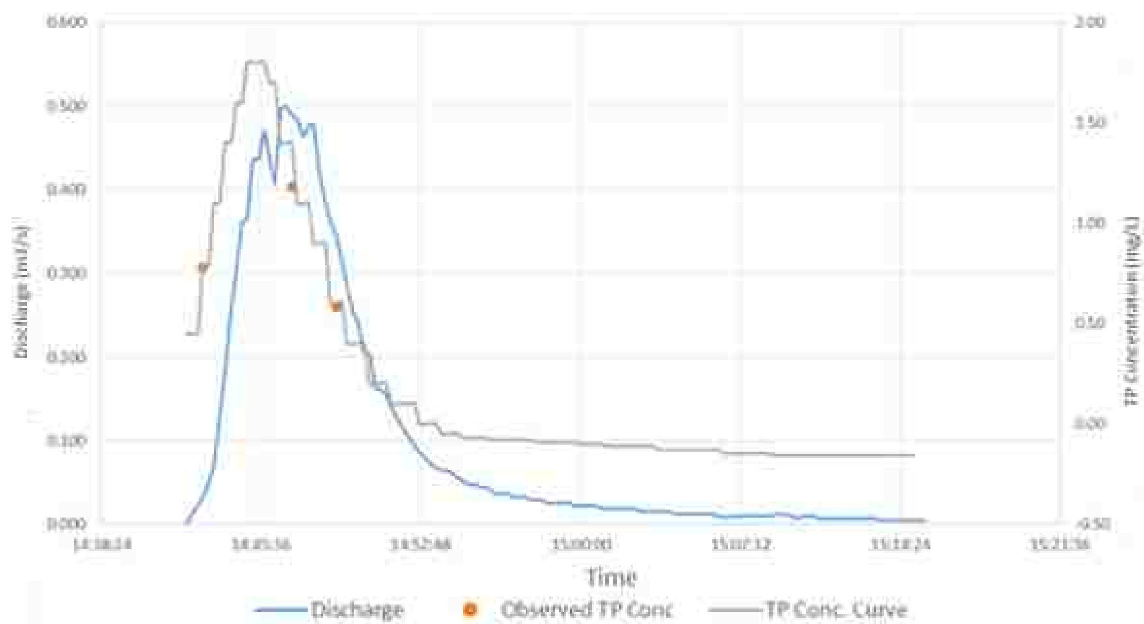


Figure C3. Concentration Curve Method used to estimate 6/12/15 event load.

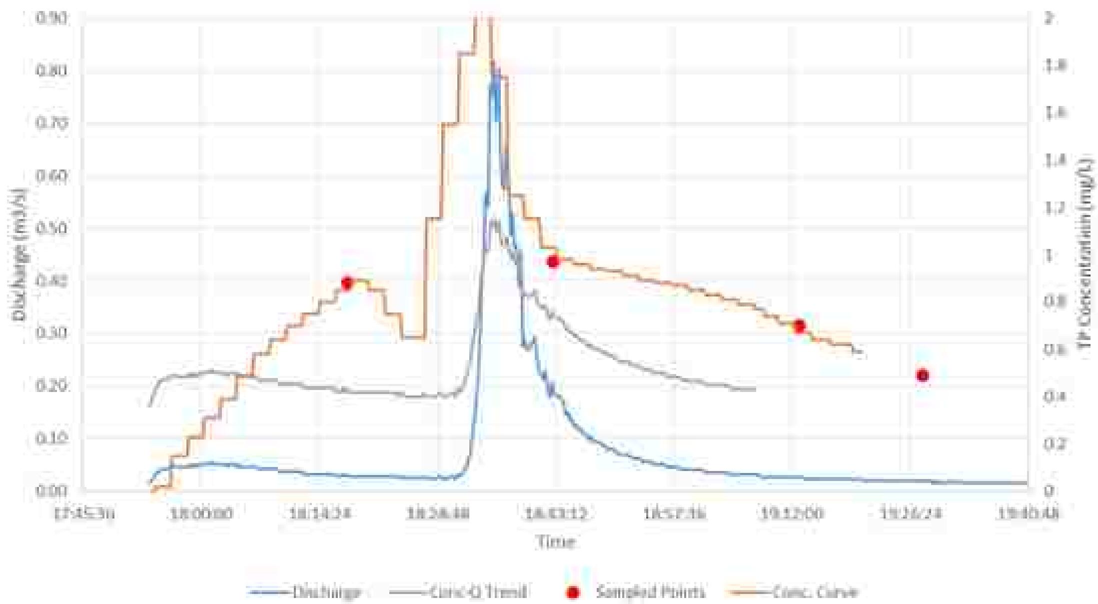


Figure C4. Concentration Curve Method used to estimate 4/9/15 event load of Channel B.

