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EVALUATION OF CATCHMENT-SCALE STORMWATER RUNOFF MANAGEMENT ON FIRST-FLUSH WATER QUALITY AND STORM DISCHARGE QUANTITY

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EVALUATION OF CATCHMENT-SCALE STORMWATER RUNOFF MANAGEMENT ON FIRST-FLUSH WATER QUALITY AND STORM DISCHARGE QUANTITY

A THESIS APPROVED FOR THE
SCHOOL OF CIVIL ENGINEERING AND ENVIRONMENTAL SCIENCE

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Abstract

Urbanization and resulting imperviousness has caused increased pollutant loadings into receiving water bodies. One of the main sources of pollutant loads is stormwater runoff. The purpose of this study was to investigate the capabilities of low impact development (LID) best management practices (BMPs) and traditional retention ponds to address pollutant loads in first-flush stormwater runoff. Two studies were completed. First, a paired watershed study, in which one basin included a suite of LID BMPs and the other included traditional curb and gutter stormwater controls, was monitored for hydrologic and water quality data. The second study examined the water quality improvement capabilities of four nearby retention ponds. LID BMPs decreased total runoff volume and peak discharge rates for most sampled events. Water quality data showed differences in all pollutants except for dissolved reactive phosphorous. The retention ponds showed decreased nutrient concentrations compared to data from the national stormwater database. The combination of LID BMPs and retention ponds used in series would likely decrease urban stormwater pollutant loads.

1 Introduction

1.1 Background

As many countries around the world become more urbanized, rural populations are moving to the cities. It is estimated that nearly two-thirds of the world's population will live in urban areas by the year 2030 (Dahu et al. 2008). The large population shift from rural communities to urban areas causes vast alterations to the landscape and changes in ecosystem functions. Urbanization has environmental impacts ranging in scale from small localized changes to larger global impacts. Urbanization substantially affects biogeochemistry at a localized scale to climate at a larger scale (Grimm et al. 2008). Urban areas are also a leading contributor to anthropogenic carbon dioxide emissions, a driver of climate change, due to heating and cooling of buildings, industrial processes, and transportation of people and goods (Grimmond 2007). While urbanization causes environmental degradation in a variety of ways, one of the biggest issues is converting a permeable, vegetated landscape to an impervious one covered by roads, parking lots, buildings, and sidewalks. This conversion reduces infiltration of stormwater into and evapotranspiration from soils, thereby causing a larger volume of water from storm events to become runoff (Walsh et al. 2012). Conversion from natural ground cover to as little as 35 to 50% impervious surface area reduces evapotranspiration by 12.5%, shallow infiltration by 20%, deep infiltration by 40%, and increases runoff by 200% (Figure 1; EPA 2000)

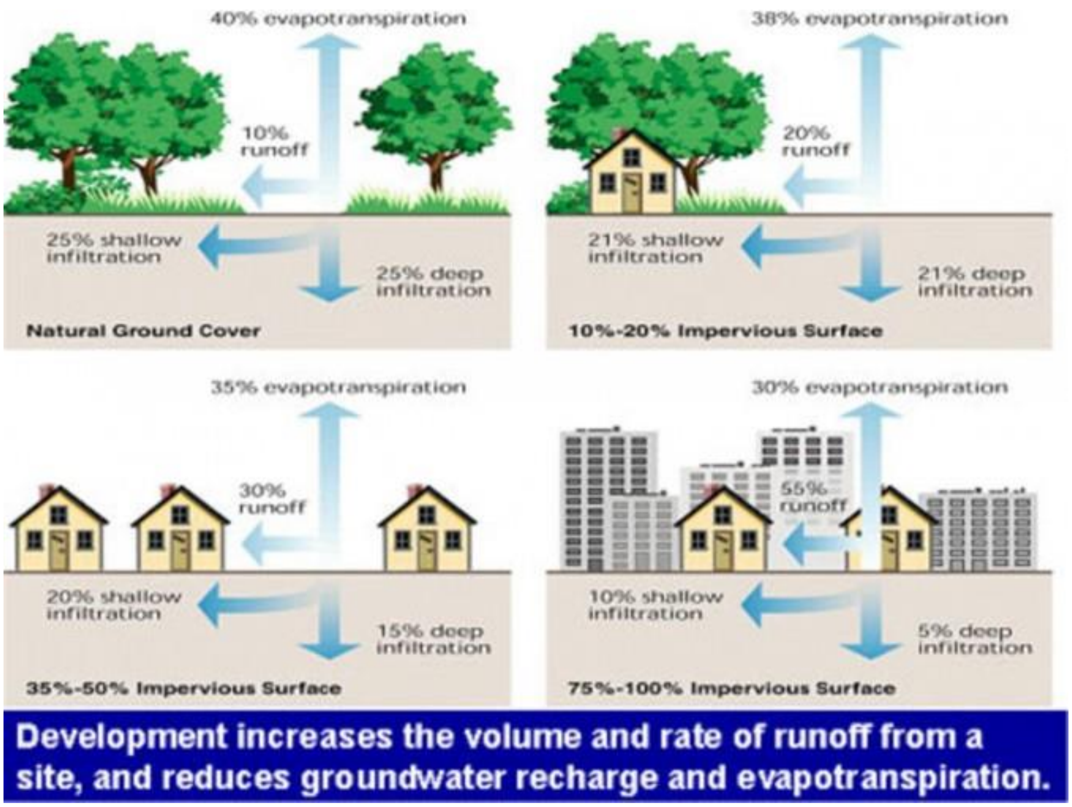

Figure 1: Changes in stormwater outcome based on changes in urbanization and the associated increased imperviousness (from EPA 2000)

Besides significantly increasing total runoff volume, higher percentages of impervious surfaces also cause peak flows to increase by a factor of two to more than 10, which has dramatic effects on the hydrology of receiving waters, especially streams (Roesner et al. 2001). Streams experience quick, large flushes, elevated concentrations of nutrients, sediment, and metals, altered channel morphology, and reduced biotic richness, abundance and diversity (Walsh et al. 2005). Impact of urbanization goes beyond hydrologic alterations with significant degradation of stormwater runoff quality by increasing concentrations of solids, oxygen-demanding matter, trace metals, nutrients, and various bacteria and pathogens (Chen and Adams 2006). Changes in stormwater flow regime and water quality have unquestionable effects on lakes, streams, and drinking water sources. Rapidly increasing urbanization combined with more attention to environmental degradation of waterways has raised the urgency in properly managing stormwater in a cost effective, environmentally responsible way.

1.2 Stormwater Runoff

1.2.1 Physical Impacts

Urbanization changes the destination of a large percentage of stormwater, diverting it from infiltration, groundwater recharge, and evapotranspiration to runoff directed into channels and streams eventually reaching rivers and lakes. Simply converting an area into 13% to 21% impervious surface area causes significant hydrological changes. Impervious surfaces cause a decrease in sustained flows while making the flow regime flashier with larger, more frequent pulse events in which peak discharge is much greater than historical means (Jennings and Jarnagin 2002). Another hydrologic change caused by imperviousness is increased annual runoff depth. Over a 30-year period converting an area from 10% imperviousness to around 30% generated an increase in annual runoff depth of 77%, while peak flows increased by 32% (Olivera and DeFee 2007). Besides producing a greater total volume of runoff, urbanization also increases velocity of flow through the conversion of rough natural surfaces into smooth urban surfaces (Jacobson 2011). Increased volume and speed of stormwater runoff can potentially alter downstream environments through increasing erosion and flood magnitudes while changing stream morphology and riparian vegetation (White and Greer 2006; Jacobson 2011; Whitney et al 2015).

One of the most noticeable impacts of urbanization is changes in the flood regime. Flooding in urban areas has widespread effects from property loss and casualties to drinking water pollution or disease breakouts (Chen et al. 2015). Increases in stormwater runoff increase peak discharges and flood magnitudes, however, the increase in magnitude is greater for more frequent floods than floods with longer return intervals (Hirsh et al. 1990; Kibler et al. 1981). Essentially urbanization increases frequency and strength of smaller flood events while also increasing the magnitude of larger more devastating flood events (Kibler et al. 1981). A predevelopment 10-year flood corresponds to a one to four-year flood once an area becomes developed (Morscrip and Montgomery 1997).

While flooding has one of the most significant impacts on people, other impacts have substantial ecological consequences. Developing an area drastically increases the amount of sediment transported by stormwater runoff into waterways (Pappas et al. 2008; Corbett et al. 1997; Charters et al. 2015; Ryan 1991). Increased sediment has detrimental effects on plants, invertebrates, and fish while also increasing turbidity, causing changes in algae populations and aesthetics of streams (Ryan 1991). Most of the sediment enters the streams occurs during construction, while after construction increased runoff coupled with a decrease in sediment loads promote bank erosion and channel widening (Wolman 1967). Stream channel sediment inputs go through three phases, non-urban, transition, and urban, with associated characteristics for each (Vietz et al. 2016). During the transition phase, an increase of sediment bedload inputs from catchment and channel sources occurs, the channel experiences enlargement and an influx of sediment, and there is an increase in transport capacity from stormwater flow (Vietz et al. 2016). After a stream has had major connections to stormwater for more than 10-years, sediment input decrease through reduction in channel sources and finished development, channel enlargement continues, and sediment removal begins while transportation capacity further increases due to stormwater flow and reduced floodplain engagement (Vietz et al. 2016). Changes in sediment influxes coupled with greater transport capacity lead to a variety of changes in stream geomorphology. While many variables can affect the degree of channel enlargement, semiarid streams experience a positive relationship of about 1.2:1 for channel enlargement to increased sediment transport capacity (Hawley and Bledsoe 2013). Changes in sediment transport capacity increase impacts to streams in other ways besides just channel enlargement. Over 40 urban/suburban northern Kentucky streams with an average drainage area of 22.42 km^2 and imperviousness of 13.7% experienced channel down cutting and widening, shortening of riffles, pools becoming deeper and longer, and changes in bed material to a more coarse and homogenous material (Hawley et al. 2013). Urbanization changes the fluvial geomorphology of many streams, which in turn alters the ecology and ecosystem services of streams.

1.2.2 Chemical Impacts

Urbanization caused-stormwater runoff has many physical impacts to streams, rivers, and lakes but greater degradation might occur due to biogeochemical changes. Urbanization has different land use types such as commercial, industrial, or residential with which different pollutant species will be most prevalent. Residential areas have the highest variation in total phosphorous (TP) and commercial land use has the highest variation in total organic carbon (TOC), with both generating greater nutrient and organic carbon loads, while industrial areas have the highest variation in total nitrogen (TN) and generate higher total suspended solid (TSS) loads (Liu et al. 2013). The variation in runoff characteristics not only changes between land use type but also between storm event type. The event mean concentrations (EMC) have wide distributions depending on total rainfall, rainfall intensity, and dilution during the event (Lou et al. 2009). The wide degree of variation is further seen in a study of the Twin Cities metropolitan with 520 stormwater events measured for TSS obtaining a minimum of 2 mg/L, a maximum of 3577 mg/L, a mean of 184 mg/L, and median of 88 mg/L (Brezonik and Stadelmann 2002). While extensive variation between land use and storm event type among other factors exists, it is still important to understand degradation potential.

Sediment loads in stormwater runoff are a major source of potential degradation. TSS is an important water quality indicator because it is common in urban stormwater, provides a mode of transportation for other contaminants, and can cause ecological damage (McCarthy et al. 2012; Sun; Djukic et al. 2016; Duncan 1999). Sediment from runoff can come from many sources like automobile brake wear, surface material degradation, soil erosion and atmospheric deposition and thus is heterogeneous in composition (Charters et al. 2015). Different sediment characteristics can have variable ecological effects. Sediment high in organic content can undergo anaerobic breakdown causing depletion of oxygen levels, and increased turbidity can cause a decrease in photosynthetic production (Ryan 1991). A large source of potential degradation comes from the ability of sediment to transport other pollutants. As the grain size of sediment decreases, the ability to transport pollutants increases. Sediment with < 63 μm grain size contributed an average of 48.5% for Mn, Cu, Zn, Pb, Cr, Ni, and Co into the Arkansas River even though it only contributed 37% of the sediment (Horowitz 2008). Another study showed correlation between Al, Fe, Pb, and Mn and particles between 0.45 and 150 μm, which represents a majority of the particles in urban stormwater (Herngren et al. 2005). Besides metals, sediment also transports oxygen, nitrogen and sulfur containing polycyclic hydrocarbons (PAHs), nitrogen and phosphorous species, and organic matter among others (Lou et al. 2009; Witter and Nguyen 2015). The transport of different pollutants by sediment in streams has biological consequences all the way up the food chain. While turbidity is not the only water quality parameter correlated to macroinvertebrate communities in streams with a narrow range of chemical results, macroinvertebrates in streams with a wider range show a strong response to organic pollution (Thorpe and Lloyd 1999; Lawler et al. 2006). Eutrophication and metal pollution have a stronger influence on benthic diatoms than macroinvertebrates, while fish are not are as susceptible to stream chemistry but are influenced more so by habitat availability (Mangadze et al. 2016; Thorpe and Lloyd 1999). Sediment is such an important water quality indicator because of the vast ways it can influence stream communities from increasing turbidity to transporting nutrients and metals to changing the habitat structure through sedimentation or erosion.

Another source of potential degradation to urban streams from urban stormwater runoff is elevated nutrient concentrations. Nitrogen and phosphorous species are of the most concern because of their roles in eutrophication. Concentrations in stormwater runoff are going to be dependent on the type of development, but for residential areas in Seattle, Washington, total nitrogen (TN) values ranged from 1.61 mg/L to 2.32 mg/L and total phosphorus (TP) ranged from 0.19 mg/L to 0.52 mg/L, while runoff from undeveloped land showed concentrations of 1.15 mg/L of TN and 0.055 mg/L of TP (Harper and Baker 2007). For a city center catchment with a total impervious area of 62%, an additional 20 g/ha of TN and about 1 g/ha of TP can be added through stormwater runoff (Valtanen et al. 2015). Medium density residential areas like subdivisions add 15.5 kg/ha/yr of TN and 1.3 kg/ha/yr of TP compared to 6.3 kg/ha/yr and 0.5 kg/ha/yr for undeveloped areas, respectively (Carey et al. 2013). These influxes inevitably end up downstream and only become more magnified. Many urban streams can have as high as 95% more TP and 122% more soluble reactive phosphorous (SRP) than their forested counterparts (Brett et al. 2005). Nitrogen is no different, having more than doubled in the Mississippi River since 1965 and showing a 3- to 10-fold increase since the 1900s in major rivers of the northeast US (Vitousek and Farrington 1997). Agricultural inputs are a potential source for increased nitrogen, but population density is highly correlated with TN suggesting urbanization still plays a key role in the substantial accumulation (Vitousek and Farrington 1997). Additions of nitrogen and phosphorous have potentially negative consequences for water quality as well the integrity of ecosystems. Depending on the ecosystem, both phosphorous and nitrogen can be the limiting nutrient, so inputs of either can be detrimental. Nitrogen is often the limiting nutrient in temperate zone estuaries and seas, while phosphorous is the limiting nutrient in most temperate-zone lakes and streams (Vitousek and Farrington 1997; Schindler 1997). An increase of either limiting nutrient can cause increased primary production and eventually eutrophication. Eutrophication of aquatic systems can lead to large algal blooms, which are problematic because they can limit light to submerged aquatic vegetation, reduce water transparency, and produce hypoxic or anoxic conditions which then impact fish populations, or release toxins (Carey et al. 2013). No matter if toxins are released or not, harmful algal blooms have substantial impacts on water quality, co-occurring organisms, and food web dynamics. Some of the impacts include mass mortalities of fish and shellfish, human illness or death from exposure by inhalation, water contact, or consumption of seafood, illness or death to marine mammals and seabirds, and lastly alteration of habitats and trophic structure (Anderson et al. 2002). Non-point source nutrient additions have vast ecological impacts and present a major problem, which will only continue to get worse.

Eutrophication of lakes and rivers is an easily noticeable environmental impact while the impact of metals and other pollutants in stormwater runoff might not be as obvious. As discussed earlier, many metals are sediment transported. Pb, Fe, Al, and Mn show a strong correlation to TSS while Zn and Cu have a lesser correlation to dissolved organic carbon (DOC) and a slight correlation exists between Cr and total organic carbon (TOC) (Herngren et al. 2005). The origin of trace metals in runoff varies a great deal from roads, roofs, parking lots, and even recreational land, but the major contributors are tires, automobile exhaust, parking dust, and building materials (Gnecco et al. 2005; Reddy and Dashtgheibi 2014). Metal pollution causes many ecological and biological problems. Toxicity tests for runoff from urban highways using freshwater species (*Ceriodaphnia dubia, Pimephales promelas*, and *Pseudokircheriella subcapitatum*) and marine species (*Strongylocentrotus purpuratus* and *Photobacterium phosphoreum*) exhibited toxicity in all storm events, with 90% of the observed toxicity attributable to dissolved Cu and Zn (Kayhanian et al. 2008). In stream ecosystems, As and Cr had significant negative impacts on macroinvertebrates, while As, Cr, and Ni negatively affected diatom communities. The other environmental concern associated with metals is bioaccumulation. Hg, Pb, and Cd are all metal species that bioaccumulate in fish and pose potential health issues for animals higher up the food chain, including humans. Metals also present a problem because while Cu has a positive correlation between bioaccumulation and concentration in water, other metal species, such as, Cr and Hg, can be below detection limits in water but still have high concentrations in fish (Roig et al. 2016).

1.2.3 Stormwater First Flush

Urbanization affects stormwater runoff and downstream waterways in many diverse ways, from altering the physical environment to changing water chemistry and community structure. Understanding these changes and potential solutions is important to prevent continued destruction of these ecosystems. Targeting stormwater runoff improvements in specific, focused ways allows greater efficiency and effectiveness than trying to solve all problems at once. One potential target is the first flush of stormwater runoff. The first flush has many definitions but generally is considered the phenomenon that a majority of pollutant loading occurring during the first portion of a rainfall event (Qin et al. 2016). The definition of what constitutes the majority of pollution and beginning of event has many interpretations. Some state 80% of the pollution load in first 30% of runoff volume (Saget et al. 1995), others suggest it is the pollution load in the first 25% of runoff volume (Vorreiter and Hickey 1994) or defining it in a more general way as a pollutant mass cumulative curve that is above the runoff volume curve (Sansalone and Buchberger 1997). In understanding the first flush, it is important to understand the variation in definitions as well as influences in predictability (Deletic 1997). The first flush has a wide range of variation with a greater magnitude for some pollutants (e.g., TSS) than others, greater prominence for smaller watershed areas, and surprisingly little correlation with antecedent dry weather period (Lee et al. 2002). The first flush from copper roofs was impacted more by roof aspect than any weather parameters such as rain depth, rain intensity, and antecedent dry period (Athanasiadis et al. 2010). Although different characteristics influence what constitutes a first flush at each individual site (Deletic 1997), initial flows contribute substantially to the first flush for TSS, TN, TP with concentrations dramatically dropping after the first sample. Parking lot runoff contributes but starts at a lower concentration and the decrease is not as great; roofs contribute even less but for TN only (Wei et al. 2010). In one study, the first 30% of runoff volume (FF30) contributed 52%-72% of TSS, 40%-50% of TN, and 45%-63% of TP (Li et al. 2007). No matter what volume or pollutant load is classified as a first flush, by targeting runoff during the early part of storms, a higher probability of reducing pollutant loads exist than targeting middle or later stormwater runoff.

1.3 Stormwater Management Practices

1.3.1 Background

The beginning of stormwater management grew from health and sanitation issues. After the Industrial Revolution, urban areas saw accelerated growth and higher population density bringing increased exposure to sewage-related sanitation and health issues (Miguez et al 2012). The rise in population density also meant more people became more susceptible to localized flooding which only worsened sanitation problems by spreading contaminated wastes (Miguez et al 2012). This led to water management practices based on water quantity control through the collection and conveyance of stormwater and wastewater away from urban areas and into nearby waterways (Zhou 2014). Although conveying water was effective, it did not solve all urban flooding problems, since transferring water as quickly as possible to the nearest waterway just meant transferring the problem downstream (Miguez et al 2012). Conventional stormwater management practices, centered on drainage-efficiency, were quickly outgrown. Continued urban development meant more impervious surfaces and runoff generation requiring frequent infrastructure investments to maintain adequate removal capacity (Barbosa et al. 2012; Miguez et al. 2012). Stormwater discharged into waterways ended up altering whole downstream river systems and, with this realization, a shift from conveyance to retention, detention, and recharge occurred (Burns et al. 2012; Niemzynowicz 1999). Stormwater control measures (SCMs) form a buffer between urban imperviousness and waterways by providing storage for a portion of the runoff volume while also attenuating the peak hydrograph (Goonetilleke et al. 2005; McPhillips and Walter 2015). By storing part of the runoff volume and smoothing out the hydrograph, SCMs return hydrologic patterns closer to natural conditions. Stormwater ponds are just one of many SCMs intended to temporarily detain stormwater, but also serve the additional role as water quality best management practices (BMPs) (McPhillips and Walter 2015). The three classifications of stormwater ponds are dry ponds, wet detention ponds, and wet retention ponds. Retention ponds have a large permanent storage, wet detention ponds drain almost completely except a small permanent pool, and dry ponds drain completely between events (Tixier et al. 2011).