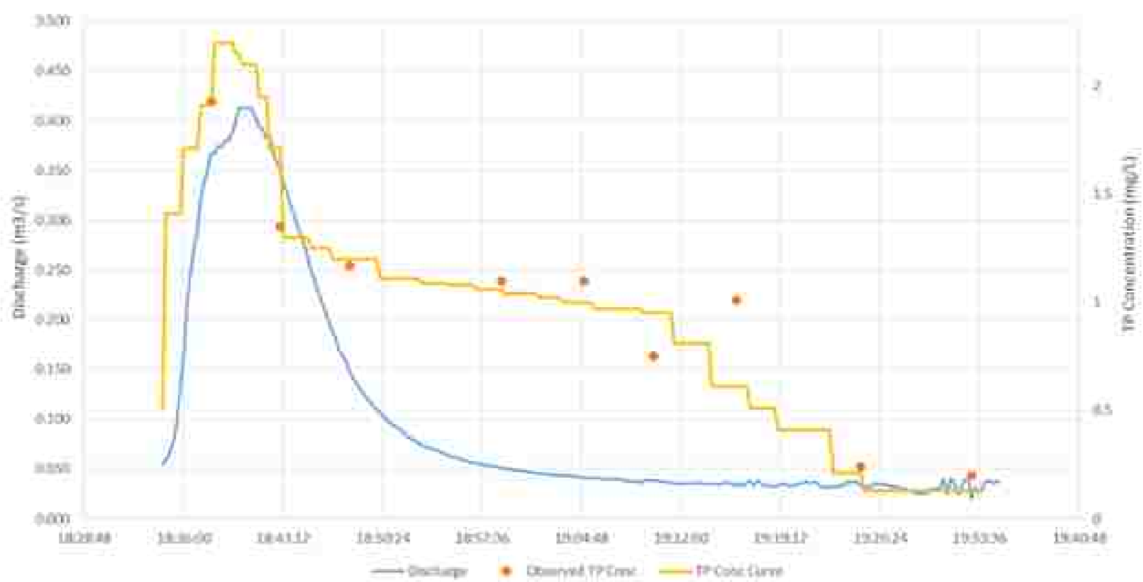


Figure C5. Concentration Curve Method used to estimate 4/9/15 event load of Channel B.

Model Comparison. The TP loads for three storm events, 4/9/2015, 6/12/2015, and 6/13/2015, were calculated using each previously described nutrient loading determination method. The results were compared using a percent change analysis, in which the loads determined using the concentration curve model were divided by the load determined using the linear regression model and converted to a percent by multiplying by 100. The results are summarized in Table C1.

	Event Loads (kg)			
	Channel A			Channel B
Model	4/9/2015	6/13/2015	6/12/2015	4/9/2015
Conc. Curve	0.47	0.92	0.18	0.51
Regression	0.36	0.8	0.16	0.28
% Change	31	15	13	82
Average % Change	35			

Table C1. TP event loading results using both models and percent change comparison.

The four event percent change comparisons between the TP load estimates using the concentration curve and regression model analysis were averaged, seen in the bottom row of Table C1. This average percent change of 35% was used as an adjustment factor in the annual load determination.

Annual Load Determination. The September 2014 to August 2015 (with winter season omitted) wet-weather discharge record for each channel (see Appendix B) and the linear regression models were used to determine a continuous TP concentration record. The wet weather record was determined by omitting continuously recorded stage data points where 24 hour antecedent precipitation totaled less than zero. This was accomplished using the recorded precipitation data (described in Appendix B). Using Eqn. C1, an annual load (in kg) was determined for each channel. This load was then multiplied by 1.35, equal to the average increase noticed between the linear regression model and concentration curve method. The adjustment factor increased total annual Channel A and B loads by 35% as the linear regression model was assumed to have under-estimated the load, and the concentration curve model was assumed to provide a more accurate estimate. The loads from each channel were then summed to determine an annual TP load into the lake. Catchment yield rates were determined by diving the annual load by the catchment area. Results are shown in Table C2.

Table C2. Annual TP loads and yields and drainage areas at each channel.

	Catchment Area (ha)	Load (kg)	Yield (kg/ha/yr)
Channel A	22	380	17
Channel B	16	560	31
Total	38	940	

The catchment yields for each channel are both higher than literature values. Typical land area yields that are used for non-point source TP pollution coefficients in watershed models employed by the USGS can range over five orders of magnitude from 0.001 to 7.2 kg/ha/yr, with accepted urban land use rates of 3.63 kg/ha/yr (Alexander *et al.*, 2004). Therefore, one would expect the estimated catchment yields of 17 and 35 kg/ha/yr to be overestimates by roughly an order of magnitude. However, the variability existing within literature values of catchment yields suggests that the accuracy of yield determinations is limited. Considering the time and resource limited data collection methods used in this project, a best estimate approach was taken in determining catchment yields. These values are useful as qualitative estimates to guide watershed planning strategies.