1.3.2 Retention ponds

Retention ponds were originally designed as stormwater storage to prevent flooding and diminish peak discharge. Generally, design requirements were to control post-development design storms, typically two- or 10-year events, to pre-development levels, while also safely being able to withstand 100-year storm events (EPA 1999). While retention ponds were designed for stormwater control, their functionality expanded to include the additional role of water quality BMP (McPhillips and Walter 2015; Tixier et al. 2011). Retention ponds have high removal efficiencies for sediment and bacteria, moderate to high rates for nutrients and metals, and moderate organic matter and oil and grease removal efficiencies (Schueler 1987). Retention ponds improve water quality predominantly through sedimentation, so sediment and pollutants that bind to particles have higher removal efficiencies while dissolved nutrients and other pollutants not readily treated by sorption or filtration have lower removal efficiencies (Stanley 1996; Collings et al. 2010). Gravitational or algal settling predominantly removes sediment while wetland plant uptake or bacterial decomposition contribute to the removal of nutrients and dissolved species (Driscoll 1983). A study completed by USGS (1985) showed retention ponds effectively decreased suspended solids concentrations by 65%, suspended lead by 41%, suspended zinc by 37%, suspended nitrogen by 17% and suspended phosphorous by 21% (Martin and Smoot 1985). While nutrients are not the primary focus of retention pond design, creating a subsurface saturated zone promotes microbial-mediated transformation of nitrate increasing denitrification potential (McPhillips and Walter 2015).

Comparing two retention ponds draining a 57% impervious basin, one designed for water-quality and another designed solely for water quantity, showed both had effective removal efficiencies for TSS, TP, SRP, and metals (Comings and Horner 1998). The designed water-quality pond had slightly better removal rates for all constituents, but dramatically greater rates for SRP with a 62% reduction versus only 3% in the water quantity design (Comings and Horner 1998). Differences in efficiencies could potentially be attributable to the designed water-quality pond having a pond surface area of 5% in relation to its basin while the other pond has 1% pond surface area to basin ratio and a detention time seven times less (Comings and Horner 1998). When comparing ponds of different sizes and different characteristics it is important be able to normalize and examine relative volume(ratio of pond volume to impervious watershed area) as one potential tool (Walker 1987).

Retention ponds do have a potential draw back, however. Retention ponds are sinks for nutrients preventing them from affecting downstream environments, but the enriched nutrient concentrations can create an environment promoting biological growth. The increased microbial metabolism has the potential to diminish water quality downstream (William et al. 2013). For example, coastal subdivision ponds had eutrophic levels of chlorophyll and phosphorous during all seasons and prevalent cyanobacteria blooms during the summer months (Serrano and Delorenzo 2008). The eutrophic levels of these ponds, along with potential toxins from cyanobacteria blooms, could be detrimental to the downstream ecosystems if the constituents were to exit in the effluent.

1.3.3 Low Impact Development Best Management Practices

1.3.3.1 **Background**

The first transition in stormwater management attempted to return the hydrologic regime of urban areas and downstream waterways to pre-development conditions through water storage and controlled release of discharges (Niemczynowicz 1999). The next transition occurred during the 1980s and 1990s when stormwater quality and its substantial pollution became a concern instead of solely water quantity (Niemczynowicz 1999; Burns et al. 2012). A range of new concepts grew from this transition including new BMPs, Low Impact Development (LID), Sustainable Urban Drainage Systems

(SUDS), and Water Sensitive Urban Design (WSUDS) (Miguez et al. 2012). The emphasis for these new concepts was on mimicking ecological systems and removing pollutants through natural processes (Miguez et al. 2012; Barbosa et al. 2012; Burns et al. 2012; Niemczynowicz 1999). LID uses design strategies to maintain or replicate the pre-development hydrologic regime and functionality in order to mitigate environmental impacts (Coffman 2000; EPA 2000). LID design strategies move down the watershed starting with source control at the origin of runoff generation from impervious surfaces then follow runoff away from buildings and finally reach the mainstream network incorporating different LID applications at each stage. LID techniques reduce runoff from localized source areas using rain barrels, green roofs, and permeable pavement. Once the runoff has been generated, LID techniques slow and filter overland water runoff, sediment, and pollution before it reaches the stream network through grassed swales and rain gardens. Lastly, restored or protected riparian buffers slow and filter runoff in or adjacent to the mainstream network (Martin-Mikle et al. 2015). LID employs a holistic approach that incorporates multiple small-scale controls at the source rather than one giant end of pipe control (EPA 2000).

LID design strategies target specific areas in the hydrological cycle while also using different functional processes to remove or degrade pollutants in stormwater runoff. Impervious surfaces in developed areas generate large amounts of runoff so implementing LID practices at these sources can have compounding effects. Green roofs, permeable pavement, and rain barrels are LID practices implemented at the source of runoff generation, immediately having an impact on hydrology and water quality. Rain barrels alter localized hydrology by disconnecting impervious surfaces of the roof and driveway (Coffman 2000). Rain barrels are inexpensive, generally aesthetically pleasing retention devices equipped with a filtration screen at the water entry point, spigot at the bottom, and an overflow outlet for large storm events (Coffman 2000). Gutters and downspouts direct runoff from the roof to the barrel where it is stored for later use. The storage capacity is dependent on the size of the barrel, but one 42-gallon rain barrel can store 0.5 in of runoff from a 133 sq. ft. roof surface (Coffman 2000).

Green roofs are another LID practice implemented at the source of runoff generation. While rain barrels have limited barriers to implementation and little maintenance, green roofs have much more extensive implementation barriers due to material costs, structural requirements, knowledge required, and difficult maintenance. Green roofs, which consist of a vegetative layer, growing media, a geotextile layer, and a synthetic drain layer, are effective at reducing stormwater runoff by decreasing the percentage of impervious surface (EPA 2000). The two types are extensive roofs which have dense, low growing, drought resistant vegetation and are found on family homes and residential buildings, and intensive roofs commonly found on commercial buildings with grasses, flowers, shrubs, trees, drainage and irrigation systems. Green roofs are effective for runoff retention, frequently achieving a 30% to 70% reduction, but are limited by holding capacity causing their efficiency to be totally rainfall dependent. Once reaching a maximum holding capacity, stormwater becomes runoff, causing the percentage of total rainfall retained to decrease. Green roofs have had mixed water quality improvement results varying from no significant nutrient retention to becoming a source of nitrate, phosphorous and metals (Abiablame et al. 2012).

Another LID practice to reduce runoff generation at the source is permeable pavement. Permeable pavement replaces impervious surfaces of driveways, roads, or parking lots with a pervious surface, reducing the amount of runoff generation while also allowing slow infiltration of surface runoff into the subsoil (Ahiablame et al. 2012; EPA 2000). Permeable pavement includes four different types: block pavers, grid systems, porous asphalts, and porous concretes (Dietz 2007). The slow infiltration of surface runoff into subsoil allows for high removal rates of sediment (80%), phosphorous (60%), nitrogen (80%), trace metals (39%), and organic matter (41%) while also contributing to groundwater recharge sometimes more than pre-development conditions (WDEQ 1999). Permeable pavement is highly effective for stormwater management, but its effectiveness is site-dependent due to requiring permeable soils, restricted traffic and suitable adjacent land uses (WDEQ 1999). While rain barrels, green roofs, and permeable pavement reduce imperviousness, provide immediate storage, and remove pollutants, a large percentage of the stormwater runoff will continue moving through the system untreated.

As stormwater runoff moves away from sources of runoff generation towards stormwater controls, it reaches the next line of LID structures. LID grassed swales and filter strips aim to slow down runoff and promote filtering of sediment and pollutants (Martin-Mikle et al. 2015). Swales are shallow open channels with gentle sloping sides populated by erosion/flood resistant vegetation providing a natural equivalent to concrete channel stormwater conveyance (Ahiablame et al. 2012). Swale systems are vegetated wet or dry drains that filter stormwater before discharge, while filter strips are gently sloping vegetated areas perpendicular to direction of flow that filter particulate matter and associated pollutants before entering receiving waters (Deletic and Fletcher 2006). Filter strips are not a standalone LID structure but are an amendment that adds another layer of runoff prohibition and filtration. Filters and swales both use the physical processes of sedimentation and infiltration as the mechanism of stormwater runoff removal. Dry swales use infiltration to reduce runoff quantity and improve runoff water quality, while the increased surface roughness reduces runoff speed, and wet swales use residence time and natural growth to reduce peak discharge and treat stormwater runoff (Coffman 2000). Stormwater runoff treatment is primarily through sedimentation, although infiltration and adsorption contribute as well (EPA 2000). Swales have shown a wide range in removal effectiveness of TSS (mean removal of 30%-98%), TN (mean removal of 14%-61%), and TP (mean removal of 24%-99%), but consistently higher removal rates for particulates and particle bound pollutants (Ahiablame et al. 2012; Deletic and Fletcher 2006). Removal efficiencies of swales and filters are highly dependent on hydrologic characteristics, soil infiltration rates, and other site characteristics but with favorable conditions, removal efficiencies can approach complete removal. Grass swales in a residential subdivision were able to remove 99% of TSS, TP, TKN, TN, total iron, and biochemical oxygen demand (Kercher et al. 1983). Swales and filters are both simple LID BMPs that favor wide implementation due to low material costs, simple designs, low maintenance requirements, and construction efforts. One potential barrier to widespread implementation is unfavorable site characteristics like insufficient space, too steep of slope, or soils non-conducive for infiltration.

Another intermediate LID BMP used to slow down runoff and filter sediment and pollutants are rain gardens. Rain gardens, or cells, are depressed areas planted with shrubs, perennials, or trees and covered with mulch (Dietz 2007). Rain gardens provide many different services including runoff capture, infiltration, evapotranspiration, groundwater recharge, stream channel protection, peak flow reduction, and pollutant load reduction (Ahiablame et al. 2012). Rain gardens use filtering, adsorption, and biological processes along with other mechanisms to decrease degradation of stormwater runoff (Coffman 2000). General design features include 2.3- 3.3 ft of sand/soil/organic media for treating infiltrating stormwater runoff, a surface mulch layer, various vegetation, orientation to allow 6 to 12 inches of runoff pooling and, when poorly draining soils are present, an underdrain surrounded by a gravel layer to allow drainage between events (Winston et al. 2016; Davis et al. 2009). The sizing of rain gardens is dependent on soil type, geology, rainfall patterns, and amount of impervious surface, but generally they should be 5% of the contributing drainage area (<2.0 acres) (Davis et al. 2009; Davis and McCuen 2005). The many physical, chemical, and biological processes that occur in rain gardens make them highly efficient in pollution reduction. Rain gardens have been effective in their pollutant removal performance for TSS (97%), TP (35-65%), TN (33-66%), Cu (36-93%), Pb (24-99%), Zn (31-99%), oil and grease (99%), and bacteria (70%) although they can also be a source of N, P, and Cu (PGC 2007; Mullane et al. 2015). While LID BMPs provide stormwater runoff abatement and water quality improvements, design capacity of each individual LID structure is not enough to treat all runoff, and even if it was, complete removal of pollutants is rarely ever achieved. However, by combining multiple LID BMPs into a single drainage area the potential for improvements across a wider range of storm events exists. Because three specific LID BMPs were evaluated in his research, they are discussed in more detail below.

1.3.3.2 **Rain Barrels**

Rain barrels are low-cost, effective LID structures easily implemented that reduce stormwater volume and provide water-savings (Steffen et al. 2013; Coffman 2000). The general design of rain barrels includes a filtration screen to prevent contamination by vegetation, debris or insects, spigot at bottom to use the water, and an overflow outlet that provides outflow during larger storm events (Coffman 2000). Rain barrels provide multiple benefits by reducing stormwater runoff while also providing a source of non-potable water. They reduce runoff by providing retention or detention and increasing the time of concentration (EPA 2000). Although they provide runoff reduction, the standard rain barrel is hardly effective for the standard house. One standard 55-gallon barrel effectively captures runoff for a 10 m^2 rooftop, when the average southeastern United States home had rooftop footprints in the 100 to 225 m^2 range (Jones and Hunt 2010). A Blacksburg, Virginia building with a 10,000 sq-ft rooftop area using three 55-gallon rain barrels can only capture 28% of rainwater per average rain event assuming empty rain barrels at storm initiation (Gowland and Younos 2008). For rain barrels to be most effective, it is important to look at their capacity as well as potential use. Water captured in a rain barrel but not being put to use minimizes impact on stormwater runoff, so it is important to consider watering needs as well when implementing rain barrels. A 50-gallon rain barrel connected to 25% of a 2,000-ft^2 roof and watering a 150-ft^2 garden can provide a 2.4-5.4% reduction in runoff during the growing season, but only 1.4-3.1% reduction of total annual runoff (Jennings et al. 2013). While increasing rain barrel capacity or number of rain barrels would increase runoff reduction, without having a use for the water there would be little impact. Rain barrels are easily implemented, provide reduction in runoff, and disconnect the imperviousness of roofs and roads or sidewalks but have only a minor impact by themselves

1.3.3.3 **Rain Gardens**

Rain gardens attempt to mimic natural environment processes including sedimentation, adsorption, filtration, volatilization, ion exchange, decomposition, phytoremediation, bioremediation, and storage capacity. Design objectives of rain gardens include groundwater recharge, pollution prevention and removal, channel protection, and peak flow reduction (PGC 2007). Emphasizing design characteristics can enhance specific processes depending on site characteristics and contaminants of concern. The hydrological performance of rain gardens is dependent on-site characteristics, especially soil infiltration rates, and design. Many studies examined rain garden effectiveness with laboratory bench top experiments or using synthetic rain events, but long-term field studies allow performance evaluation with a wider range of conditions (Davis et al. 2003; Lucke and Nichols 2015; Davis et al. 2009; Houdeshel et al. 2015). In addition, when comparing rain gardens, it is important to analyze equivalent metrics. For example, rain gardens with low permeability soils (infiltration rates <1 in/hr) should have underdrains and comparing water leaving the underdrain is not equivalent to surface flow leaving the system (PGC 2009; Champman and Horner 2010; Hunt et al. 2006). The hydrologic performance of rain gardens has a great deal of variability. While three rain gardens in North Carolina had an average total volume reduction of 50%, but significantly less ($p<0.05$) reduction in winter than other seasons (Hunt et al. 2006). Two studies completed in Australia found a residential rain garden reduced total volume by 50.3%, while a 100% impervious car park had a 33% reduction in total volume and an

80% reduction in peak flow rates (Mangangka et al. 2015; Hatt et al. 2009). A retrofitted rain garden with a surface area to drainage area ratio of 2.1% had 97% reduction in flow volume and 91% reduction in peak flow rates (Debusk and Wynn 2011). While all rain gardens successfully deceased total volume and peak flow rates, the wide range of effectiveness is attributable to temperature, storm events, and design. Mangangka et al. (2015) found longer antecedent dry period and lesser moisture content of rain garden media enhanced treatment capacity. Hatt et al. (2009) found total rainfall or inflow volume had the largest influence on water retention and that vegetation growth could counteract compaction and clogging of the media. Storm event size also had an influence on hydrologic efficiency in Virginia, where lower retention volumes correlated with larger storms, although the substantially higher reductions are partially attributable to removing bedrock and cracks in the surrounding soil (Debusk and Wynn 2011).

Hydrologic retention and diminished peak discharge rates return runoff conditions closer to predevelopment levels, but runoff composition is still vastly different. Rain gardens alter the amount of pollution leaving the drainage area and reaching receiving areas in two ways, through reduction in concentrations and mass loads. Nutrient removal efficiencies have had mixed results (Chen et al. 2013; Hunt et al. 2008; Hatt et al. 2009). Rain gardens have been shown to be effective in decreasing total nitrogen and ammonium loads but have been inconsistent with nitrate load reductions (Hatt et al. 2009; Hunt et al. 2008; Chen et al. 2013; Dietz and Clausen 2006). Some of the variation in nitrogen concentrations is due to different designs. It is believed that including an internal water storage layer or saturation zone creates an anaerobic environment promoting denitrification and increased nitrate removal (Chen et al. 2013; Dietz and Clausen 2006;

Hunt et al. 2008). While in many cases, rain gardens with internal storage show increased nitrogen removal, it has not been 100% effective either (Hunt et al. 2006). Total phosphorous has had similar mixed results as well. Removal of phosphorous can range from an 85% decrease in concentrations and an 86% decrease in load, all the way to a 400% increase in mass load (Davis et al. 2006; Hatt et al. 2009). The increase in phosphorous is likely due to the soil or filter media. Dietz and Clausen (2006) found the effluent TP concentrations were higher at the beginning of the two-year study but decreased over the course of the study. While Hatt et al. (2009) found that filter media with TP concentrations of 150 mg/kg reduced TP concentrations by 80%, the filter media with 380 mg/kg TP increased TP loads by 400%.

Sediment and metals removal has been much more consistent. Rain gardens have been effective in the removal of TSS and metal species with removal rates often greater than 50% and sometimes approaching 100% (Ahiablame et al. 2012; Hunt et al. 2008; Hatt et al. 2009). There can still be site impacts with one site in the North Carolina Piedmont having a 330% increase in Fe (Hunt et al. 2008). The increase in Fe is attributable to the soils of the region having a large amount of clay with high Fe content (Hunt et al. 2008). Rain gardens are an effective stormwater management practice, but it is important to understand what priority pollutants there may be to ensure design specifications that maximize removal and retention

1.3.3.4 **Permeable Pavement**

Permeable pavements are similar to rain gardens in the fact they are designed to temporarily store runoff and allow infiltration into the subsoil. The four different types of permeable pavements are block pavers, plastic grid systems, porous asphalts, and porous concretes (Dietz 2007). Permeable pavement reduces runoff volume from 29% all the way up to 93% while reducing peak discharge by 67% (Rushton 2001; Dreelin et al. 2006; Bean et al. 2004). The main mechanism behind runoff reduction is infiltration, although permeable pavement also has a large storage capacity so that plays a role in runoff reduction as well. Rushton (2001), using porous pavement and a grass swale, showed a 65% reduction in runoff compared to asphalt and 31% reduction compared to asphalt with a grass swale but the dimensions of the permeable pavement were not presented. Dreelin et al. (2006) achieved a 93% reduction in stormwater runoff but the design retained 7-8 cm of a storm using a 25-cm gravel base and a majority of sampled events were smaller storms. The storage capacity is further emphasized in the fact that the Dreelin et al. (2006) site had clay soils with infiltration rates of 4.8-16.7 cm/hr while Rushton (2001) study sites were located in the more permeable sandy soils of Tampa, FL. Besides reducing runoff and peak flow rates, permeable pavement was also able to delay peak outflow by 78 minutes (Bean et al. 2004). Permeable pavement effectively reduced TSS, TP, TN, NH4, and the metals Zn, Cu, Fe, Pb, and Mn mass loads (Bean et al. 2004; Rushton 2001; Brown and Borst 2015). Nitrate reduction was more variable with some instances of nitrate production (Drake et al. 2014; Bean et al. 2004). The source of the nitrate production is most likely nitrification since the permeable pavement also raised pH values to ranges that promote microbial nitrification (Bean et al. 2004; Drake at al. 2014). One potential drawback of permeable pavement is that it works efficiently due to pore space within the concrete, but that pore space can easily become clogged with fine sediments so periodic maintenance is required to ensure maximum efficiency (Sansalone et al. 2012; Bean et al. 2004). Permeable pavement is an effective LID structure that can improve water quality and return hydrologic conditions closer to predevelopment conditions, but also requires significant resources and maintenance.

1.3.3.5 **Multiple Low Impact Development Structures**

Each individual LID structure has specific pollutants and storm events for which they achieve the highest efficiency. The combination of multiple LID structures within one watershed promotes higher removal and retention rates than any individual LID structure on its own. Multiple LID structures within a single watershed are effective at reducing runoff and nutrient concentrations but most have not returned to predevelopment levels (Line and White 2015; Dietz and Clausen 2005). An LID subdivision in Waterford, CT utilizing permeable pavement, bioretention, grass swales, and shared driveways had the same runoff and pollutant concentrations as the predevelopment values, but that could be partially due to having more permeable soils than the surrounding area (Diet and Clausen 2008).

When compared to traditional development neighborhoods, peak discharge was 1,100% greater and amount of rain needed to create runoff was 100% greater (3 mm vs 6 mm) for the traditional neighborhood than the LID neighborhood. The lag times were also significantly greater for small (<25.4 mm) and short duration (<4 h) storms for the LID neighborhood than the traditional neighborhood (Hood et al. 2007). This suggests the incorporation of multiple LID structures closely mimics natural conditions but once reaching capacity, the return diminishes.

Another LID neighborhood in Cross Plains, WI had a strong hydrologic impact, reducing total annual discharge by 1.3 to 9.2 times but did not improve water quality. The LID neighborhood actually had increased annual loads for TSS and TP but 70% of those loads were associated with just two large storm events. The events had precipitation depths between 4.89 to 6.21 in and intensities from 1.13 to 1.2 in/hr further highlighting decreased effectiveness once capacity is reached (William and Bannerman 2008).

A 3-acre neighborhood in Wilmington, MA with rain gardens and permeable paving did not have such drastic results with no statistically significant differences between pre- and post-LID conditions. Runoff quantity for storms with 0.25 inches or less of precipitation decreased, but no significant differences in concentrations or loads between pre- and post-installation were demonstrated (Waldron et al. 2010).

LID neighborhoods have been implemented around the country but are mostly concentrated on the coasts or northern states (Kloss and Calarusse 2006). The wide variation in effectiveness depending on site characteristics and storm event type highlights the importance of testing LID neighborhoods in a multitude of conditions. Oklahoma presents an excellent opportunity because of its diverse weather patterns and less permeable clay soils.

1.4 Hypotheses

- Multiple LID BMPs decrease peak discharge rates and total discharge for all event types compared to traditional curb and gutter stormwater management practices
- The combination of multiple LID BMPs decreases first-flush nutrient concentrations and maintains physical water quality parameters of stormwater runoff compared to traditional curb and gutter stormwater management practices
- Retention ponds with the greatest pond surface area to developed land ratio will have decreased nutrient concentrations compared to retention ponds with smaller ratios

1.5 Objectives

The objectives of this research were as follows:

- Use storm hydrographs to demonstrate LID BMP effectiveness in decreasing peak discharge and total discharge compared to traditional curb and gutter stormwater management techniques.
- Quantitatively describe first-flush stormwater quality pollutant concentrations and loads leaving a basin with LID BMPs versus traditional curb and gutter.
- Compare retention pond water quality to determine impact based on volume of retention pond, surface area of retention pond, ratio of developed and undeveloped land, and percent impervious surface area within the basin.

2 Methods

2.1 Site Descriptions

2.1.1 Lake Thunderbird Watershed

The study site is in central Oklahoma in the Lake Thunderbird watershed (Figure 2). The watershed is a 666 km2 mixed-use watershed with 40 percent considered residential (Martin-Mikle et al. 2015). The urbanization of the watershed has predominantly occurred in the last 40 years with significant population growth occurring the last 30 years, causing pastures to be converted into urban areas (ODEQ 2010; Martin-Mikle et al. 2015; OCC 2010). Lake Thunderbird provides drinking water for over 200,000 people in the cities of Norman, Del City and Midwest City, as well as recreation for the surrounding areas.

Urban development within the watershed has caused excessive nutrient loading leading to eutrophication of the lake. One of the major excessive nutrient loads is phosphorous with an estimated 18,000 kg of P entering Lake Thunderbird each year. The primary source of phosphorous is non-point source pollution from increased impervious surfaces due to urbanization (OCC 2010). The excessive loading of Lake Thunderbird has led to it being classified as an impaired water body for turbidity, dissolved oxygen, and chlorophyll-a with total maximum daily loads (TMDL) to be developed for sediment and dissolved oxygen impairments (ODEQ 2010; OCC 2010). The major tributary of Lake Thunderbird is the Little River. The study sites consisted of a neighborhood paired-watershed, Trailwoods, and four neighborhood retention ponds; Carrington Lakes, Shadow Lake 1, Shadow Lake 3, and Deerfield, all located within the Little River watershed.

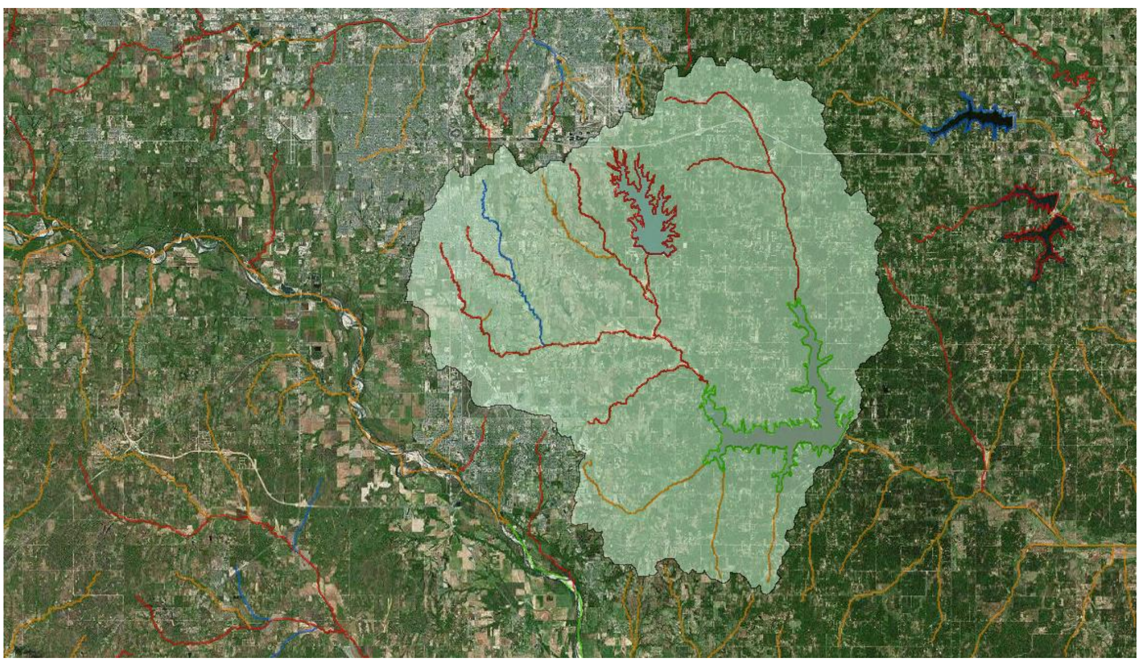

Figure 2: Map of Lake Thunderbird watershed and major tributaries

2.1.2 Paired Watershed Study

The Trailwoods neighborhood located in north central Norman, OK within the Little River watershed was selected as the location for the paired watershed study. A portion of the neighborhood was divided into two equal drainage areas (Figures 3-4).

Figure 3: The location of the paired watershed study site, the Trailwoods neighborhood, highlighted on the map.

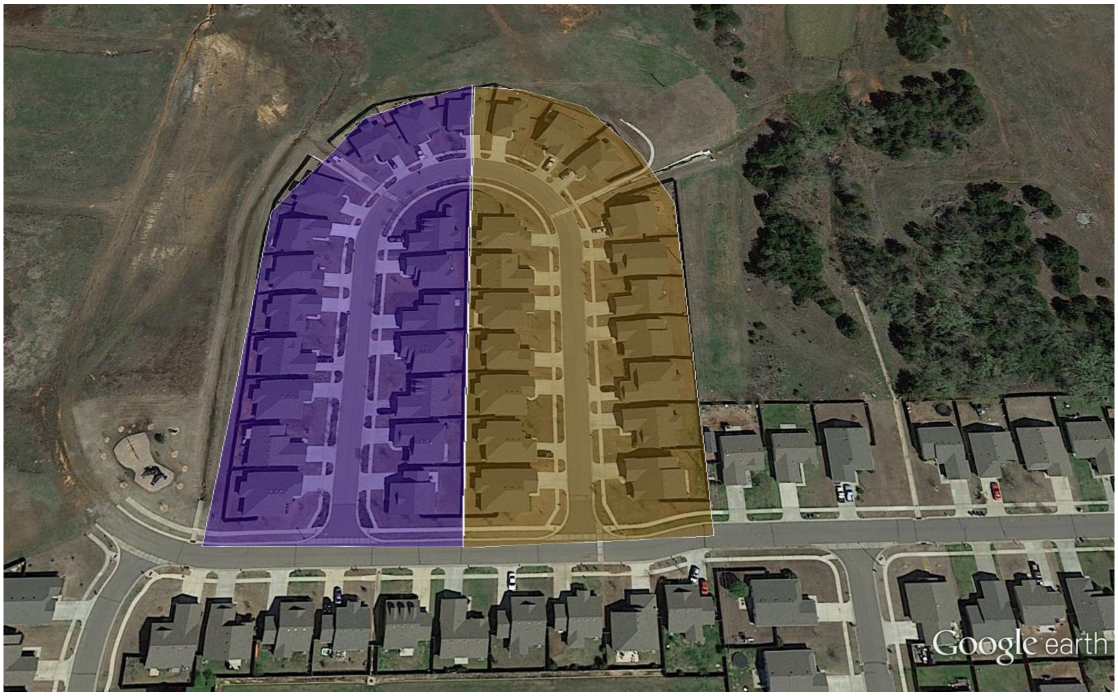

Figure 4: The treatment watershed, westside, highlighted in purple, and the control watershed, eastside, highlighted in yellow, for the paired watershed study

The control drainage area, approximately 2.28 acres, was built using the traditional stormwater management practice employing curb and gutter conveyances. Stormwater runoff is quickly directed away from structures and into the street where it eventually reaches a concrete storm channel. The concrete channel is approximately 140 feet long and 4.25 feet wide. A pre-constructed plastic-coated Fiberglass 18” by 45° trapezoidal flume with a throat of 6 inches was located at the end of the concrete channel. The experimental basin, approximately 2.31 acres, was built incorporating LID BMPs. The LID BMPs included diverted downspouts, an aesthetically pleasing rain barrels located at the front corner of each house with close proximity to gardening areas, rain gardens installed in front of each home between the sidewalk and road, and a small section of permeable pavement which was located at the beginning of the concrete stormwater channel. The rain barrels were 50-gallon barrels equipped with a downspout from the roof to direct runoff into the barrel, an insect screen at the top opening, and a spigot at the bottom to allow the water to be used. The rain gardens were installed at each house with an average area of 265 sq. ft. The rain gardens are 36” deep at the top of the basin, then decrease to 18” deep at the bottom of the basin. The composition of the rain gardens includes a substrate mix of 70% expanded clay, 20% sand, and 10% compost by volume, 4” layer of sand, and a 6” perforated flexible pipe that was surrounded by 5/8” limestone aggregate and enclosed with landscape fabric. The permeable pavement section was approximately 12’ long by 10’ wide. The pad was composed of 3’ of aggregate with 6” of permeable concrete on top of the aggregate. Stormwater runoff is directed into a concrete stormwater channel approximately 60’ long and 4’ wide leading into a pre-constructed plastic-coated Fiberglass 18” by 45° trapezoidal flume with a throat of 6 inches. Stormwater runoff has two potential pathways to reach the test flume. Runoff can flow through the rain gardens and come out the downstream side of the permeable pavement or it can over flow from the street over the permeable pavement.

An ISCO 1640 liquid level actuator, secured to the side of the trapezoidal flume at the end of each concrete channel, was connected to an ISCO 6712 portable sampler and a 730-bubble module. The liquid level actuator and bubble module were used to measure water levels passing through the flume at one-minute increments which were recorded by the portable sampler. The recorded levels were downloaded from the sampler and entered in the flume equation developed for a trapezoidal flume to convert water levels to volumetric flow in cfs (Eqn. 1). The volumetric flows were plotted against time to develop hydrographs for the control and experimental watersheds to determine LID BMP impact on stormwater runoff total volume and peak discharge rates.

$$Flow\ (CFS) = 2.853 * ((level\ (ft) + 0.13558)^{2.497}\ (1)$$

2.1.3 Retention Pond Study

Four retention ponds located within the Little River watershed were studied. All provide stormwater retention for residential development (Figure 5).

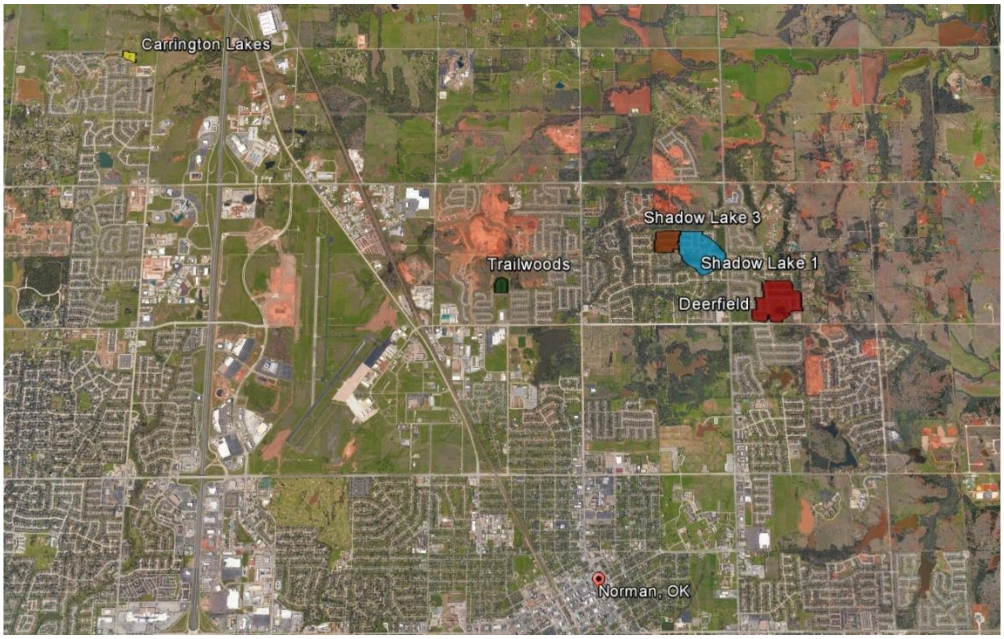

Figure 5: Map showing the locations and drainage areas of the areas retention ponds in relation to the location of the paired watershed study site, Trailwoods

The Carrington Lake subdivision, located in northwest Norman, had a developed retention pond with a surface area of 0.2 acres draining 3.59 acres of residential development with a runoff coefficient of 0.57 (Figure 6). A bioretention cell was built at the inlet of the retention pond. The Deerfield neighborhood, located in north central Norman, two miles east of the Trailwoods subdivision, had a drainage basin of 45.4 acres with a runoff coefficient of 0.59 that drained into a 2.33-acre surface area pond (Figure 7). The other two retention ponds were both located in the Shadow Lake neighborhood and were preexisting farm ponds converted into residential retention ponds. Shadow Lake is located in north central Norman between Trailwoods and Deerfield. The larger retention pond, Shadow Lake 1, had a drainage area of 47.32 acres with a runoff coefficient of 0.56 (Figure 8). The retention pond has a surface area of 2.43 acres. The Shadow Lake 3 drainage area is 15.23 acres with a runoff coefficient of 0.66 and a retention pond surface area of 0.35 acres (Figure 9).

Figure 6: The portion of the Carrington Lakes neighborhood that drained into the retention pond used in this study

Figure 7: The Deerfield retention pond and the corresponding drainage area analyzed during this study

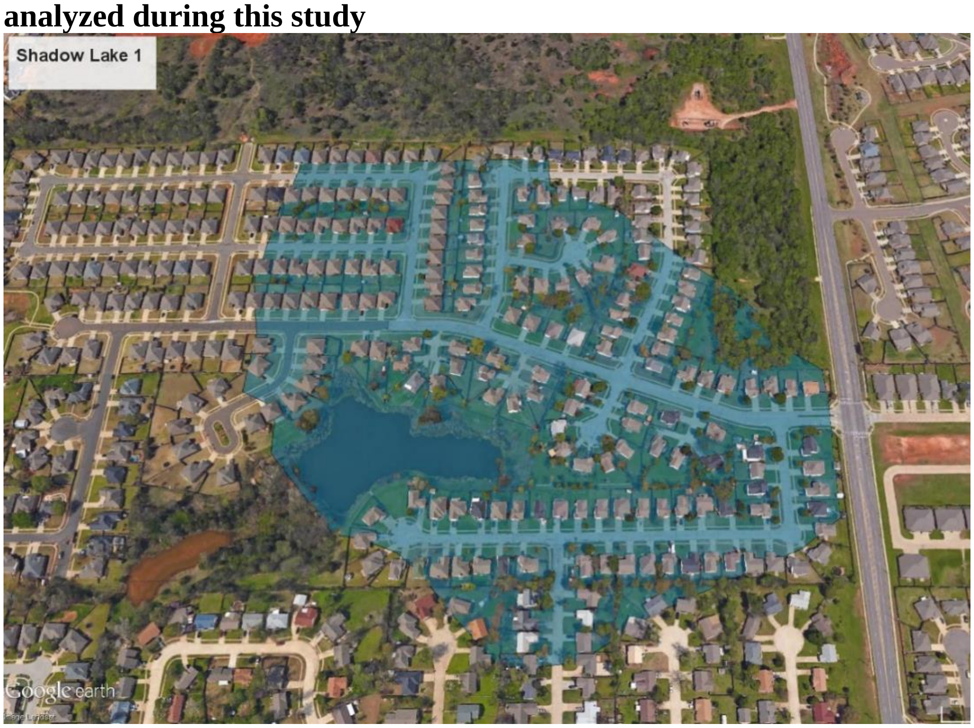

Figure 8: Section 1 of the Shadow Lake neighborhood with the sampled retention pond and correlated drainage area highlighted

Figure 9: The drainage area of the Shadow Lake #3 drainage area and its corresponding retention pond

2.2 Water Quality Data Collection

2.2.1 Paired Watershed Study

An ISCO 6712 portable sampler was installed nearby each test flume to collect water samples. A water level of 0.15 ft in the flume enabled the start of the program for water collection of the storm event first flush. Samplers rinsed the line three times before collecting 3.5-L of water into a composite 5-L bottle housed in the portable sampler unit. The physical collection of the composite samples occurred during each appropriately sized event, or as soon as possible after the event, and were returned to the Center for Restoration of Ecosystems and Watersheds (CREW) laboratories. The composite samples were then divided into clean bottles for analysis of each specific laboratory parameter.

2.2.2 Retention Pond Study

The four retention ponds were equipped with Hach Sigma 900 portable samplers and rain gauges deployed near the outlet structure of each retention pond. The Sigma tipping bucket rain gauge measured the total rainfall and rainfall intensity which were stored in the Sigma 900 controller module. Once a rainfall intensity of 0.3 in/hour was measured, the sampling program was enabled. The sampling program waited 30 minutes after enabling to ensure enough stormwater runoff had reached the retention pond to cause outflow. The sampling program rinsed the line three times, then collected 4-L of water distributed into four 1-L bottles. The samples were retrieved as quickly as possible after the event, returned to the CREW laboratories, and divided into new clean bottles for the analysis of each specific laboratory parameter.

2.3 Laboratory Sample Analysis

Field parameters (pH, dissolved oxygen concentration, dissolved oxygen saturation, temperature, conductivity, salinity, specific conductance, resistivity, and total dissolved solids) were measured using an YSI 600QS multi-parameter data sonde and 650 MDS controller (Table 1). If sufficient water was present in the flume, field parameters were measured in-situ but, if not, measurements were taken from the collected water sample (Table 1). The laboratory analyses consisted of measuring total suspended solids (TSS), biochemical oxygen demand (BOD), dissolved reactive phosphorous (DRP), total phosphorous, nitrate-nitrogen, ammonia-nitrogen, total nitrogen, and a suite of metals (Table 2).

Table 1: Field parameters and methods

Field Parameter	Units	Instrumentation (method)
pH	Standard Units	YSI600QS (CREW 2004)
Dissolved oxygen	mg/L and %	YSI600QS (CREW 2004)
Temperature	°C	YSI600QS (CREW 2004)
Salinity	ppt	YSI600QS (CREW 2004)
Conductivity	mS/cm	YSI600QS (CREW 2004)
Resistivity	Ohm-cm	YSI600QS (CREW 2004)
Specific conductance	mS/cm	YSI600QS (CREW 2004)
Total Dissolved Solids	g/L	YSI600QS (CREW 2004)
Volumetric Discharge rate (Only at Trailwoods)	CFS	ISCO 730 bubbler flow module

Table 2: Laboratory parameters and methods

Laboratory Parameter	Units	Method reference
Total suspended solids	mg/L	EPA 160.2 (1999)
Biochemical oxygen demand	mg/L	EPA 5210b (1999)
Total phosphorous	mg/L	EPA 365.3 (1978)
Dissolved reactive phosphorous	mg/L	EPA 365.3 (1978)
Nitrate-nitrogen	mg/L	EPA 352.1 (1971)
Ammonia-nitrogen	mg/L	HACH 8038 (2014)
Total Nitrogen	mg/L	HACH 10071 (2014)
Metals (Al, As, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, Pb, and Zn)	mg/L	EPA 3015 (2007) and 6010 (2007)

2.4 Data and Statistical Analysis

The results of this project include data collected from August 2013 through May 2015. A total of 42 stormwater runoff events for both watersheds with measurable hydrographs occurred, while a total of 20 sampled events occurred. An overview of precipitation during this period provides a reference point for both the water quality and quantity results, as well historical context. The water quantity analysis is broken down by comparisons of observed versus ideal theoretical hydrographs, total runoff volume, and peak discharge. Total runoff is analyzed by two different categorizations, a temporal categorization and a depth categorization. The temporal categorization uses divisions of equal time increments, 24 hours, for both watersheds to determine total runoff. This allows both watersheds to have the exact number of data points to determine runoff volumes. The second categorization used flume water depth levels as the source of division. The depth categorization was necessary to provide a more concise picture due to the water level actuator being slightly above the base of the flume, having a measurement error for water levels below 0.03”, and the flume equation having a residual flow of 1.1656 CFS for depth levels of zero. The temporal categorization likely over estimates actual total runoff volumes, while the depth categorization likely under estimates actual volumes, so by using the two different categorizations a more comprehensive picture is formulated.

The control basin and treatment basin water quality data were categorized by event type to determine effectiveness under differing storm event conditions. The different storm characteristics were total rainfall, rainfall intensity, and time since rainfall (Table 3). Student t-tests were completed to determine significant differences between the treatment watershed and control watershed stormwater runoff water quantity and quality.