Lake Eutrophication Modeling. A conventional mass balance modeling technique (Davis and Masten 2009) was used to determine external nutrient load reductions that followed an equation of the form:

Eqn. C2 $$V\frac{dC_{out}}{dt} = QC_{in} - QC_{out} - kC_{out}V$$

where V is the lake volume determined by the average depth and surface area; Q is the sum of flows at Channel A and B, the mean-annual flow into the lake (m^3/s); C_{in} and C_{out} are TP influent and effluent concentrations (mg/L); and k is a settling rate (s^{-1}). The model parameters included known physical lake characteristics surface area (20,235 m^2)

and average depth (1.5 m), a settling rate k (0.000013 s^{-1}) determined from observed event influent and effluent concentrations, the desired lake water quality TP concentration (0.025 mg/L), and the previously calculated mean-annual mass rates into the lake (0.08 g/s) to determine the external load reduction required. A model concept schematic is seen in Figure C6.

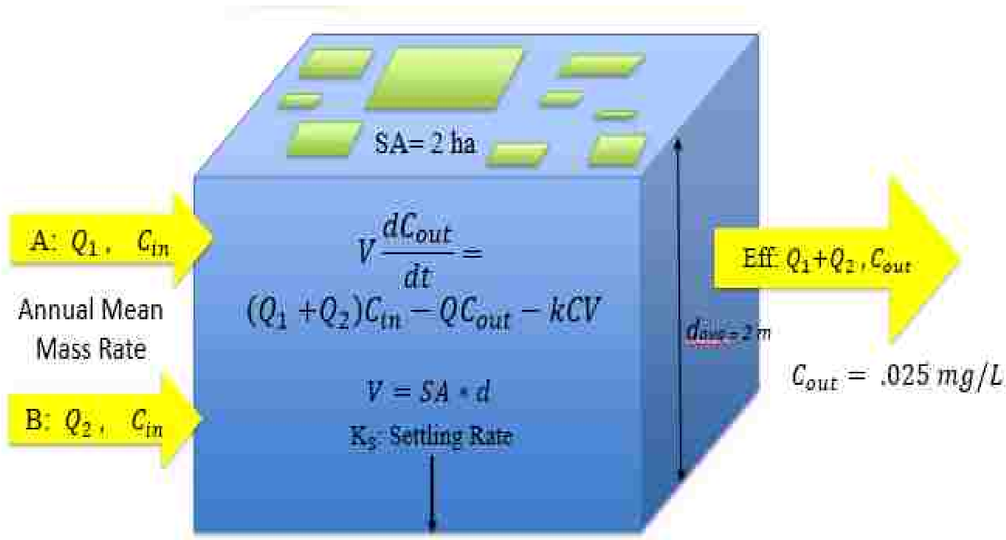


Figure C6. A conceptual diagram of the mass balance model used for Frisco Lake.

Determining Annual-Mean Mass Rates. The mean-annual mass rate used in the mass balance model were calculated by averaging the TP loading rates using only the wet-weather portions of the continuous discharge and TP concentration record. The record included discharge and concentration information at 15 second intervals. This study used approximately one year's worth of discharge and water quality data and therefore serves as an estimated mean-annual rate. Many more years of data would be required to provide a more accurate estimate of the annual expected TP loading from the catchments.

Determining Reaction Coefficient k. The settling rate, k, input was determined by running the mass balance model for three separate events with known influent and effluent TP fluxes and solving for k as the output parameter. The storm events used in the analysis included 9/17/2014, 6/13/2015, and 6/26/2015 and all had corresponding recorded lake effluent TP concentrations. The influent mass rates were determined by averaging the loading rates from the continuous discharge and concentration record during the time interval of which there was hydrographic activity during the storm event. The most conservative, in this case the smallest, observed settling rate k of the three events was 0.0000134 s^{-1}. Therefore, a k of 0.000013 s^{-1} was used in the mean-annual mass balance nutrient modeling.

Table C3. Model input and output parameters for three monitored storm events used to determine settling rate k for mean-annual mass balance loading analysis.

	Input Parameters			Output
	C_{in} Q	Q	C_{out}	k
Event Date	Avg Mass Rate (g/s)	Avg Discharge (m^3/s)	Effluent Conc. (mg/L)	Settling Rate (s^{-1})
9/17/2014	0.13	0.19	0.22	1.3E-05
6/13/2015	1.34	0.78	0.33	0.00011
6/26/2015	0.62	0.42	0.06	0.00033
Most Conservative Observed k				1.3E-05

Running the Model. The model was simplified by assuming a steady state condition, which limits the preciseness of output values due to the complex internal phosphorus loading mechanisms that will cause the reaction constant k to vary depending on seasonal conditions. Table C3 shows model input and output parameters that are based upon present lake conditions and observed loading rates. The lake and effluent concentration, $C_{out,}$ input value of 0.025 mg/L was taken from literature observing it to be the threshold concentration of impaired eutrophic lakes (Davis and Masten 2009). The lake volume was determined by multiplying the surface area of the lake determined by a geospatial analysis using ArcMap 10 (ESRI), and the average depth was based upon observation and historical knowledge. The model output mean annual mass rate was then compared with the observed mean annual mass rate to estimate a loading reduction of 39%. This estimate was used as guidance in planning upstream BMP solutions.

Table C4. Model inputs and outputs representative of present lake conditions.

INPUT		
V (SA*D)	Volume (m^3)	30,350
SA	Surface Area (m^2)	20,235
D	Mean Depth (m)	1.5
C_{out}	Lake TP Con. (mg/L)	0.025
k	P Settling Rate (s^{-1})	0.000013

OUTPUT		
QC_{in}	Mean Annual Mass Rate (g/s)	0.04

Load Comparison		
$(QC_{in})_{obs}$	Observed Mean Annual Mass Rate (g/s)	0.07
Loading Reduction for Lake Improvement		**39%**

Mass Balance Model Input Parameter Sensitivity Analysis. A sensitivity analysis was completed for each input variable to the Frisco Lake Mass Balance model. The estimated catchment mean-annual mass rates, settling rate, and mean lake depth from the annual stormwater quality monitoring results were used as input parameters in the current condition model denoted as “observed.” Then, a separate analysis for each variable was run in which a single parameter was varied to an upper and lower value from the calculated observed estimation from the field monitoring results. The upper and lower limits were determined using realistic measurement error ranges.

Table C5 shows the sensitivity analysis for the influent mass rate. The observed discharge (Q_{obs}) was the mean annual result from the continuous discharge record, a summation of flows at Channel A and B and the observed mean-annual P flux (QC_{in}) into the lake determined by summing the observed mass rates at each Channel. The upper and lower limits were +/- 20% greater or less than the observed QC_{in} value. According to Baade and Liese (2002), for small (<1 km2) catchments, error in hydrological monitoring estimates typically fall between 7 to 20%. Increasing and decreasing the discharge by 20% changed the lake P concentrations by 15% and -17% respectively.

An additional sensitivity analysis, presented in Table C6, was completed for the lake volume by varying the mean lake depth by +/- 40%. The surface area of the lake stays relatively constant; however, a lake bottom survey was not completed for this project and the mean lake depth was estimated using historical and best judgement methods. Upper and lower mean depth limits were 2.1 and 1 m where the observed depth used to run the current state model was 1.5 m. The analysis showed decreases and increases of -24% and 36% from the current condition lake water quality if the mean lake depth was increased to 2.1 or decreased to 1 m respectively.

The sensitivity analysis conducted on the settling rate k had upper and lower limits that were +/- 20% of the observed, most conservative k value chosen for the current state Frisco Lake mass balance model. Increasing the settling rate by 20% resulted in an output lake P concentration that was change by -14%. Decreasing the settling rate by 20% resulted in a higher lake concentration by 19%. Results are summarized in Table C7.

Table C5. Results of sensitivity analysis of the mean annual mass rate input parameter on the Frisco Lake Mass Balance model.

Mass Rate Sensitivity Analysis		Input		
Parameter		Q_{obs}	Q_{upper}	Q_{lower}
Discharge	Q (m^3/s)	0.098	0.1176	0.0784
Mass Rate In	QC_{in} (g/s)	0.081	0.097	0.065
Settling Rate	k (s^{-1})	0.000013	0.000013	0.000013
Volume	V (m^3)	30352.5	30352.5	30352.5
Depth	D (m)	1.5	1.5	1.5
Area	A (m^2)	20235	20235	20235
			Output	
Lake Conc.	C_{out}	0.16	0.19	0.14
Percentage Change			15	-17

Based upon the results of the three sensitivity tests, the lake P concentration is most sensitive to the potential errors associated with determining the mean lake depth, suggesting 1) it is important for the lake mean depth and resulting volume be accurately determined and 2) removing the lake sediment will not only reduce the P recycling within the lake, but provide additional benefits from the deepening of the lake. Additionally, from this analysis it can be concluded that additional methods and testing to adequately determine a representative settling coefficient may be needed to improve model accuracy.

Table C6. Results of sensitivity analysis of the mean lake depth input parameter on the Frisco Lake Mass Balance model.

Mean Lake Depth		Input		
Parameter		D_{obs}	D_{upper}	D_{lower}
Discharge	Q (m^3/s)	0.098	0.098	0.098
Mass Rate In	QC_{in} (g/s)	0.0812	0.081	0.081
Settling Rate	k (s^{-1})	0.000013	0.000013	0.000013
Volume	V (m^3)	30352.5	42493.5	20235
Depth	D (m)	1.5	2.1	1
Area	A (m^2)	20235	20235	20235
			Output	
Lake Conc.	C_{out}	0.16	0.12	0.22
Percentage Change			-24	36

Table C7. Results of sensitivity analysis of the settling rate input parameter on the Frisco Lake Mass Balance model.