The stormwater water quality constituents were compared between each individual retention ponds. The data were also normalized by retention pond and watershed characteristics such as percent impervious surface area, drainage area, and runoff coefficient to provide a more direct comparison between the retention sites. Using the National Stormwater Database, similar sites to each individual retention pond were grouped by drainage area and land use, then site characteristics and sample values were averaged together to create a reference point.

Table 3: Storm event characteristic divisions

Paired Watershed Study	Storm Characteristic Divisions		
5-min Rainfall Intensity (in/hr)	<1	>1	
Total Rainfall (in)	<0.39	0.39 – 1.18	>1.18
Time Since Last Storm (days)	0-2	2-6	>6

3 Results/Discussion

3.1 Paired Watershed Study

3.1.1 Precipitation Data

The 21-month study showed large variability in storm characteristics. Total rainfall, five-minute rainfall intensity, and antecedent dry period were used as surrogates for classifying the storms. Total rainfall ranged from 0.05 to 10.16 inches with a mean of 0.75 inches and a median of 0.39 inches, intensity ranged from 0.06 to 4.55 in/hr with a mean of 0.89 in/hr and median of 0.48 in/hr, and antecedent dry period ranged from 1 to 26 days with a mean and median of 6.1 and 4.0 days, respectively (Table 4). In historical context, the 21-month study was slightly drier than average with study period mean and median monthly rainfall totals being 3.01 inches and 2.05 inches compared to the historical means and medians of 3.23 and 3.13 inches, respectively (Table 5). Additionally, only 6 of the 21 months had monthly rainfall totals greater than their historical average (Figure 10).

Storm events were also broken down into frequency distribution diagrams based on intensity, total rainfall, and days since last rain. The rainfall intensity distribution shows the greatest frequency of storms, 31%, had less than 0.31 in/hr, while 67% of storms had intensities less than 0.81 in/hr (Figure 11). Total rainfall followed similar patterns with 29% falling in the lowest bin at rainfall totals less than 0.18 inches and 71% of storms falling within the first four bins with rain totals below 0.59 inches (Figure 12). Lastly, days since last rain showed that 43% of the events occurred in the 1 to 3-day range (Figure 13). The 21-month study was represented mostly by small, less intense storms with relatively short periods of time since the last measurable precipitation.

Table 4: Statistical summary of storm characteristics

	Five-Minute Max Intensity (in/hr)	Total Rainfall (in)	Antecedent Dry Period (days)
Mean	0.90	0.77	5.71
Median	0.54	0.40	4.00
Std. Dev	0.92	1.54	5.19
Max	4.55	10.16	26.00
Min	0.12	0.05	1.00
Std. Error	0.14	0.24	0.80

Table 5: Monthly rainfall statistical summary September 2013-May 2015 (OCS 2018)

	Monthly Total Rainfall (inches)	Rainfall >0.01 (days)	Rainfall >0.1 (days)	Greatest 24 hour Rainfall (inches)	Historical Total Rainfall Averages (1981-2010)
Mean	3.01	8	4.62	1.03	3.23
Median	2.05	7	4	0.76	3.13
Std. Dev	4.88	3.89	3.71	1.04	
Maximum	23.39	19	15	4.67	
Minimum	0.1	3	0	0.05	
Std. Error	1.06	0.85	0.81	0.23	

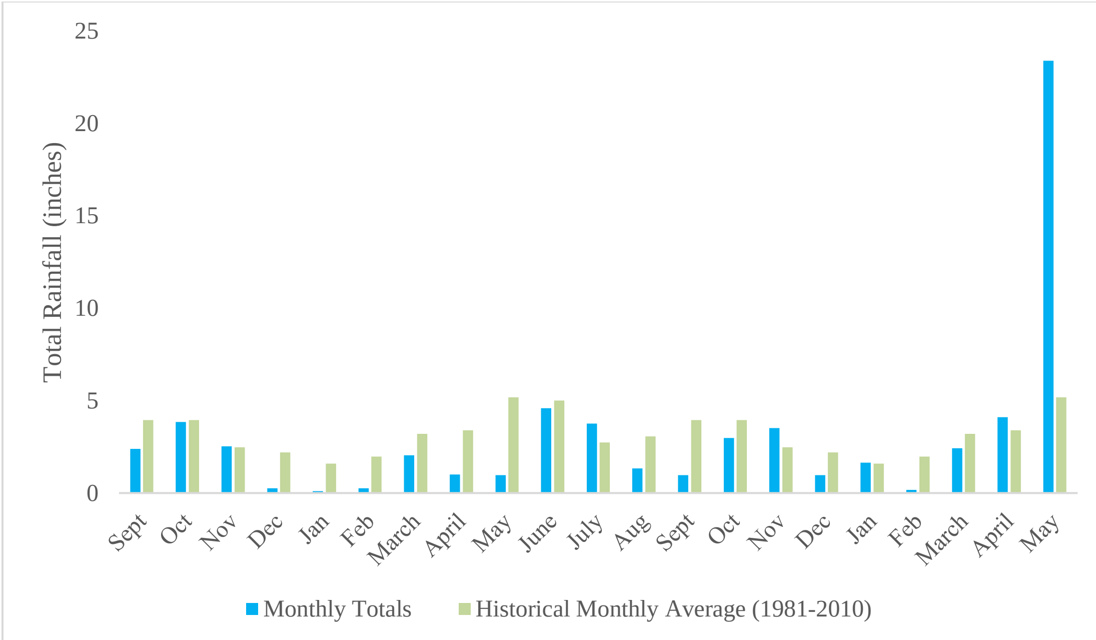

Figure 10: Observed monthly rainfall totals and historical averages from 1981-2010 (OCS 2018)

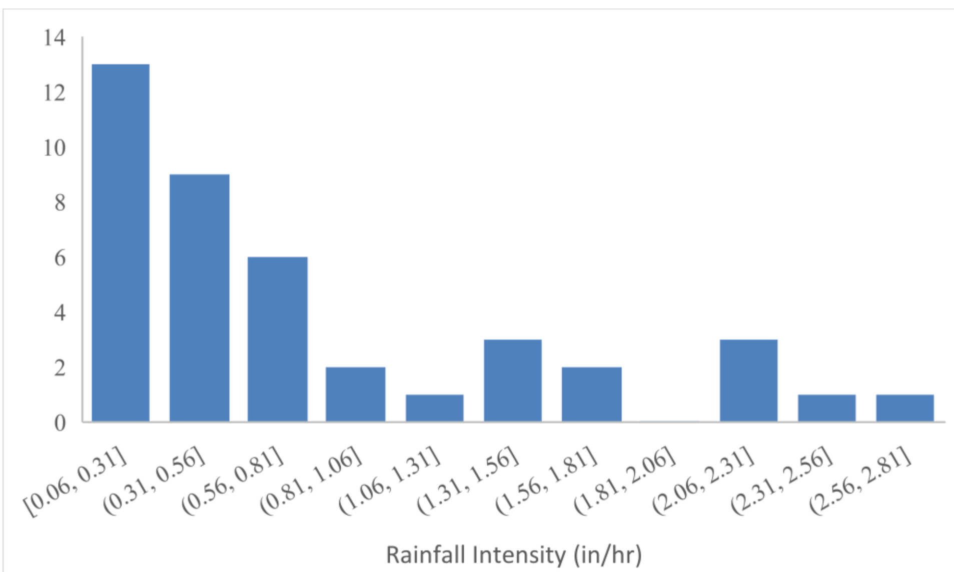

Figure 11: Frequency distribution of five-minute max rainfall intensity (in/hr) (OCS 2015)

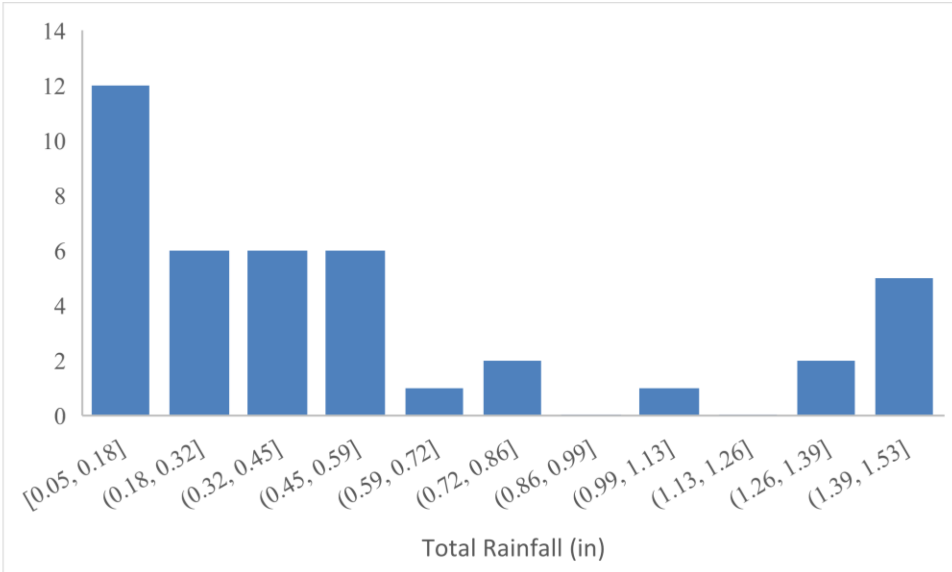

Figure 12: Frequency distribution of total rainfall (in) (OCS 2015)

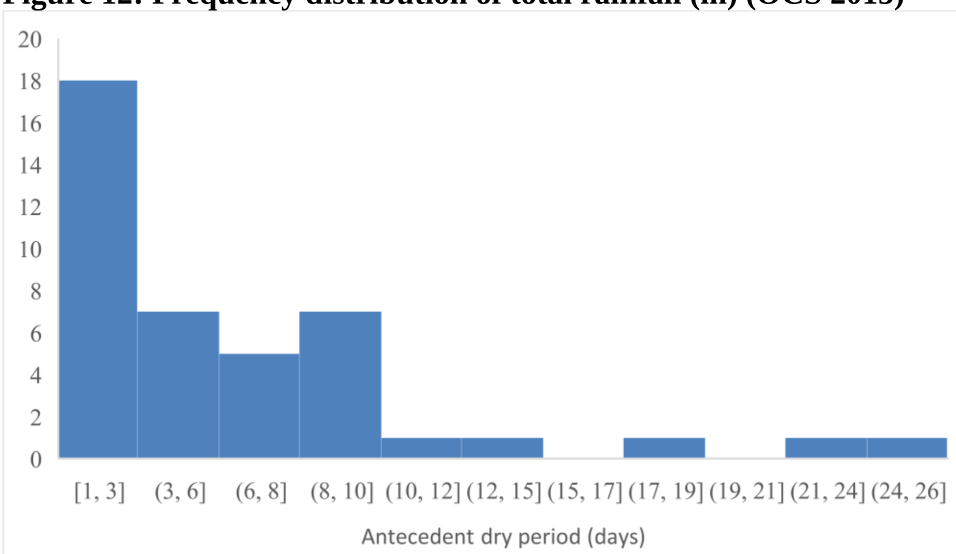

Figure 13: Frequency distribution of antecedent dry period (days) (OCS 2015)

3.1.2 Hydrology Data

Conversion of an undeveloped area into a developed urban area dramatically alters the hydrology of stormwater runoff. The hydrograph becomes flashier with higher peaks, more total runoff, and has a shorter duration between beginning and end (Coffman 2000; Arnold and Gibbons 1996). These expected trends were seen in this study (Figures

14-15). Using total runoff, peak discharge, and runoff duration as surrogates for watershed hydrology gives an assessment of potential developmental impact. Reductions in total runoff can decrease flooding, increase groundwater infiltration, and limit stormwater infrastructure requirements (EPA 2000). Reductions in total volume also inevitably result in reduced peak discharge and increased lag time (Taylor 2013). The recent stormwater philosophy of volume-based hydrology (VBH) believes focus variables like velocity, peak flow, impervious percent, and event mean concentration (EMC) reduction are just proxies for increased volume and if volume becomes the primary focus, then the other variables will follow (Reese 2009). Hydrologic changes help prevent erosion and habitat degradation while increasing groundwater infiltration and will more closely resemble predevelopment conditions (Coffman 2000).

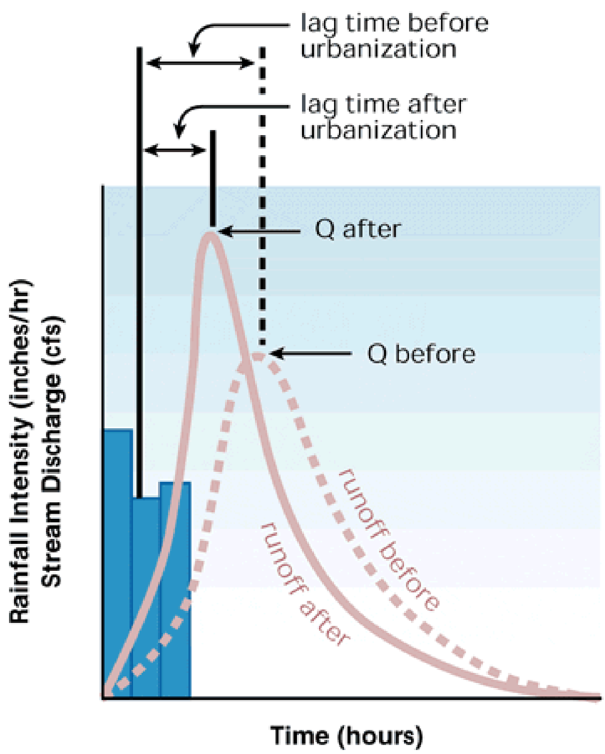

Figure 14: Changes in hydrographs after development created by the EPA for Federal Interagency Stream Restoration (EPA 2017)

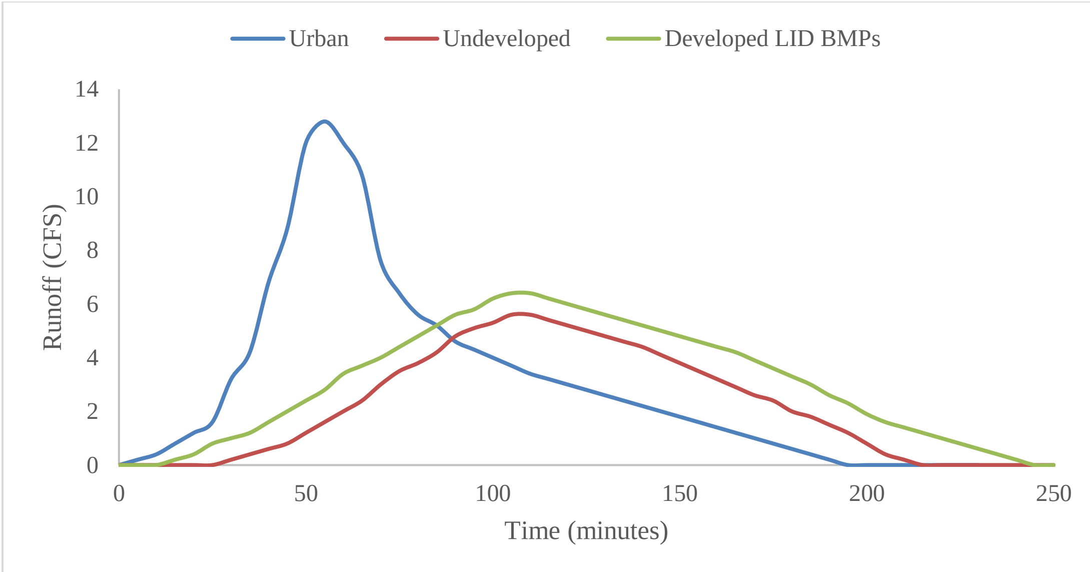

Figure 15: Theoretical hydrographs of undeveloped land, urban development, and urban development with LID BMPs Modified from EPA 2000; Coffman 2000; and Dams et al. 2008

3.1.3 Hydrograph Comparisons

Traditional development increases the peak hydrograph, increases the speed that rainwater begins running off, and causes more water to leave the localized area (Coffman 2000; Arnold and Gibbons 1996; Jefferson and Jarnagin 2002; Olivera and DeFee 2007).). LID BMPs have been shown to effectively reduce total discharge, peak discharge rate, and increase lag time in many cases (Figures 14-15) (Coffman 2000; EPA 2000; William and Bannerman 2008; Hood et al. 2007). In this study, the hydrographs more closely resemble theoretical developed area hydrographs than undeveloped hydrographs but do show improvements (Figures 16- 20).

Using rainfall intensity and peak discharge to determine representative hydrographs, five representative examples were selected. The five hydrographs are representative of seasonal variations in Oklahoma storm characteristics, with each event deviating from the mean rainfall intensity, total rainfall, or time since last storm (Table

6). The early spring event is characterized by infrequent, low intensity storms (Figure 16). The late spring/early summer had higher intensities and rainfall totals, and more frequent storms (Figure 17). A mid-summer storm had average intensity but higher than average total rainfall and days since previous rain storm (Figure 18). A late summer/ early fall event having medium storm frequency, but intensity and total rainfall amounts slightly less than the stronger late spring/early summer storms was also included (Figure 19).

All five hydrographs show similar times of concentration, or time it takes water to reach the end of the catchment from the furthest point, for both watersheds with hydrograph peaks occurring at essentially the same time (±3 minute) for the representative events. This is likely due to the small size of the study watersheds. The treatment watershed also had reduced peak discharge values in every scenario except the low intensity, low total rainfall early spring event (Figure 16). The treatment watershed follows the theoretical developed hydrograph, although at subdued levels, much more closely than the undeveloped hydrograph (Dams et al. 2008). The more similarities with developed hydrographs than undeveloped hydrographs could be explained with a variety of localized conditions. The small catchment size means time of concentration are much shorter, potentially minimizing noticeable impacts. The similar timing of the hydrograph peaks is most likely attributable to stormwater that is not encountering any of the LID BMPs, such as runoff from the roads or driveways. While the runoff should at a minimum encounter the permeable pavement, it was commonly observed that large flows of water would move over the permeable pavement. The potential causes of the runoff sheet flows over the permeable pavement will be discussed in more detail later.

The second hydrograph had variable, peak discharge, exhibiting a stronger response to the LID BMPs. The LIB BMP hydrographs had decreased peak discharge rates for all the representative hydrographs, except the below average intensity rainfall/below average peak discharge event. The cause of this result could again be related to runoff not interacting with the LID BMPs, especially in this instance in which the rainfall intensity was substantially lower than the study average. It is possible that the water that reached the sampling point was only runoff from driveways and roadways. While LID BMP hydrographs do not resemble the theoretical LID BMPs hydrographs presented by Coffman (2000), there are improvements. Dissecting the specific components of the hydrographs; peak discharge, total runoff, and lag time, provides insight into areas of effectiveness and potential improvements.

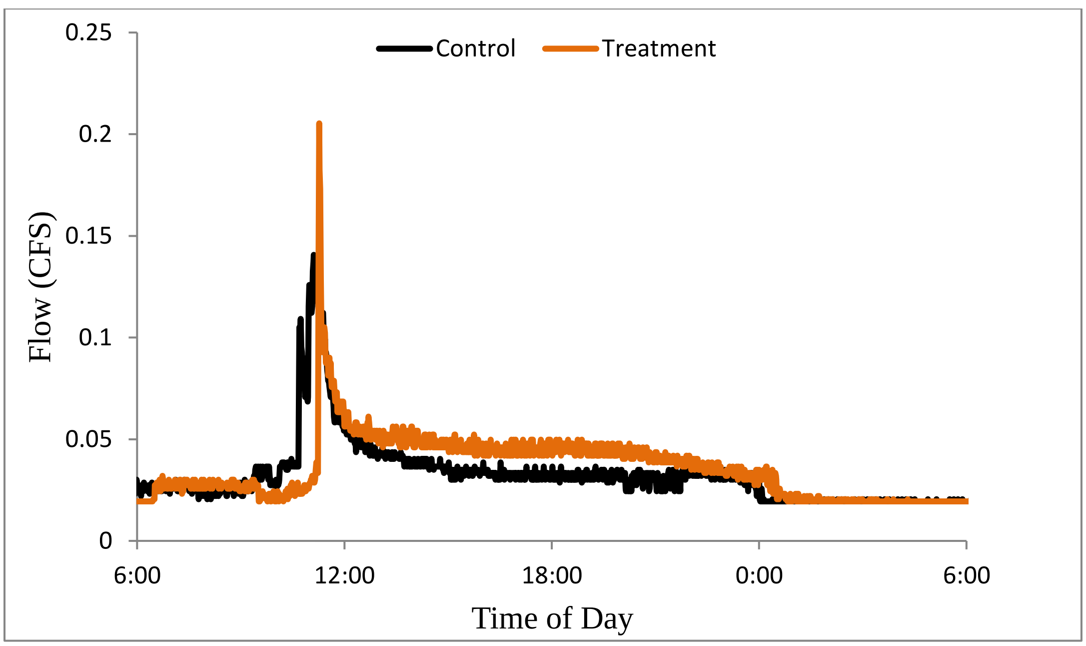

Figure 16: Early spring storm hydrograph

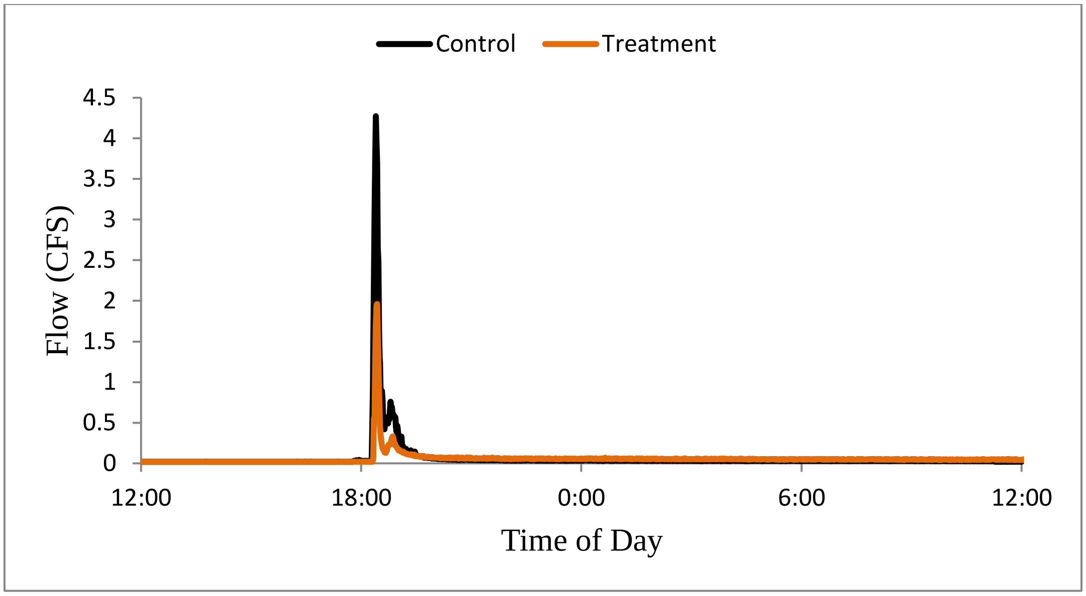

Figure 17: Late spring storm hydrograph

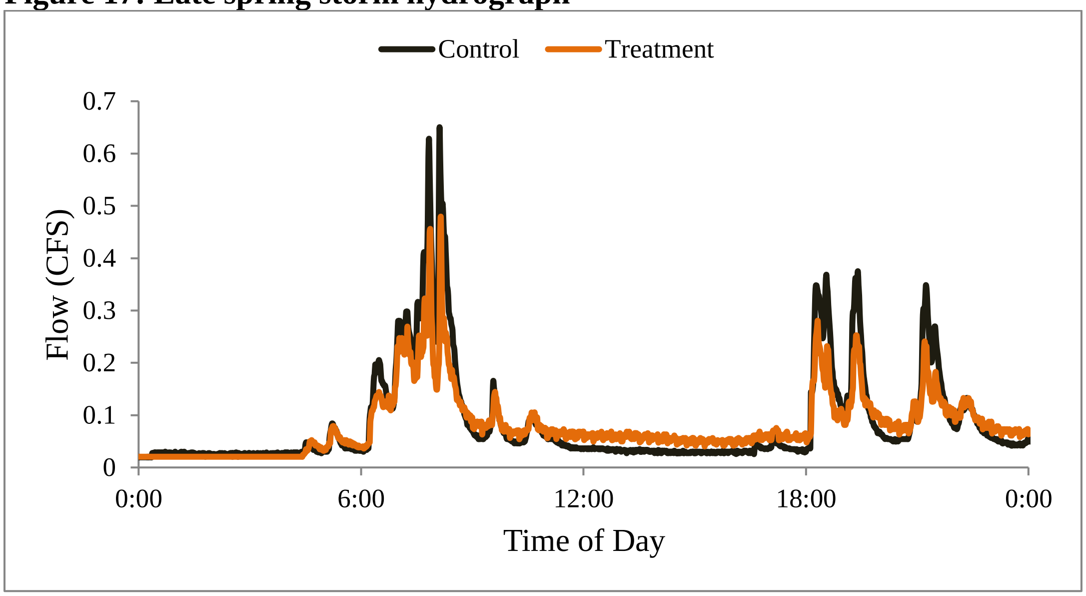

Figure 18: Mid-summer storm hydrograph

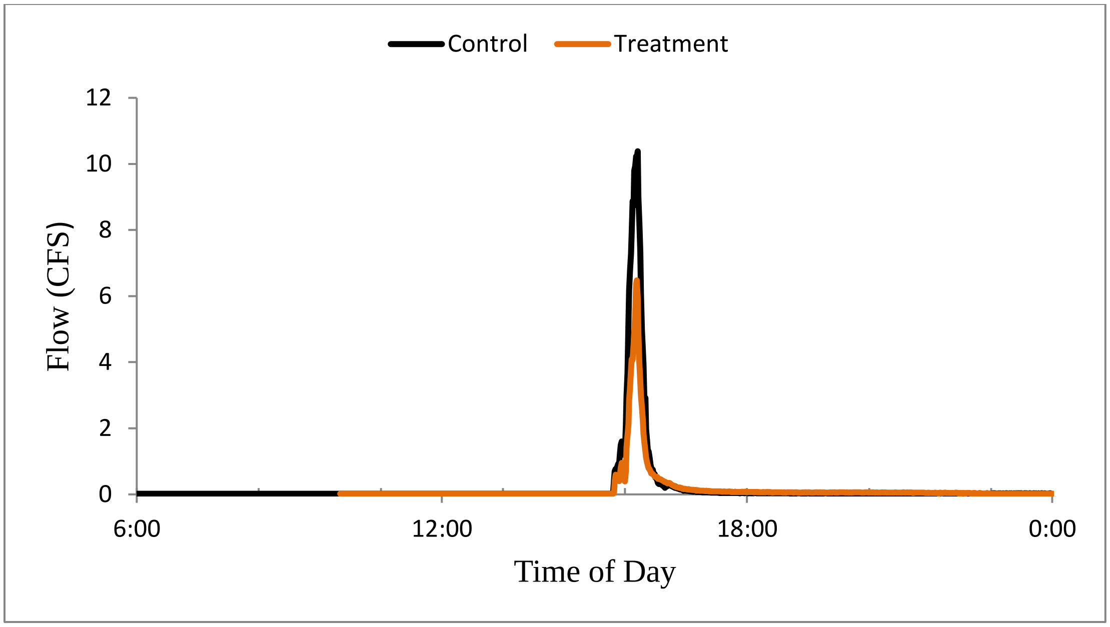

Figure 19: Late summer storm hydrograph

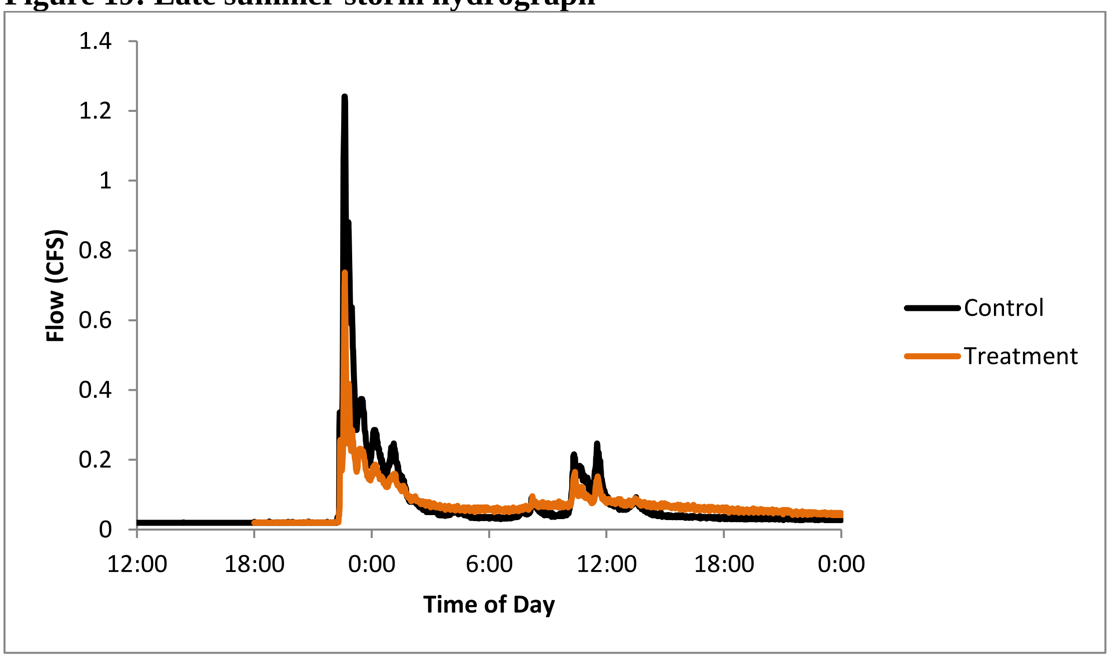

Figure 20: Fall storm hydrograph

Table 6: Rainfall characteristics used for 5-selected hydrographs

	Five-Minute Max Intensity (in/hr)	Total Rainfall (in)	Antecedent Dry Period (days)
Mean of All Events	0.89	0.75	6.1
3/5/2015	0.12	0.07	2
3/25/2015	2.76	0.47	5
7/30/2014	0.72	1.44	12
8/9/2014	1.8	0.76	2
10/13/2014	1.32	1.01	1

3.1.4 Total Runoff

Numerous individual LID BMPs have effectively reduced runoff with bioretention removing 40% to 97%, permeable pavement effectively reducing volumes 50 % to 93%, and rain barrels removing their available capacity (Rushton 2001, DeBusk and Wynn 2011, Droelin et al. 2006, Ahiablame et al 2011). However, incorporation of multiple LID BMPs into a single watershed has fewer examples with more mixed results (Hood et al. 2007, Hinman 2009, Line and White 2015, Bedan and Clausen 2009).

Over the two-year study, a total of 42-rainfall events producing at least 2000 ft^3 of runoff and having measurable precipitation occurred. Using hydrographs determined temporally with volumes calculated for equal time increments of at least 24 hours for both watersheds, the mean runoff volumes were calculated (Table 7). Treatment watershed runoff volumes were not significantly different than control watershed volumes ($p < 0.05$). The standard deviation in total runoff was 31% lower, substantially larger reduction than the mean values.

Table 7: Summary table of temporal categorization of total runoff (n=42)

	Control	Treatment	Difference	Percentage
Mean	10024	8870	1153	12
Median	6089	6741	-105	-4
Std. Dev.	21134	12992	8676	31
Maximum	142671	87372	55299	39
Minimum	1907	1912	-6341	-115
Std. Error	3301	2029	1355	-
P-value	0.20	-	-	-

While using temporal divisions of the hydrographs gives an exact comparison of measured runoff moving through the flumes, it also introduces a degree of uncertainty in total volumes. The flume equation has a residual value of 1.23 ft^3 even if the water levels are zero, causing an over estimation in runoff volumes. Additionally, the bubbler module deployed had increased uncertainty with depth levels below 0.03 inches. Taking these two factors into consideration, hydrographs were also determined only using the volumes once rain caused the level to reach above 0.031 inches, until the level was below 0.035 inches for five consecutive readings. The control watershed had an average volume of 8,650 ft^3 while the treatment watershed had 7,805 ft^3 on average for a difference of 845 ft^3 or 10 % (Table 8). Neither categorization of total runoff volumes was statistically different, but the large degree of variance between storm events limits the statistical power when comparing the runoff datasets. While statistical differences between the runoff volumes did not exist, a variety of trends did occur based on storm characteristics.

Table 8: Summary table of total runoff depth categorization (n=42)

	Control	Treatment	Difference	Percentage
Mean	8650	7805	845	10
Median	4821	5433	-240	-8
Std. Dev.	21865	14070	8275	58
Maximum	146240	93756	52484	67
Minimum	161	372	-4713	-213
Std. Error	3415	2197	1292	9
T-Test	0.258	-	-	-

Analyzing the performance of the treatment watershed versus the control watershed and differences in runoff volume over time appears to show an increasing trend in LID BMP effectiveness as the study progresses. However, none of the trends were significant do to the great variability in storm and runoff characteristics.

3.1.5 Effects of Storm Characteristics

Analyzing performance under different rainfall conditions can provide insight as to if certain storm characteristics are causing changes in LID BMPs' effectiveness. Runoff volume and the difference between control and treatment volumes are weakly correlated with total rainfall and rainfall intensity. While there is a general increasing trend in the treatment runoff reduction as rainfall totals and rainfall intensity increase, the correlation between both variables and reduction is extremely weak. Runoff reduction percentage also has similar trend with a slight increase as total rainfall and intensities increase but weak correlation. These results are to counter what many other studies have found (William and Bannerman 2008; Hood et al. 2007). Many studies report the volume reduction based on inflow with small storms having 100 percent reductions and larger storms having small reductions (Taylor 2013). Taylor (2013) suggests the single volume percentage reporting is more of a weighted average flow reduction corresponding to LID storage capacity and frequency distribution of storm sizes, rather than a measure of the LID effectiveness. A North Carolina residential LID subdivision on low permeable soils saw no difference in runoff volume compared to a conventional subdivision while a LID subdivision with more permeable soils in coastal Connecticut had substantially less annual runoff than a nearby control neighborhood (Hood et al. 2010; Bedan and Clausen 2009). Site characteristics heavily influence volume of runoff and LID effectiveness in treating runoff, so it is important to consider when comparing LID impact across types of storm events. A Wilmington, Massachusetts LID retrofitted neighborhood was most effective for small storms, reducing runoff for storms with less than 6.35 mm of total rainfall but not more (Zimmerman 2010). Others have been effective for all storm types. For example, a coastal Connecticut LID subdivision significantly decreased runoff compared to a traditional subdivision for storms with less than 25.4 mm of total rainfall as well as storms greater than 25.4 mm (Hood et al. 2007). However, using the individual LID structures that comprise the implemented LID in basin wide studies shows they become less effective with greater rain totals (Ahiablame et al. 2012; Debusk and Wynn 2011; Chapman and Horner 2010; Hunt et al. 2012; Collings et al. 2008). The idea is that once storage capacity becomes exhausted a rainfall-runoff relationship similar to traditional development occurs (Davis et al. 2010; Taylor 2013). This is contrary to the treatment watershed trends that had less of an impact for smaller runoff events characterized by less intense storms.

Table 9: Summary statistics of storm characteristics

	Total Rainfall (in)	Five-Minute Rainfall Intensity (in/hr)	Antecedent Dry Period (days)
Mean	0.75	0.89	6.10
Median	0.39	0.48	4.00
Std. Dev.	1.56	0.94	5.62
Maximum	10.16	4.55	26.00
Minimum	0.05	0.12	1.00
Std. Error	0.24	0.14	1.99

Over the course of the study, a stronger correlation existed between higher rainfall intensities and totals to LID BMP runoff effectiveness than smaller rainfall totals or intensities, which is opposite the findings of many LID studies (Selbig and Bannerman 2008). The combination of LID BMPs was more effective with high intensity storms removing both greater volumes and a greater percentage than less intense storms.

The smaller difference between control and treatment runoff and percent reduction for the smaller storms could be partially attributable to the design and placement of the LID BMPs. Any rainfall on the driveways and roads flow into the flume with the only potential LID contact being the permeable pavement. While the permeable pavement had a potential storage capacity of 291 ft^3, there was a constant reserve level. The depth to water level was measured over a two-month period with a variance between 8 ¾ inches and 12 ¾ inches with 8 ¾ being shortly after a storm and 12 ¾ being after an antecedent dry period of 16 days. Line and White (2016) studied a subdivision in the Piedmont region of North Carolina where LID measures made no difference in runoff reduction. The lack of impact was attributable to conventional grading during construction and low natural permeability soils (Line and White 2016).

Cleveland County Oklahoma is characterized by clayey and humus rich soils on very gentle slopes with predominantly Mollisols and Alfisols soil types (Carter and Gregory 2008) With low permeability clay soils and due to delays in construction, the permeable pavement had large amounts of topsoil from nearby lots wash over and into the flume before sampling for the study began. These two factors could explain the lack of draining of the permeable pavement reservoir. While these results are atypical for other developmental scale LID implementation projects, in a climate characterized by weather extremes with long periods of drought and short periods of intense wet weather, it could be an unexpected benefit. While parking lots, highways and other high pollutant areas capturing 75% of events would be desirable, during extreme weather events when water is scarce, and the streams, ponds, and lakes desperately need water it could be advantageous especially if pollutant loads were still decreased (Hunt et al. 2002, Reese 2009, Taylor 2013).

A series of storm events occurring May 5th through May 11th, 2015 provided an ample opportunity to analyze LID effectiveness during unusually wet conditions. The total rainfall of 10.16 inches is more than five times any storm occurring during the study, while the maximum 5-minute intensity of 4.55 in/hr is nearly double (Table 10). Not only was the storm an outlier for the study, but also a rare event for Oklahoma historically. The rainfall totals ranged from statistically occurring every 2 years (0.86 in in 0.25 hour) to every 50 years (10.5 in in 168 hours) with a majority having a 5 to 10-year reoccurrence intervals (Table 11) (Tortoreli et al. 2005).

Table 10: Rainfall intensity and total rainfall for the May 2015 storm divisions

	Five-Minute Intensity (in/hr)	Total Rainfall (in)
Storm 1	3.36	3.94
Storm 2	4.55	2.41
Storm 3	1.68	0.68
Storm 4	3.60	1.90
Storm 5	2.28	0.98

Table 11: Total rainfall for different time intervals during the May 5th through May 11th, 2015 series of storm events and the historic return interval (Tortoreli et al. 2005).

Duration (hour)	0.25	0.5	1	2	3	6	12	24	48	72	168
Max Total Rainfall (in)	0.86	1.5	2.3	2.5	2.7	3.0	3.8	5.1	6.3	8.1	10.5
Reoccurrence interval (year)	2	5	10	5	5	5	10	10	25	25	50

These large, high intensity storms saw 140,000 cubic feet of runoff exit the control watershed flume and 87,000 cubic feet exit the treatment watershed flume. The LID BMPs design effectively reduced runoff by 55,000 cubic feet, a 39% reduction, over the course of the entire storm event. Dividing the event into singular events by selecting the midpoint between peaks shows similar results with the treatment watershed having reduced runoff totals between 1,700 cubic feet (19%) up to 19,000 cubic feet (42%) (Table 12).

Table 12: Runoff totals and differences for the May 2015 series of storm events with midpoint before peak to midpoint after peak used as divisions.

May 5th- 11th Runoff	Storm 1	Storm 2	Storm 3	Storm 4	Storm 5
Control Watershed	41857	45794	8524	34409	12081
Treatment Watershed	25181	26774	6866	19651	8890
Difference	16676	19020	1658	14759	3191
Percent Reduction	39.84	41.53	19.45	42.89	26.42

LID BMPs in the treatment watershed maintained their hydrologic effectiveness throughout the entire early May storm event with highest total reduction occurring on the 2nd storm and the highest percentage occurring the 4th storm (Table 12). Examining total runoff reduction and percent runoff reduction for the series of storm events shows a closer correlation of hydrologic effectiveness to rainfall characteristics than recent conditions.