Settling Rate			Input	
Parameter		k_{obs}	k_{upper}	k_{lower}
Discharge	Q (m^3/s)	0.098	0.098	0.098
Mass Rate In	QC_{in} (g/s)	0.0812	0.081	0.081
Settling Rate	k (s^{-1})	0.000013	1.56E-05	1.04E-05
Volume	V (m^3)	30352.5	30352.5	30352.5
Depth	D (m)	1.5	1.5	1.5
Area	A (m^2)	20235	20235	20235
			Output	
Lake Conc.	C_{out}	0.16	0.14	0.20
Percentage Change			-14	19

APPENDIX D

WATERSHED STORMWATER MANAGEMENT PLAN

Watershed Evaluation Methods

Overview. To reduce external TP loading into Frisco Lake and provide long-term improvements to water quality, upstream implemented BMPs within the watershed are necessary. This section provides the process behind formulating a watershed stormwater quality improvement plan. Stormwater improvement planning involved a catchment specific approach that modeled TP loading using standard Simple Method (Schueler 1987) techniques, standardized literature values for BMP stormwater retention and nutrient removal efficiencies, and observed TP concentrations at various outfalls throughout the watershed. Watershed land-use and land-cover percentages used to calculate runoff coefficients were completed using geospatial analysis software ArcMap 10 (ESRI). The calculated TP load reductions from the watershed BMP improvements were then applied to the lake mass balance model with adjusted physical parameters to account for the proposed lake dredging to estimate lake P concentrations.

Watershed and Lake Characteristics. The study watershed is located in Rolla, Phelps County, MO (37.9551°N, 91.7672°W) within the Ozark Plateau physiographic region. Its surficial geology is characterized by the outcroppings of the Jefferson City formation comprised of medium to finely crystalline, argillaceous, cherty dolomite with lenses of conglomerate and shale (Dennis 1999). Watershed topography is gently to moderately sloping with gravelly silt loam soils (NCRS 2002). This drainage area is located within the larger Dry Fork sub-basin, part of the Meramec watershed in which waters eventually drain into the Mississippi River (Blanc *et al.*, 1998). The study area is located in a humid continental zone characterized by cool to cold winters and long, hot summers. The total annual precipitation is approximately 42 inches with two thirds of the rain occurring in April through September. The average seasonal snowfall is about 17 inches (USDA, 2002).

Frisco Lake is a small, shallow man-made catchment, 0.02 km^2 in size with an average depth of 1.5 m, residing within a community park surrounded by urban residential areas. The lake was built in the 1860s by the Frisco Railroad to be used as a reservoir for watering stream locomotives. In 1982, the city of Rolla partially drained and excavated a portion of deposited sediment of the lake after severe flooding occurred.

Since then, it has remained undisturbed (Dennis 1999). It is currently listed in the EPAs 303(d) Impaired Waters for containing elevated concentrations of mercury attributed to inputs from atmospheric deposition (MDNR 2014) and experiences seasonal algal blooms, ultimately reducing its aesthetic and recreational functioning.

Contributing catchments A and B areas totaling 0.31 km^2 drain into Frisco Lake, and impervious surface land cover was categorized into parking lot or low traffic areas, roadways, or building roofs. The breakdown of each land use and cover of both catchments in terms of percent watershed area is summarized in Table D1. Geospatial analyses and mapping was completed using ArcMap 10 software (ESRI).

Figure D1. Delineated impervious surface land covers within Frisco Lake watershed using geospatial analysis software ArcMap 10 (ESRI).

Catchment A was comprised of 23% road, 19% parking lot, and 15% roof, summing to a total of 56% impervious watershed. Catchment B was comprised of 16% road, 17% parking lot, and 25% roof. The percent impervious area of each catchment was used in stormwater quality modeling as detailed in the following sections, with particular regard to the percent parking lot as it was used to determine potential impervious to pervious pavement conversion.

Table D1. Land Use/Land Cover areas in Catchments A and B

	Catchment A		Catchment B	
Land Use	Area (ha)	% of Basin	Area (ha)	% of Basin
Residential	18	80	6.6	41
Commercial	2.7	12	0	0
Institutional	1.7	8	9.4	59
Total	22	100	16	100
Land Cover	Area (ha)	% of Basin	Area (ha)	% of Basin
Road	5	23	2.6	16
Parking Lot	4.1	19	2.7	17
Building/Roof	3.2	15	4	25
Total Impervious	12.3	56	9.3	58

Green Infrastructure Planning. Modeled catchment load improvements were to be achieved using GI implements. Proposed pervious pavement and bioretention facility plans are shown in Figure D1. The inclusion of green roof implements were considered, but not included in the watershed improvement plan due to the complexity and potential nutrient addition to stormwater flows from the green roof media. Stormwater quality improvements from the proposed implements are modeled in the following section.