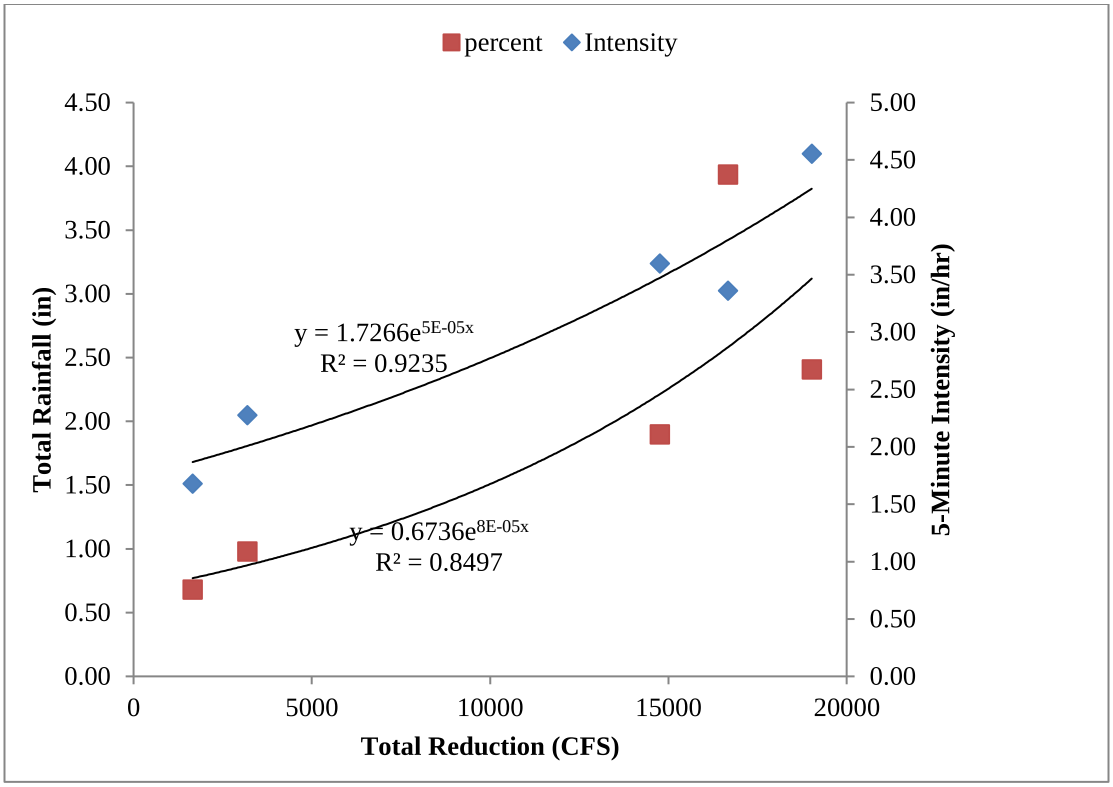

Figure 21: Differences in total runoff between the treatment and control watershed for the historic May 2015 event

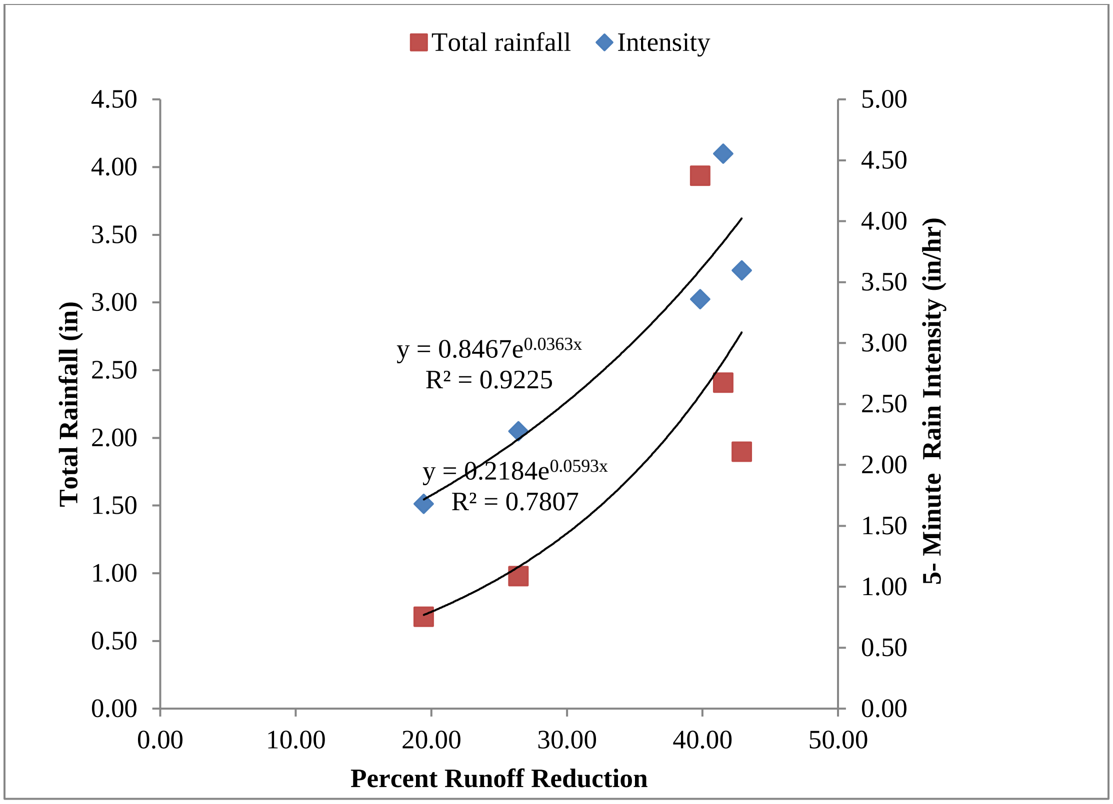

Figure 22: Runoff reduction percentage compared to total rainfall and rainfall intensity

Relationship between total runoff reduction (r^2=0.84) and percent reduction (r^2=0.78) have an exponential trend with total rainfall and rainfall intensity (r^2=0.92 for both watersheds) (Figures 21-22). The early May storm event is representative of an irregular historical event yet, the LID BMPs of the treatment watershed effectively decreased urban runoff. These results are contrary to the results of many other studies (Gilroy and McCuen 2009; Damodarm et al. 2010; Ackerman and Stein 2008). The general findings indicated LID BMPs performance was skewed because of the high-density LID BMP implementation in small catchment areas often caused an increase in the number of smaller storms producing no flow but were ineffective for larger flood causing events (Taylor 2013; Damodaram et al. 2010). However, as evident by the treatment watershed's increased effectiveness during the entire multiday event compared to the traditional watershed during an historical, potential flood-causing event, it appears Trailwoods LID BMP's may be functioning differently than LID BMPs in other studies. One potential source of improved performance could be similar to what Ackerman and Stein (2008) observed, i.e.: LID BMPs operating in series may increase performance.

3.1.6 Peak Discharge

While total runoff shows the amount of water moving through the system, peak discharge provides insight into the speed water moves. The higher the peak discharge the faster water moves through, giving it less time to infiltrate and increasing potential degradation. The mean peak discharge of the 42 events was significantly different ($p < 0.05$) for the treatment watershed (0.86 CFS) than the control watershed (1.52 CFS) Not only was it significantly different, the average treatment watershed peak discharge was 0.66 CFS less, for a 44% difference, in peak discharge compared to the control watershed (Table 13). Unlike total runoff, the treatment watershed impact on peak discharge was more consistent. Of the 42 events, only nine events had higher peak discharge rates for the treatment watershed than the control watershed, with all except two being less an 0.1 CFS difference and the other two being 0.13 CFS and 0.15 CFS, respectively. These events were characterized by smaller storms with significantly different ($p < 0.005$) intensities and total rainfall amounts ($p < 0.05$), but similar antecedent dry periods.

Table 13: Peak discharge summary statistics (n=42)

	Control	Treatment	Difference	Percent
Mean	1.52	0.86	0.66	20.
Median	0.35	0.30	0.09	29
Std. Dev.	3.09	1.59	1.53	34
Maximum	16.89	8.20	8.69	59
Minimum	0.06	0.07	-0.15	-111
Std. Error	0.48	0.25	0.24	5
T-Test	0.003			

Comparing storm characteristics to differences in peak discharge between the treatment and control watersheds and the difference as a percentage of the control peak discharge, shows a stronger correlation than volumetric reductions. While the antecedent dry period had an extremely weak influence on either volume reduction or discharge rates (r2=0.009 and r2=0.027) and the percent different compared to the control values (r2=0.002 and r2=0.005), the stormwater characteristics of total rainfall and intensity had much stronger correlations. One interesting trend with antecedent dry period is a negative correlation between dry periods and difference in peak discharge rates. The longer it has been since the previous rain event the more similar the discharge rates of the control and treatment watersheds become. This could be caused by low soil moisture content allowing more runoff to infiltrate into the soil than during wetter periods.

3.1.7 Runoff Duration

Runoff duration had an increasing trend as the study progressed for both watersheds although the trend was weak in both treatment (r^2=0.094; slope of 44.205) and control results (r^2= 0.0089; slope of 13.336). The variation in degree of increased runoff duration also means the difference between them increased (r^2= 0.3027; slope of 30.87). Over the course of the study, runoff took on average 322 minutes or 20%, longer to finish moving through the treatment watershed (Table 14). The increased runoff duration on the treatment side was significantly ($p < 0.05$) longer than what occurred on the control watershed (Table 14). Increased runoff duration has the potential for positive hydrologic and ecosystem impacts by increasing opportunities for infiltration, evaporation (Jacobson 2011; Jennings and Jarngin 2002).

Table 14: Runoff duration statistical summary

	Runoff Duration (Minutes)			
	Control	Treatment	Difference	Percent
Mean	1612	1935	322	20
Median	1062	1575	121	13
Std. Dev.	1710	1810	668	100
Maximum	7636	8444	3115	395
Minimum	40	131	-1064	-70
Std. Error	264	279	103	15
P Value	0.002			

3.2 Water Quality Data

The second component of determining LID BMP effectiveness was through water quality comparisons. The water quality data were compared between watersheds for the entire course of study, as well as for different storm event characteristics. Due to collecting the first flush, and the variability in first flush runoff volumes and timing, limited volumes of stormwater runoff were collected for certain events. In those events the collected volume was divided to attempt sampling of the maximum number of water quality parameters.

3.2.1 Total Suspended Solids

A total of 16 events were sampled for TSS with the control watershed having a higher TSS concentration for 11 of the events. The mean concentrations were 95.81 ± 39.90 mg/L for the control watershed and 65.12 ± 32.77 mg/L for the treatment watershed for a difference of 30.69 ± 7.12 mg/L. The treatment watershed TSS concentrations were significantly different ($p<0.05$) than the control watershed on average by 32% (Table 15). Figure 23 shows the differences in TSS concentration ranges between the control watershed and treatment watershed. These results are mixed compared to other catchment wide studies. While the 32% average reduction in TSS concentrations is more than the 14.3% Cheng et al. (2003) observed, it is not nearly as efficient as some other studies where TSS removal efficiencies ranged from 54% (Clausen 2007) to 79% (Line and White 2015). However, as White and Line (2015) discussed, maintaining and enhancing soil permeability is a critical component of LID effectiveness that is often altered during the construction process. The original soil structure and permeability may only return to its preconstruction state after a prolonged period (White and Line 2015).

Table 15: Summary statistics for TSS (n= 16)

	TSS Concentration (mg/L)		
	Control	Treatment	Difference
Mean	95.81	65.12	30.69
Median	54.45	31.41	23.04
Std. Dev.	159.60	131.10	28.50
Maximum	698.80	565.60	133.20
Minimum	4.10	5.97	-1.87
Std. Error	39.90	32.77	7.12
P Value	0.011	-	-

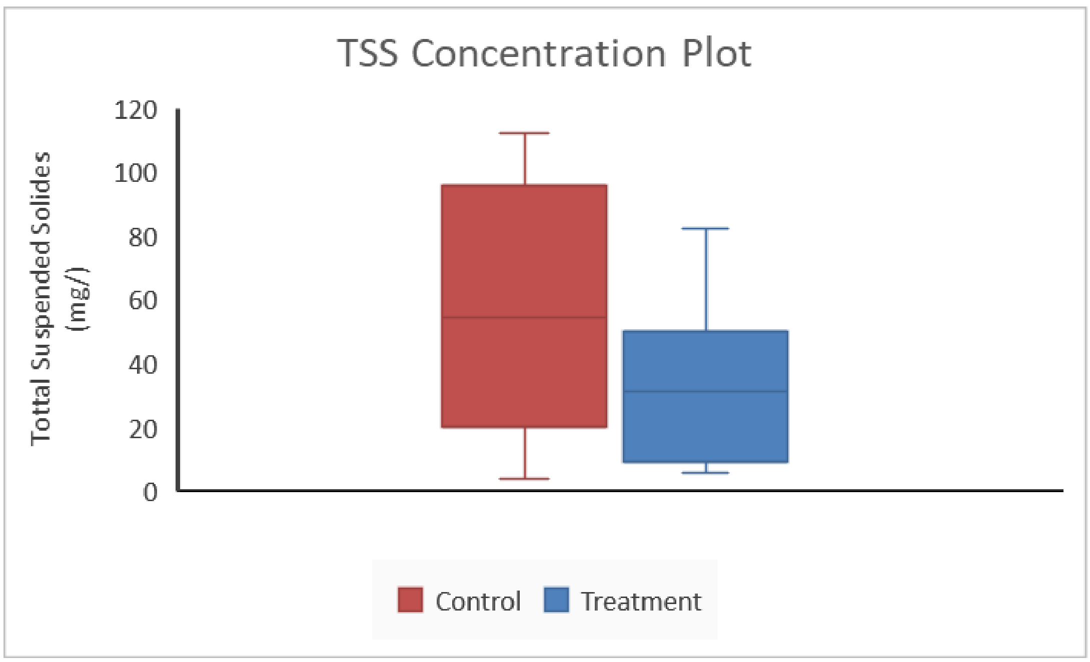

Figure 23: Box and Whisker plots for TSS

Stormwater samples were analyzed for three different nitrogen components: TN, NH_3-N, and NO_3-N (Tables 16- 18). None of the nitrogen compounds were significantly different between the watersheds ($p < 0.05$). The treatment watershed had three events of greater TN concentrations, four events of greater NH_3-N concentrations, and six events of greater NO_3-N concentrations than the respective control concentrations. Figures 24-26 show the Box and whisker plots for nitrogen compounds.

There is wide variability in the literature regarding nitrogen compound concentrations in LID BMPs with differences ranging from 70% reduction in NH_3-N and 78% reduction in NO_3-N (Clausen 2007) to a 2.74% reduction in TN and a 34.8% reduction in NO_3-N (Cheng et al. 2005) to no difference in TN, but increases in NH_3-N and NO_x-N (Line and White 2015). The large degree of variability in recent literature highlights mechanisms influencing LID BMP's abilities to decrease nitrogen species concentrations leaving the watershed. One mechanism is that permeable pavement with water storage capabilities provides a suitable environment for nitrifying bacteria to convert total ammoniacal nitrogen (TAN) into $NO_{2,3}$-N and therefore increases the nitrate concentrations (Braswell et al. 2018). Bioretention cells can also play a role in nitrogen exports. While ammonia is generally removed through cation exchange, if a media filter dries out the opportunity for biological nitrification increases (Chen et al. 2013). Additionally, since nitrate and nitrite compounds have poor adsorption to a soil, they can easily pass straight through the bioretention cell unless specific design strategies like increasing retention time, creating anoxic soil conditions, or creating a saturated zone to promote microbial denitrification (Chen et al. 2013). While the study does not show large reductions in nitrogen compounds between the control and treatment, it also does not have an increase in nitrate caused by the LID BMPs.

Table 16: Summary statistics of total nitrogen (n=16)

	Total Nitrogen (mg/L)		
	Control	Treatment	Difference
Mean	9.31	7.93	1.39
Median	9.35	7.85	1.50
Std. Dev.	3.51	3.06	0.44
Maximum	15.90	17.20	-1.30
Minimum	3.6	3.8	-0.20
Std. Error	0.87	0.76	0.11
P Value	0.14		

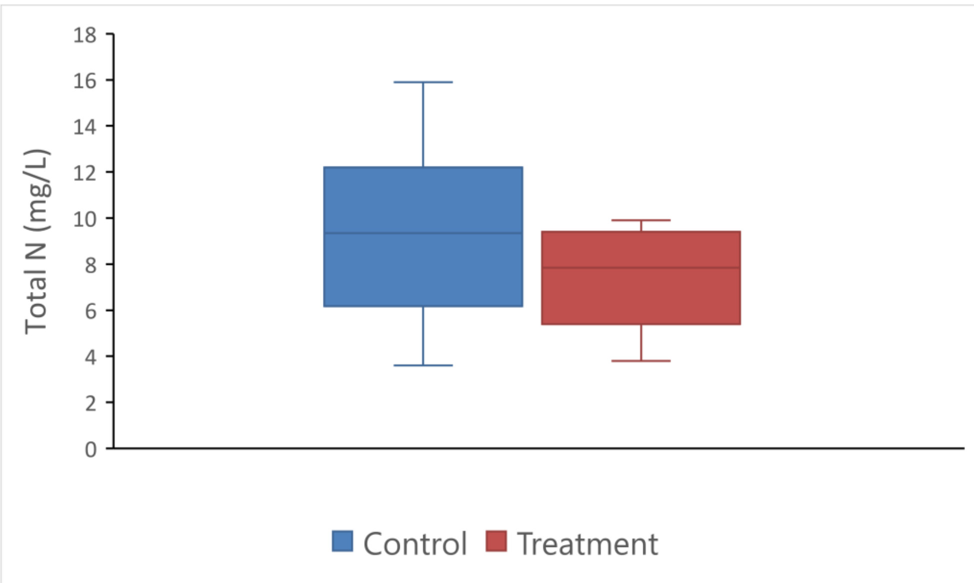

Figure 24: Box and whisker plot for total nitrogen concentrations (mg/L) with 16 samples for each watershed

Table 17: Summary statistics for ammonia (n=14)

	NH_3-N Concentration (mg/L)		
	Control	Treatment	Difference
Mean	4.1	3.5	0.6
Median	3.4	3.2	0.2
Std. Dev.	1.91	1.27	0.64
Maximum	9	7.6	1.4
Minimum	2.6	2.5	0.1
Std. Error	0.51	0.34	0.17
P Value	0.05		

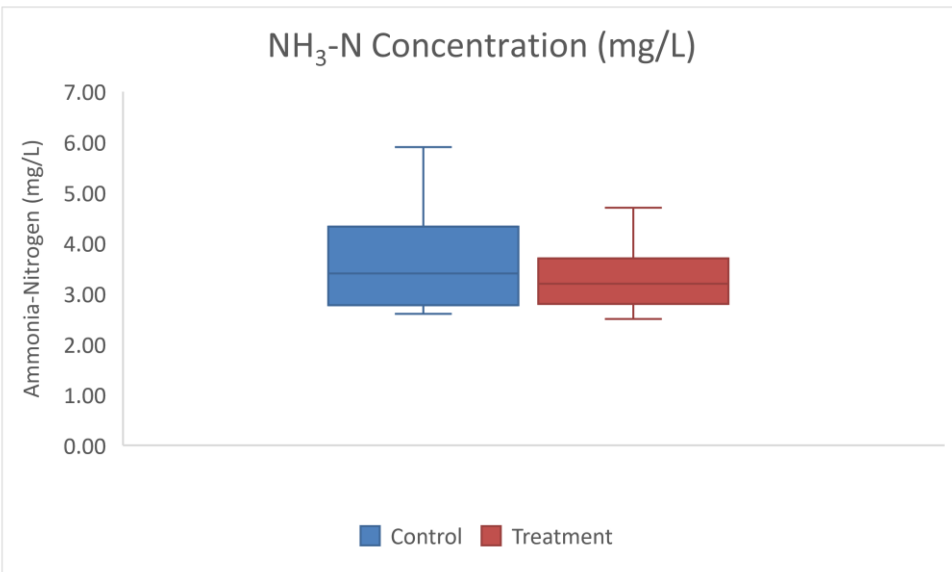

Figure 25: Box and Whisher plot for NH_3-N concentrations (n=16)

Table 18: Summary statistics for nitrate (n=16)

NO_3-N Concentration (mg/L)			
	Control	Treatment	Difference
Mean	3.01	2.26	0.75
Median	1.96	1.94	0.02
Std. Dev.	2.47	1.60	0.87
Maximum	7.93	7.06	0.87
Minimum	0.15	0.10	0.05
Std. Error	0.62	0.40	
P Value	0.10		

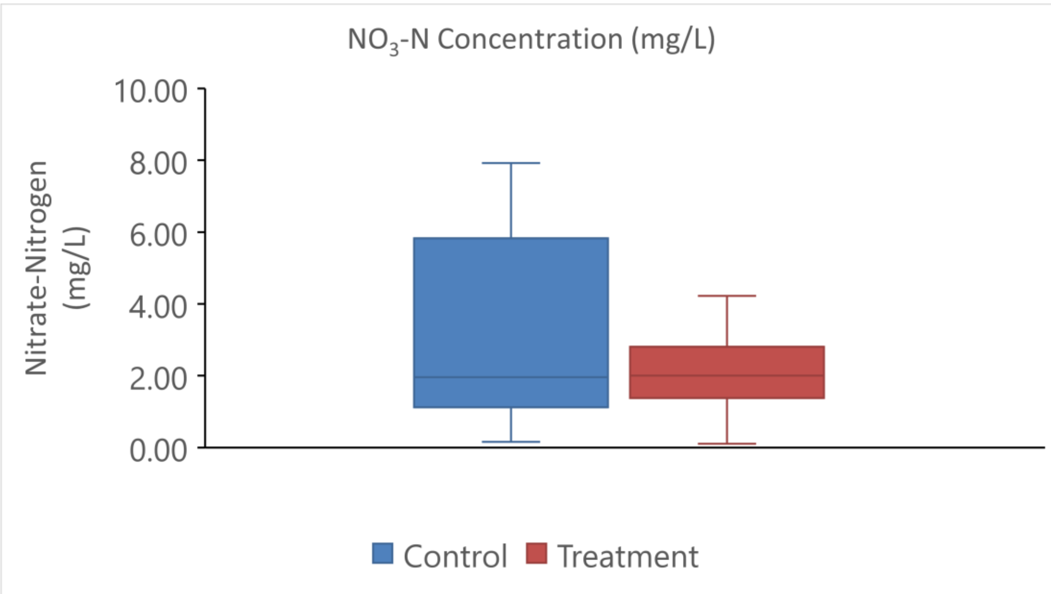

Figure 26: Box and Whisher plot for NH_3-N concentrations (n=16)

3.2.2 Phosphorous

The phosphorous compound analyzed during the study was DRP. DRP had an average concentration for the treatment side which was significantly greater ($p < 0.05$) than the average control DRP concentration (Table 19). Figure 27 shows that the treatment watershed produced higher concentrations than the control watershed. This is problematic for Lake Thunderbird since a key goal is to reduce the phosphorous load from the current approximate 18,000-23,000 kg/yr with the primary contributing source of this phosphorous loading coming from urbanization (OCC 2010). Although not specifically evaluated in this study, phosphorous likely leached from the rich organic substrate used in the rain gardens, contributing to these elevated concentrations

Table 19: Summary statistics for dissolved reactive phosphorous (n=15)

Dissolved Reactive Phosphorous (mg/L)			
	Control	Treatment	Difference
Mean	0.130	0.419	-0.289
Median	0.086	0.210	-0.124
Std. Dev.	0.139	0.460	-0.321
Maximum	0.493	1.641	-1.149
Minimum	0.003	0.007	-0.004
Std. Error	0.036	0.119	-0.083
P Value	0.022		

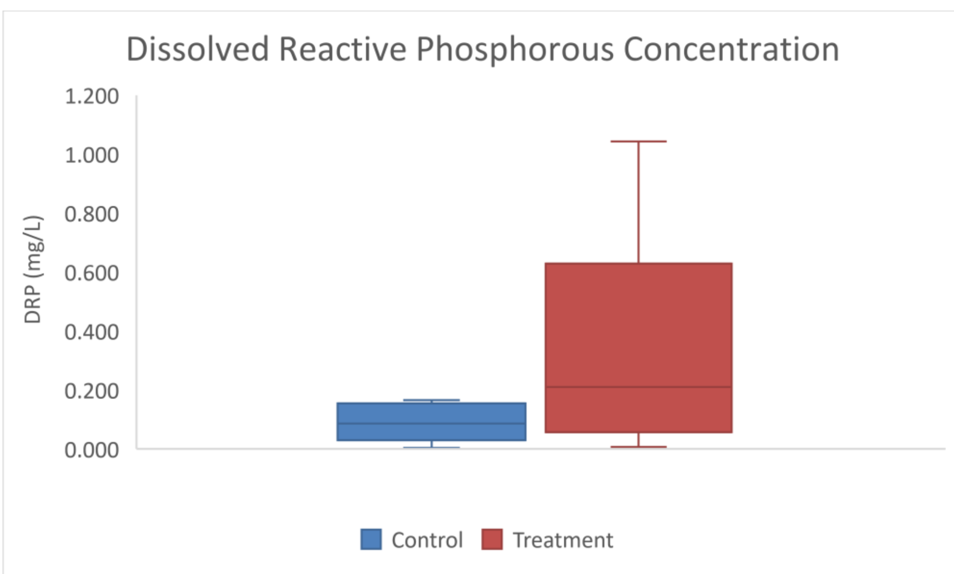

Figure 27: Box and Whisher plot for DRP concentrations (n=15)

3.2.3 Biochemical Oxygen Demand

The Biochemical Oxygen Demand (BOD) did not vary much for either watershed. Overall BOD concentrations of the watersheds generally mimicked each other with the average difference over the 11 sampled events being -0.13 mg/L higher on the treatment side (Table 20).

Table 20: Summary statistics for biochemcial oxygen demand (n=11)

Biochemical Oxygen Demand (mg/L)			
	Control	Treatment	Difference
Mean	9.90	10.03	-0.13
Median	9.68	9.95	-0.27
Std. Dev.	2.13	2.00	0.13
Maximum	12.94	12.69	0.25
Minimum	4.79	4.51	0.28
Std. Error	0.64	0.60	0.04
P Value	0.32		

3.2.4 Total Metals

Stormwater samples were analyzed for a suite of metals with minimal differences in most metal concentrations. Trace metal (Cd, Co, Cr, Cu, Ni, Pb, and Zn) concentrations showed the greatest differences, with Cr, Cu, and Zn having significantly higher concentrations ($p < 0.05$) in the treatment watershed runoff compared to the control. The only other metal with a significantly different concentration ($p < 0.05$) between watersheds was Ca, which had a higher concentration in treatment runoff than the control runoff, likely due to leaching from the rain garden substrate and permeable pavement. The differences in trace metal and cation concentrations between the watersheds can be seen in Figures 28-33.

Table 21: Summary statistics for Al, Ca, and Cd

Total Metal Concentrations (mg/L)									
	Al			Ca			Cd		
	Control	Treat.	Diff.	Control	Treat.	Diff.	Control	Treat.	Diff
Mean	4.06	2.30	1.76	22.15	25.65	-3.51	0.001	0.001	0
Median	2.99	1.79	1.21	14.12	21.01	-6.89	0.001	0.001	0
Std. Dev.	4.08	2.09	1.98	19.31	17.60	1.71	0	0	
Maximum	16.06	8.83	7.23	90.06	86.64	3.42	0.002	0.001	0
Minimum	0.04	0.24	-0.20	6.87	8.86	-1.99	0.001	0.001	0
Std. Error	0.94	0.54	0.40	4.43	4.55	-0.11	0	0	0
P-Value	0.076			0.023			0.145		

Table 22: Summary statistics for Co, Cr, and Cu

Total Metal Concentrations (mg/L)									
	Co			Cr			Cu		
	Control	Treat	Diff.	Control	Treat.	Diff.	Control	Treat.	Diff.
Mean	0.001	---	---	0.008	0.004	0.003	0.047	0.016	0.031
Median	0.001	---	---	0.005	0.003	0.002	0.030	0.013	0.017
Std. Dev.	0.001	---	---	0.007	0.003	0.004	0.042	0.011	0.031
Maximum	0.003	---	---	0.028	0.015	0.013	0.160	0.037	0.123
Minimum	0.001	---	---	0.002	0.002	0.000	0.007	0.004	0.002
Std. Error	0.000	---	---	0.002	0.001	0.001	0.01	0.003	0.007
P-Value	---	---	---	0.030	---	---	0.002	---	---

Table 23: Summary statistics for Fe, K, and Mg

	Total Metal Concentrations (mg/L)								
	Fe			K			Mg		
	Control	Treat.	Diff.	Control	Treat.	Diff.	Control	Treat.	Diff
Mean	3.75	2.10	1.65	4.37	4.07	0.30	2.87	3.19	-0.32
Median	2.16	1.20	0.96	3.75	3.84	-0.09	2.12	2.49	-0.37
Std. Dev.	4.30	2.77	1.53	2.59	1.47	1.12	2.29	2.27	0.02
Maximum	15.86	11.77	4.09	11.34	6.64	4.69	9.30	8.20	1.11
Minimum	0.04	0.17	-0.13	1.63	1.54	0.09	0.58	0.58	0
Std. Error	0.99	0.72	0.27	0.59	0.38	0.21	0.53	0.59	-0.06
P-Value	0.056	---	---	0.461	---	---	0.165	---	---

Table 24:Summary statistics for Mn, Na, and Ni

	Total Metal Concentrations (mg/L)								
	Mn			Na			Ni		
	Control	Treat.	Diff.	Control	Treat.	Diff.	Control	Treat.	Diff
Mean	0.11	0.06	0.05	4.03	4.36	-0.33	0.02	0.01	0.01
Median	0.06	0.03	0.03	3.84	3.35	0.49	0.02	0.01	0.01
Std. Dev.	0.14	0.09	0.05	1.65	3.01	-1.36	0.002	0.000	0.002
Maximum	0.50	0.38	0.12	7.64	11.67	-4.04	0.02	0.01	0.01
Minimum	0	0	0	1.92	1.52	0.40	0.02	0.01	0.01
Std. Error	0.03	0.02	0.01	0.38	0.78	-0.40	0.00	0.00	0.00
P-Value	0.055	---	---	0.369	---	---	---	---	---

Table 25: Summary statistics for Pb and Zn

Total Metal Concentrations (mg/L)						
	Pb			Zn		
	Control	Treat.	Diff.	Control	Treat.	Diff.
Mean	0.03	0.02	0.01	0.10	0.05	0.05
Median	0.03	0.03	0	0.05	0.03	0.02
Std. Dev.	0.01	0.00	0.01	0.11	0.05	0.06
Maximum	0.04	0.03	0.01	0.39	0.21	0.18
Minimum	0.02	0.02	0.00	0.01	0.01	0.00
Std. Error	0.00	0.00	0.00	0.03	0.01	0.02
P-Value	0.095	---	---	0.032	---	---