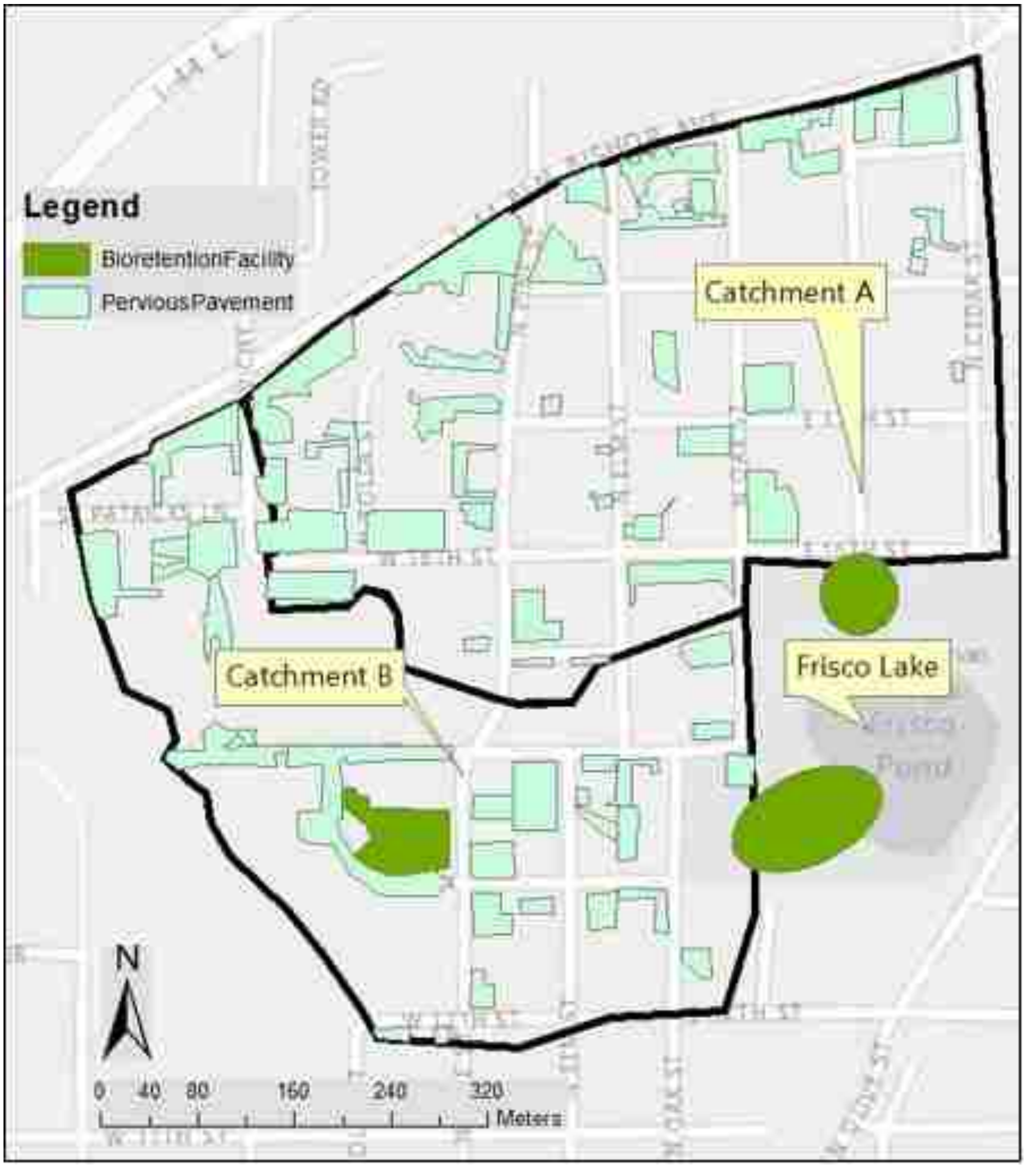


Figure D2. Watershed improvement plan including proposed bioretention and pervious pavements implements.

The proposed watershed plan includes one bioretention facility at the effluent of Catchment A, and two bioretention facilities located in Catchment B, one at the campus effluent and one at the Catchment B effluent water quality sampling locations. Redundancy and dispersal of facilities throughout drainage area increases stormwater improvement capabilities (UACDC 2010), and were therefore utilized in this plan. For Catchment B, one upstream bioretention facility is located at a campus drainage outfall in a low-lying, open space detention area in between the Physics and Inter-Disciplinary Engineering buildings on the S&T campus. Retrofitting a bioretention facility in this location was considered feasible as the area is already serves as a stormwater retention area when flows exceed infrastructure stormwater conveyance capacity. The other areas chosen for placement of a bioretention facilities is located at Catchment A and B outfalls at the heads of Channels A and B. The noted areas are currently open, widely unutilized, unaesthetically pleasing as a drainage channel in a city park. These areas have the potential to benefit, aesthetically, ecologically, and functionally by retrofitting a green infrastructure implement. The most downstream portions of the proposed bioretention facilities at the lake inlet channels lie within the lake boundaries and are typically permanently inundated with water, posing potential for a constructed wetland. The specific design and construction of the GI implements are beyond the scope of this project. Figure D3 shows the proposed bioretention facility location at the campus effluent in Catchment B.

Figure D3. Detention area at the proposed bioretention facility location at the campus outfall in Catchment B.

Watershed impervious parking lot and low traffic areas, denoted as “Pervious Pavement,” were also delineated in Figure D2. Catchment B is comprised of 17% parking lot, whereas Catchment A is comprised of 19% as determined by a geospatial analysis using ArcMap 10 software (ESRI). The parking lots are located in intuitional, commercial, and residential areas and feasibility of implementation would be dependent upon community wide interest. Legislative action promoting economic incentives for implementing stormwater BMPs could increase residential and commercial interest. Institutions could see additional benefits as providing potential educational opportunities and self-promotion by engaging in environmentally progressive and responsible practices. Implemented pervious pavement at Thomas Jefferson Hall on the Missouri S&T campus exists just outside the study area catchments. In this pervious pavement design, roughly 10% of the parking lot area is converted to pervious land area that treats and infiltrates runoff flows from the impervious parking lot. More recently the campus implemented swatches of pervious pavers outside Butler Carlton Hall under the bike racks, suggesting that the campus is interested and willing to construct GI on campus.

Bioretention Impacts on Stormwater Quality. Nutrient removal capabilities of bioretention facilities used in watershed stormwater quality modeling were based upon accepted literature values from urban watershed planning models. For bioretention facilities, nutrients can leach from the facility media causing potential increases or decreases in nutrient concentrations to values of 0.05 to 0.18 mg/L (Davis *et al.*, 2009) regardless of influent TP concentrations. Stormwater quality (TP concentrations) were sampled at each proposed bioretention location to properly estimate the influent concentration into the bioretention facility. Stormwater quality improvements from proposed bioretention facilities were modeled using observed influent TP concentrations and expected effluent concentrations, rather than reported, often variable removal efficiencies. Observed average TP concentrations were collected at various outfalls within the study watershed are summarized in Table D2. Standard deviations from the mean were also determined and are presented. TN data was assessed to determine TP as the limiting nutrient leading to the algal blooms in Frisco Lake. Stormwater concentrations at the campus and Channel B outfalls within Catchment B and at Channel A in Catchment A ranged from 1.07 mg/L to 1.40 mg/L, values greater than national averages (NRC 2008), making them viable candidates for bioretention mitigation, as effluent concentrations can be expected to be decreased to 0.18 mg/L or less.

Table D2. Observed mean TP concentrations at water quality monitoring sites.

	Mean TP Concentration (mg/L)			
	Channel		Effluent	
	A	B	Campus	Lake
First Flush	1.07 +/- 0.87	0.94 +/- 0.48	1.40 +/- 0.31	N/A
Peak Flow	0.90 +/- 0.47	0.88 +/- 0.63	0.55 +/- 0.27	N/A
Receding Flow	0.52 +/- 0.29	0.50 +/- 0.37	N/A	0.18 +/- 0.11

In addition to reducing effluent concentrations, literature has shown bioretention facilities to reduce annual runoff volumes by 40 to 90% (Davis *et al.,* 2009). For the bioretention facilities in Catchment B, 80% stormwater retention of was assumed. A better removal efficiency can be expected from the biofiltration plan because having an upstream and downstream bioretention facility will mitigate the peak runoff flows that cause the reductions in stormwater retention and subsequent decreases in removal efficiency. A schematic diagramming expected stormwater quality and quantity improvements at the campus outfall is shown in Figure D4.

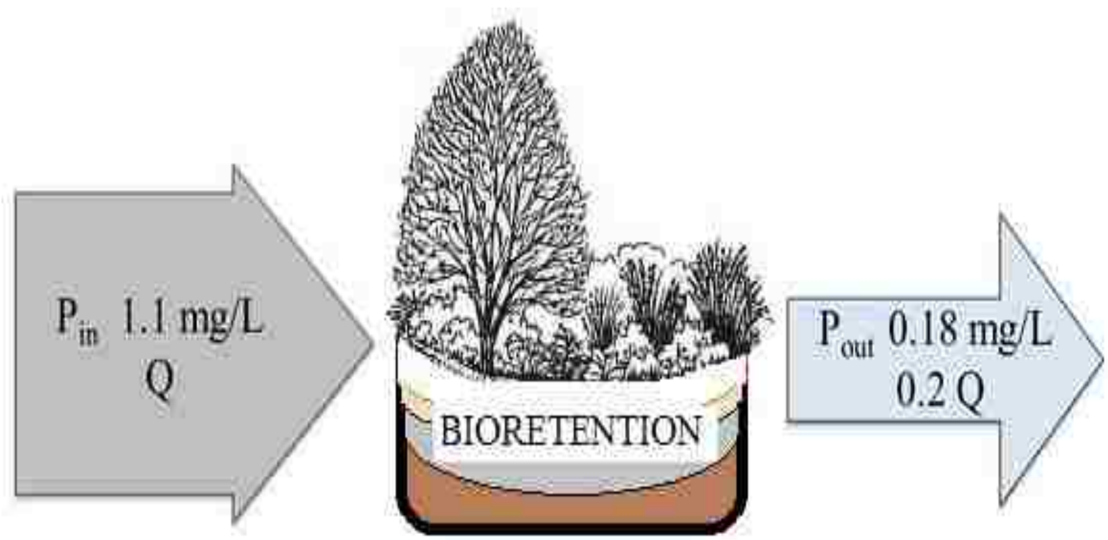


Figure D4. Schematic showing influent and effluent stormwater quality and quantity at proposed upstream bioretention facility at the campus outfall.

The reduced nutrient load from Catchment B with the proposed GI plan was estimated by using the mean-annual observed influent TP concentration at Channel B, 0.43 mg/L, and a conservative 0.18 mg/L as the effluent concentration. Effluent concentrations of 0.18 mg/L were applied to the continuous wet-weather discharge record determined and discussed in Appendices B and C to calculate a new, reduced load mass rate. To account for the stormwater retention, the mean annual flow rate was reduced by 80%. Applying a runoff retention of 80% to the reduced nutrient flux from Catchment B, improvements of 92% reduction to the mean-annual mass rate were estimated.

Similar methods were used to determine the nutrient load reductions from Catchment A after bioretention implementation, and an additional analysis was completed considering the conversion of impervious pavements to pervious. Considering bioretention alone, stormwater quality at the Catchment A outfall is estimated to be improved to an average concentration of 0.18 mg/L from its observed mean-annual concentration of 0.43 mg/L. The biorention facility for Catchment A is located at the downstream end of the drainage area, without an upstream facility, so an estimated runoff retention rate of 20% was used in analysis using the same methods used for Catchment B.

Pervious Pavement Impacts on Stormwater Quality. The impacts of proposed pervious pavements on nutrient loading were based upon a common watershed modeling technique from Schueler (1987) termed the Simple Method due to its limited parameters. The method Equations D1 and D2 are listed:

$$L = P * f * R * C * A * 0.2267 \quad \text{(D1)}$$

$$R = 0.05 + 0.009 * I \quad \text{(D2)}$$

where L is the pollutant load (lb), P is the precipitation (in), f is a correction factor for storms with no runoff, typically 0.9, R is the runoff coefficient of the watershed, C is the event mean concentration (mg/L), A is the catchment area (ac), and I is percent impervious. The conversion of impervious parking lots to permeable pavements will decrease the percent impervious by the percentage of land area converted. In the proposed watershed plan, a possible 19% of Catchment A could be converted to pervious pavements resulting in a watershed percent impervious reduction from 56 to 37. Holding all other parameters equal, the reduced I value will reduce the watershed runoff coefficient and resulting nutrient loads. Using Equations D1 and D2, expected TP loads from Catchment A are estimated to decrease by 31%.

Lake Water Quality Modeling. The same mass balance model used in Appendix C was used to model the reduced loading impacts of the proposed lake remediation and GI implementation plan on the lake TP concentration. The influent TP concentrations and discharge input parameters for the model were determined by the prior analyses. Improved mean-annual outlet concentrations and discharge estimates were 0.18 mg/L and 0.035 m^3/s respectively for Catchment A, and, for Catchment B, 0.18 mg/L and 0.011 m^3/s, respectively, for mean-annual total discharge of 0.05 m^3/s at a mass flux of 0.001 g/s into Frisco Lake as presented in Table D3. The sum of the resulting mass rates at channels A and B were used as the total mean-annual discharge and mass rate input parameters in the mass balance model.

Table D3. Estimated mean-annual mass rates, inlet concentrations, and discharges from channels after watershed improvements.

	Mean-Annual		
	Q (m^3/s)	Influent Concentration (mg/L)	Mass-rate (g/s)
Channel A	0.044	0.18	0.008
Channel B	0.054	0.18	0.010
Total	0.098		0.018

In addition to adjusted nutrient loading rates, the assumed average lake depth after dredging was adjusted to 3 m, resulting in a lake volume of 60,705 m^3. The settling coefficient was the same used in prior analysis for consistency; however, it is likely to be improved with less internal P loading expected after dredging of contaminated sediment. Input parameters and the model outputs can be summarized in Table D4. With the proposed watershed implements that have performance rates as planned, lake water quality can be expected to be below the eutrophication threshold into the upper oligotrophic range (Davis and Masten 2009).

Table D4. Watershed improvement plan model inputs and output.

INPUT		
V (SA*D)	Volume	60,705 m^3
SA	Surface Area	20,235 m^2
D	Mean Depth	3 m
Q	Mean Annual Discharge	0.05 m^3/s
QC_{in}	Mean Annual Mass Rate	0.01 g/s
k	P Settling Rate	0.000013 s^{-1}

OUTPUT		
C_{out}	Lake TP Con.	0.01 mg/L

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