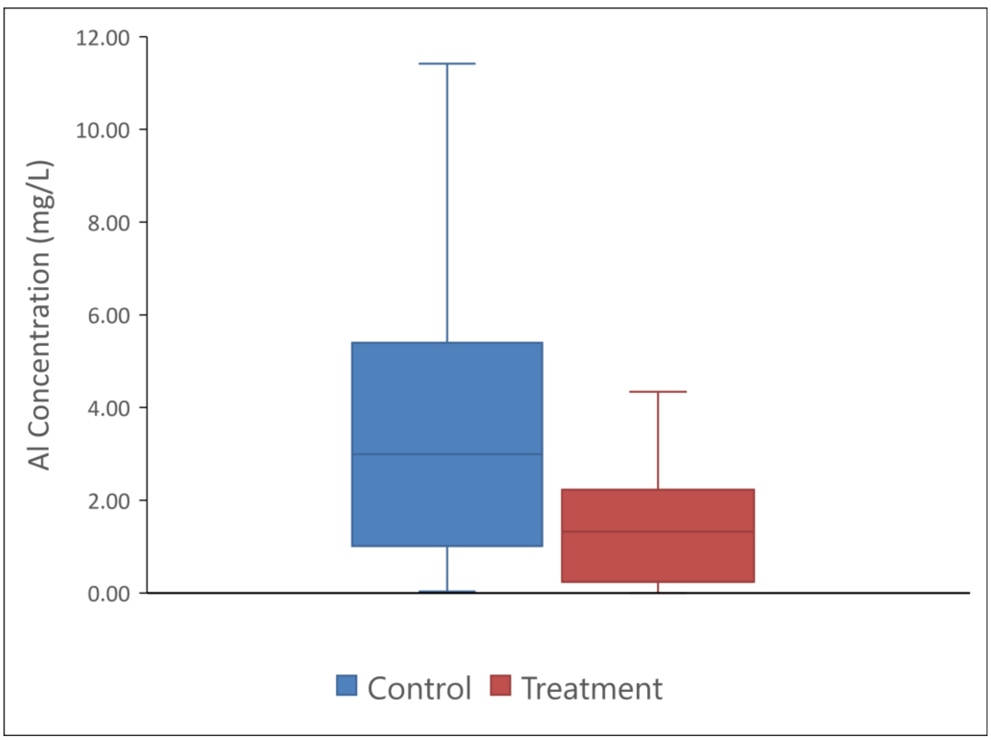

Figure 28: Box and Whisher plot for Al concentrations

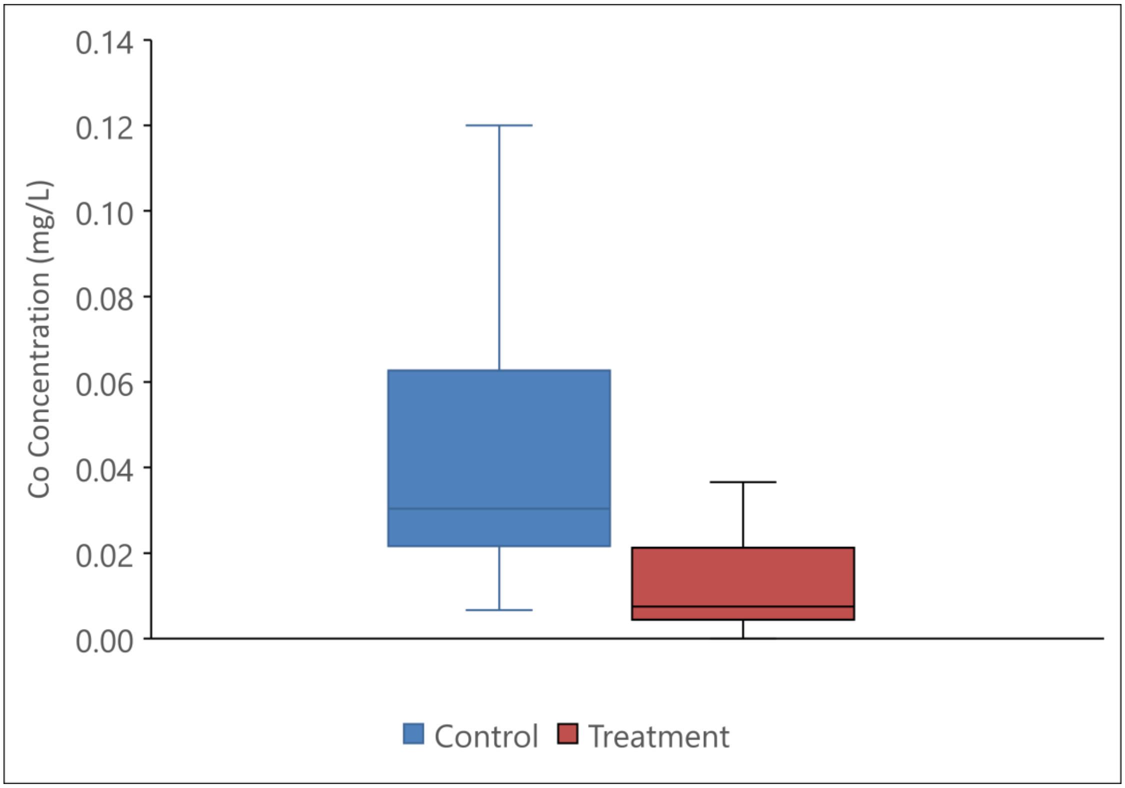

Figure 29: Box and Whisher plot for Co concentrations

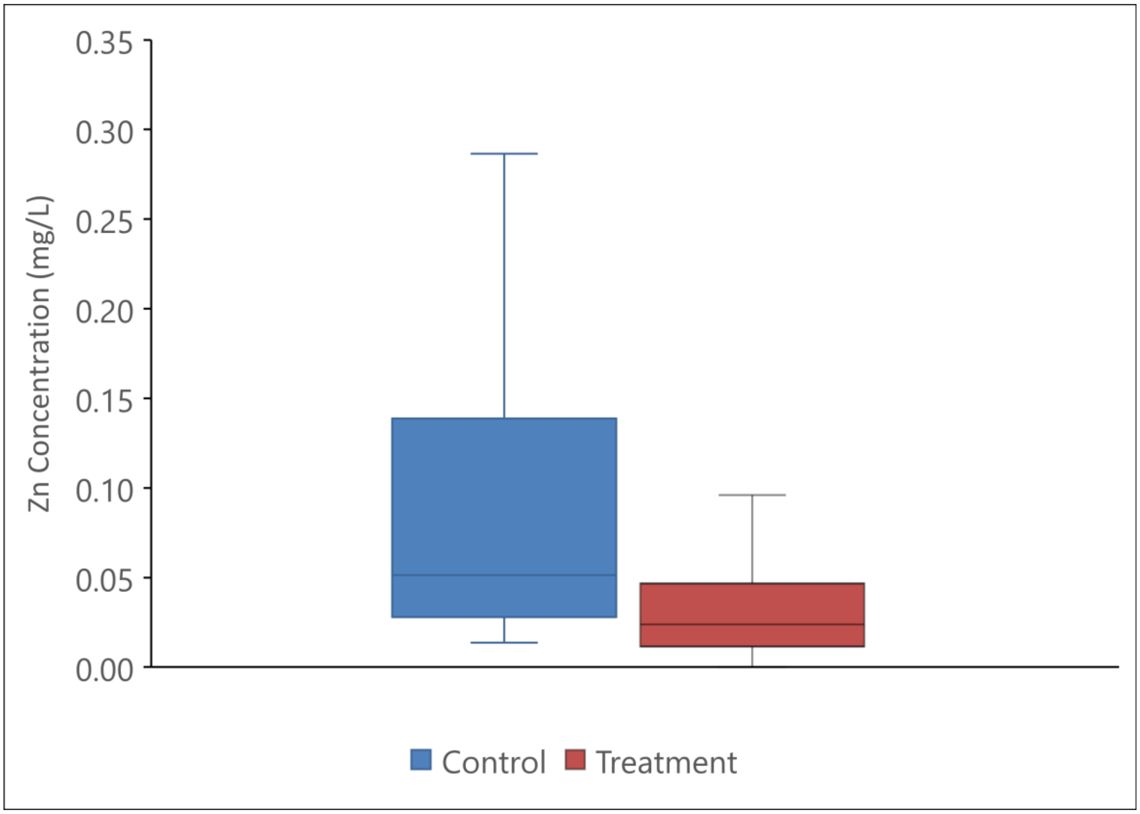

Figure 30: Box and Whisher plot for Zn concentrations

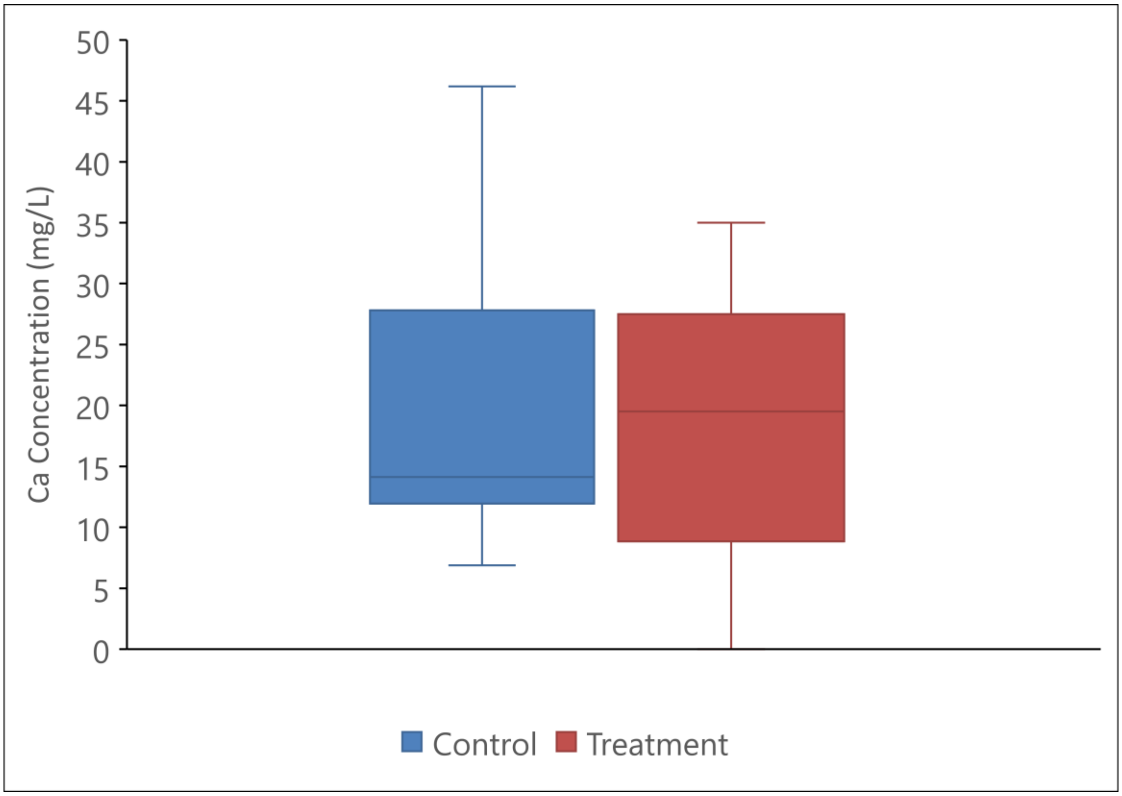

Figure 31: Box and Whisher plot for Ca concentrations

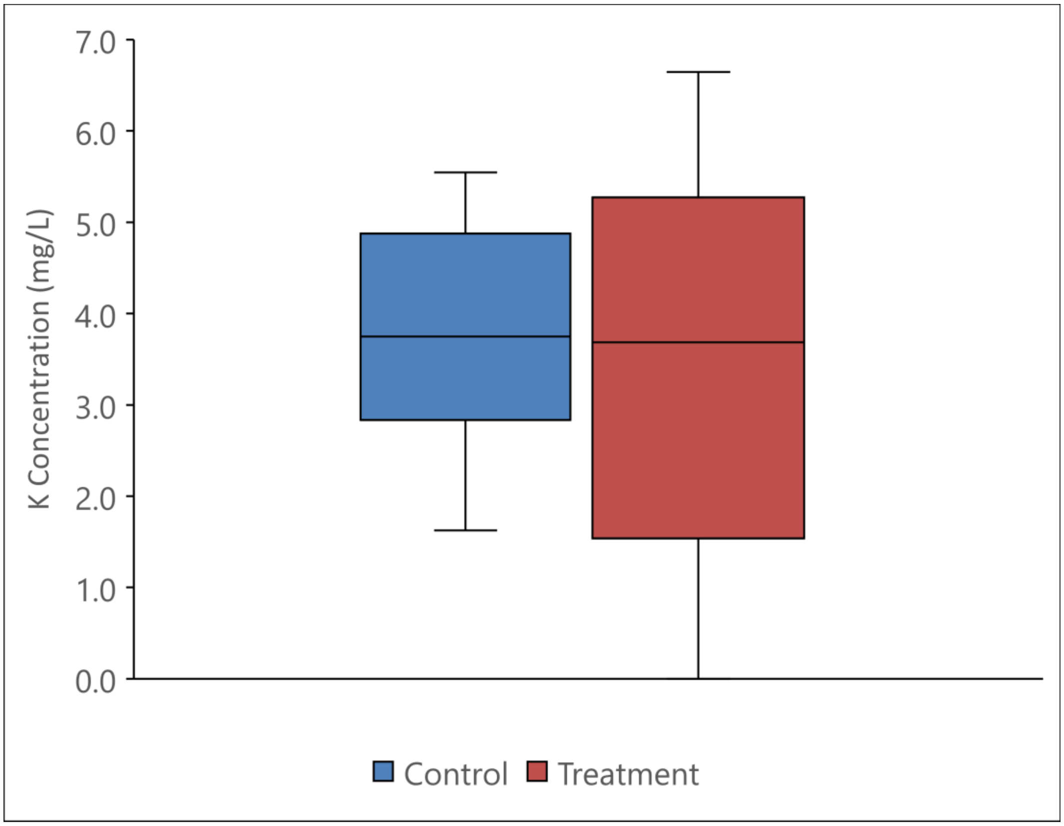

Figure 32: Box and Whisher plot for K concentrations

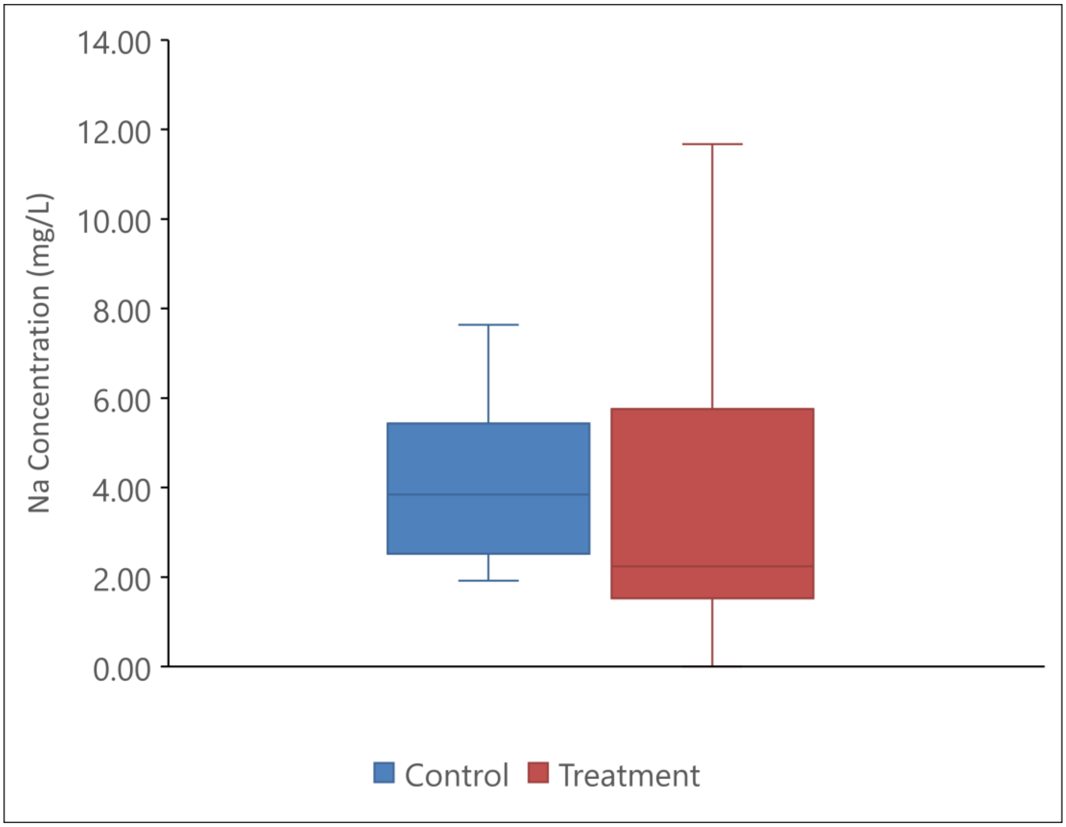

Figure 33: Box and Whisher plot for Na concentrations

3.2.5 Analysis of Water Quality Data by Storm Characteristics

In order to further analyze water quality differences, the data were divided into groups to determine LID BMP effectiveness with different storm parameters. The different storm parameters evaluated were total rainfall, rainfall intensity, and time since rainfall, and each highlight different aspects of the treatment watershed.

Storm event precipitation amounts were divided between < 0.39 in (n=3), 0.39 in-1.18 in (n=8) and >1.8 in (n=6) and a statistical comparison between control and treatment watersheds for each division. For storms < 0.39 in, no statistical differences for any of the analytes existed (Table 26). Storms with 0.39 in- 1.18 in of rainfall depth showed a statistically significant difference in pH ($p < 0.05$), but it was storms with >1.18 in rainfall depth that most significant differences existed between the control and treatment watersheds. TN, DRP, TSS, SPC, and TDS all showed $p < 0.05$ for >1.18 in storms. While there might not have been statistical differences in concentrations, there are numerous concentration reductions compared to the control. Total N, ammonia, and TSS all showed decreased concentrations for the treatment watershed for all three storm divisions. The treatment watershed produced more nitrate for < 0.39 in events, and greater TP and DRP concentrations for all three storm divisions.

Rainfall intensity divisions were < 1 in/hr (n=7) and > 1 in/hr (n=10). Statistical differences existed for DRP for high intensity storms and TSS for low intensity storms. The control watershed had produced greater quantities of TN, nitrate, ammonia, TSS, and peak discharge while TP and DRP were lower for low intensity but higher for high intensity storms (Table 27)

Table 26: Sampled storm event divisions by total rainfall

Total Rainfall		< 0.39 in		0.39 in- 1.18 in		>1.18 in	
	Units	Control	Treat.	Control	Treat.	Control	Treat.
Total N	mg/L	9.47	9.37	8.44	6.99	10.90	6.52
Nitrate	mg/L	2.32	3.43	2.94	2.30	1.80	1.70
Ammonia	mg/L	3.23	2.87	4.28	3.33	4.42	4.25
DRP	mg/L	0.001	0.03	0.09	0.30	0.06	0.51
TSS	mg/L	62.26	35.04	142.20	119.81	62.32	23.23
BOD	mg/L	8.62	9.78	10.16	10.24	8.58	8.60
pH		7.81	7.97	7.56	8.60	7.49	7.37
SPC		0.087	0.10	0.10	0.09	0.08	0.27
Cond		55.00	68.00	82.25	77.50	62.25	217.25
DO %		116.40	120.80	94.50	91.40	104.83	151.25
DO	mg/L	14.58	14.45	9.21	8.73	10.75	15.11
TDS		0.06	0.07	0.07	0.06	0.05	0.17
n		3	3	8	8	6	6
Peak Discharge	CFS	2.90	1.83	3.49	2.14	4.08	1.78

Table 27: Sampled storm event divisions by rainfall intensity

Rainfall Intensity		low (< 1in/hr)		high (>1in/hr)	
	Units	Control	Treatment	Control	Treatment
Total N	mg/L	8.56	6.87	9.66	8.74
Nitrate	mg/L	2.88	2.15	2.45	2.41
Ammonia	mg/L	3.70	3.05	4.22	3.91
Total P	mg/L	0.0526	0.0479	0.0743	0.0977
DRP	mg/L	0.3019	0.1541	0.0927	0.6408
TSS	mg/L	60.90	28.28	48.35	31.65
BOD	mg/L	10.01	10.60	9.44	9.55
pH		7.32	8.77	7.65	7.60
Spc		0.0767	0.1217	0.0830	0.1703
Cond		65.67	102.33	62.57	134.57
DO %		106.73	113.23	98.38	125.48
DO	mg/L	10.63	11.28	10.39	12.72
TDS		0.0500	0.0790	0.0539	0.1089
n		7	7	10	10
Peak Discharge	CFS	1.35	0.98	5.16	2.67

Table 28: Sampled storm event divisions of antecedent dry period

		0-2 days		3-5 days		>6 days	
	Units	Control	Treat.	Control	Treat.	Control	Treat.
Days since rain	days	1.86	1.86	3.33	3.33	10.43	10.43
Total N_c	mg/L	8.23	7.58	11.27	7.67	9.96	6.96
Nitrate	mg/L	2.95	2.30	4.49	3.56	1.50	1.75
Ammonia	mg/L	4.14	3.9	4.47	2.93	3.9	3.53
Total P	mg/L	0.067	0.076	0.039	0.027	0.0756	0.096
DRP	mg/L	0.052	0.213	0	0.017	0.087	0.478
TSS	mg/L	44.04	22.93	80.92	41.48	160.14	119.06
BOD	mg/L	8.69	8.82	10.46	11.24	10.11	10.51
pH		7.47	8.11	X	X	7.62	7.78
Spc		0.0898	0.1398	X	X	0.0724	0.1716
Cond		73.60	106.6	X	X	53.4	143.2
DO %		89.32	96.18	X	X	115.98	152.93
DO	mg/L	9.04	9.75	X	X	12.25	15.35
TDS		0.058	0.089	X	X	0.047	0.111
n		7	7	3	3	7	7
Peak Discharge	CFS	3.4485	1.602	1.542	1.158	4.806	3.028

The time since rainfall was divided into three categories as well: 0-2 days (n=7), 3-5 days (n=3), > 6 days (n= 7). The only statistically significant difference found was for TN and DRP concentrations for events with more than 6 days since the previous storm (Table 28).

3.3 Retention Ponds

The paired watershed study compared runoff quantity and quality between LID BMP and traditional development practices, and inevitably runoff leaving both watersheds moved from the catchments to a nearby retention pond study. These retention ponds which are often required for residential developments provide a buffer between residential and roadway runoff and the lakes and streams that flow into Lake Thunderbird. Water quality parameters analyzed for the paired watershed study were also analyzed for the retention ponds. The retention pond sample collection started during a period of drier weather, with only one month having close to or above average total monthly rainfall amounts. Additionally, that specific month, November 2014, had a 24-hour period with 2.12 inches of total rainfall which was 86% of the historical monthly average and made up 60% of the month’s rainfall total (OCS 2015). These precipitation conditions forced a move of the water sampling intake tube from the bottom of the outflow structure of each pond to a position near the outlet structure but not physically in it, ensuring the potential for sufficient samples to be collected even if outflow did not occur during specific rain event. This fact, combined with not having influent water quality concentrations, makes extrapolation from collected sample to drainage basins condition more difficult. Since influent data were not generated, a direct comparison of concentrations entering and exiting each specific retention pond was not possible. Therefore, retention pond sampled nutrient concentrations were comparatively analyzed through normalization by physical characteristic ratios and in comparison, to the data from the national stormwater database (Pitts et al. 2008).

The four neighborhood retention ponds, Deerfield (DF), Carrington Lakes (CL), Shadow Lake 1 (SL1), and Shadow Lake 3 (SL3), had varying total drainage areas, pond surface areas, ratios of pond surface area to total drainage area, and percentage of imperviousness (Table 29). Additionally, Carrington Lakes had four rain garden LID BMPs installed within its 5-acre drainage basin (Coffman 2014). The normalization of retention ponds is seen frequently in the literature with characterizations such as permanent pool volume to average storm volume, water quality surcharge volume to permanent pool volume, permanent pool volume to average stormwater runoff for the area, pond surface area to watershed area, and permanent pool volume to watershed area (Brink and Kamish 2018; Barret 2008; Geosyntec and Wright 2013).

Table 29: Physical attributes of the four retention ponds, corresponding drainage area, and ratios used for normalization between retention ponds

	Deerfield	Carrington Lakes	Shadow Lake 1	Shadow Lake 3
Total area (acres)	45.4	3.59	47.32	15.23
Road (acres)	7.24	0.28	6.48	2.59
Road area (square feet)	315379	12120	282204	112820
Pond Size (acres)	2.33	0.19	2.43	0.35
Pond area (square feet)	101707	8426	105975	15466
Green Space (acres)	5.45	0.64	4.29	1.36
Houses and sidewalks (acres)	12.4	0.86	17.36	5.98
Total imperviousness (acres)	19.64	1.14	23.84	8.57
Imperviousness (%)	43	32	50	56
Pond/Total Area Ratio	0.050	0.054	0.054	0.023
Pond/Imperviousness Ratio	0.119	0.102	0.041	0.167
Pond/Green Space Ratio	0.428	0.297	0.566	0.257
Pond/2 Year Storm Volume Ratio	1.153	2.292	1.784	0.657

3.3.1 Water Quality Data

Comparing mean and median TSS values normalized by pond surface area to physical site characteristics did not show a correlation between TSS concentrations and pond surface area to total area, total imperviousness, green space or two-year storm volume design (Figure 34). These results are similar to other studies that have shown retention pond surface area and permanent pool volume to be not correlated with TSS removal capabilities (Park and Roesener 2013; Barret 2004; Barret 2008). Barrett (2008) states a strong correlation exists between influent and effluent TSS concentrations, and if storm events produced less runoff than the permanent pool, the effluent concentrations should be independent of influent. A lack of correlation exists between permanent pool volumes and mean TSS, total phosphorous, nitrogen species, and metal discharge concentrations when the average storm produces an equal or lesser volume than the volume contained in the retention pond permanent pool (Barret 2008). The lack of correlation found during this study could be attributable to the drier conditions preceding and during the beginning of sampling period causing pond volumes to be less than their permanent pool design or because of the specific storm events and the small number of samples obtained. Comparing ratios of physical drainage characteristics to constituent concentrations shows the biggest retention pond had the highest mean TSS concentration as well as the strongest response to pond surface area normalized by area of green space.

Table 30: Summary statistics for TSS concentrations of the four retention ponds

	Total Suspended Solids (mg/L)			
	DF	CL	SL1	SL3
Mean	13.67	21.40	105.76	9.44
Median	11.60	21.40	71.00	5.20
Std. Dev.	10.81	2.20	122.65	5.86
Maximum	32.40	23.60	370.40	16.80
Minimum	2.80	19.20	6.93	4.00
n	6	2	6	5
Std. Error	4.41	1.56	50.07	2.62

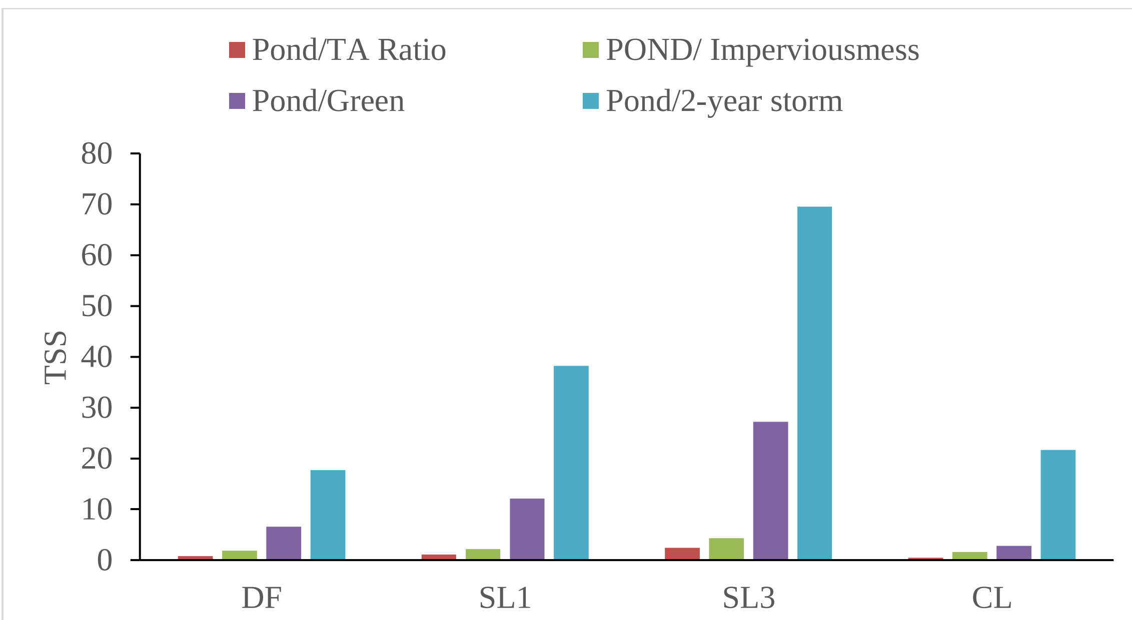

Figure 34: TSS concentrations normalized by ratios of pond surface areas to other drainage area physical characteristics

Nitrogen compound data are presented in Tables 31-33. The nitrogen concentrations were much lower than the nitrogen compounds found in the runoff from both watersheds of the paired watershed study. The ratio of pond surface area to total area had the strongest correlation to TN and ammonia for all retention ponds except SL1 which had a stronger correlation between surface area and imperviousness (Figures 35-36). Nitrate also followed a similar trend with a relationship between pond surface area and total drainage area for all retention ponds except SL1, which had a lesser response than it did with the other two nitrogen compounds (Figure 37).

Table 31: Retention pond summary statistics for total nitrogen

	Total N (mg/L)			
	DF	SL1	SL3	CL
Mean	2.4	3.5	3.8	1.8
Median	2.3	3.2	2.8	1.9
Std. Dev.	1.1	1.5	2.7	0.2
Maximum	4.6	5.9	9.0	1.9
Minimum	1.2	1.9	1.2	1.5
n	6	6	5	3
Std. Error	0.45	0.60	1.21	0.11

Table 32: Retention pond summary statistics for ammoina

NH_3-N (mg/L)				
	DF	CL	SL1	SL3
Mean	3.7	2.7	3.9	3.6
Median	2.7	3.0	3.8	3.8
Std. Dev.	1.7	0.6	1.0	0.4
Maximum	6.7	3.2	5.2	3.9
Minimum	2.2	1.8	2.2	2.9
n	6	3	6	5
Std. Error	0.70	0.36	0.40	0.16

Table 33: Retention pond summary statistics for nitrate

NO_3-N (mg/L)				
	DF	CL	SL1	SL3
Mean	0.172	0.056	0.089	0.167
Median	0.140	0.056	0.078	0.066
Std. Dev.	0.059	0.009	0.038	0.185
Maximum	0.288	0.066	0.153	0.527
Minimum	0.132	0.047	0.051	0.032
n	5	2	5	5
Std. Error	0.026	0.007	0.017	0.083

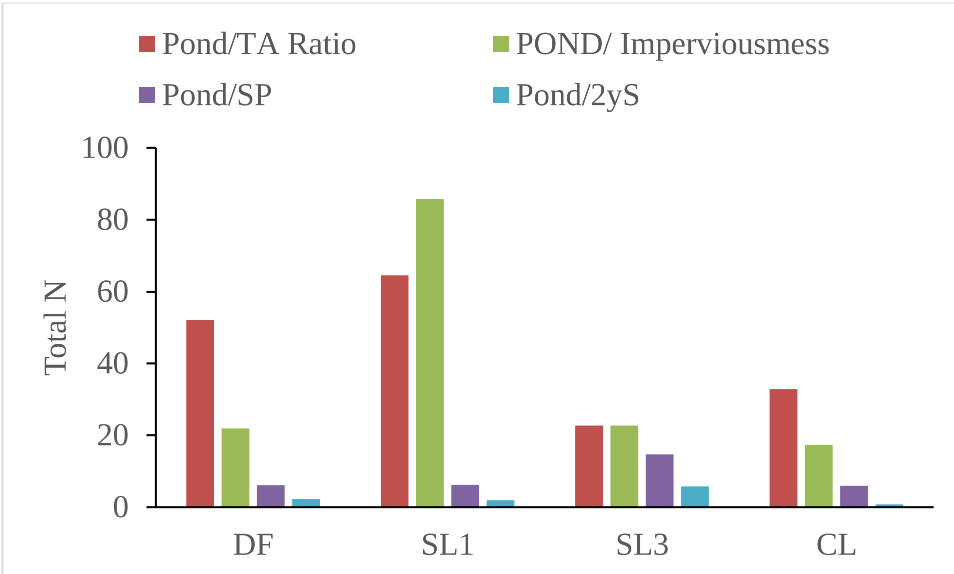

Figure 35: TN concentrations normalized by ratios of pond surface areas to other drainage area physical characteristics

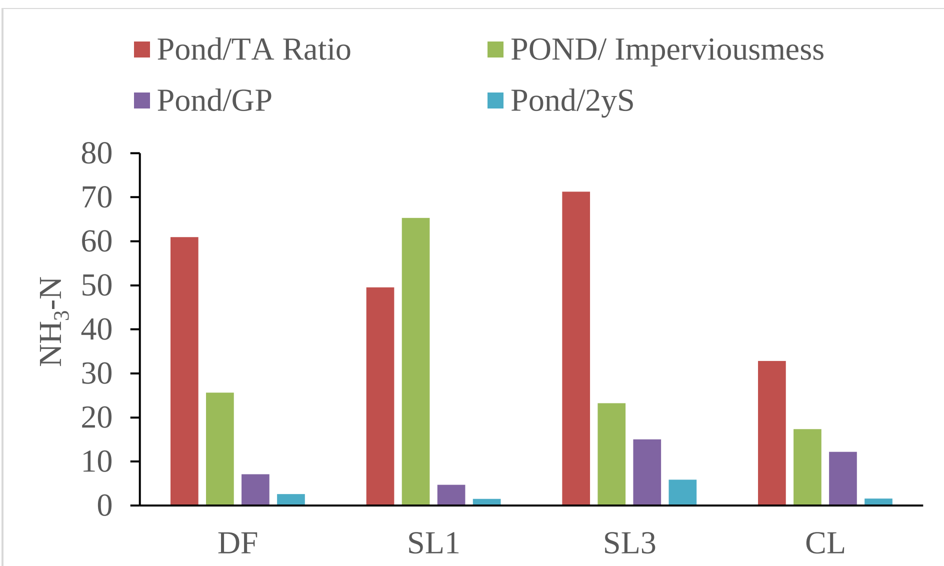

Figure 36: NH_3-N concentrations normalized by ratios of pond surface areas to other drainage area physical characteristics

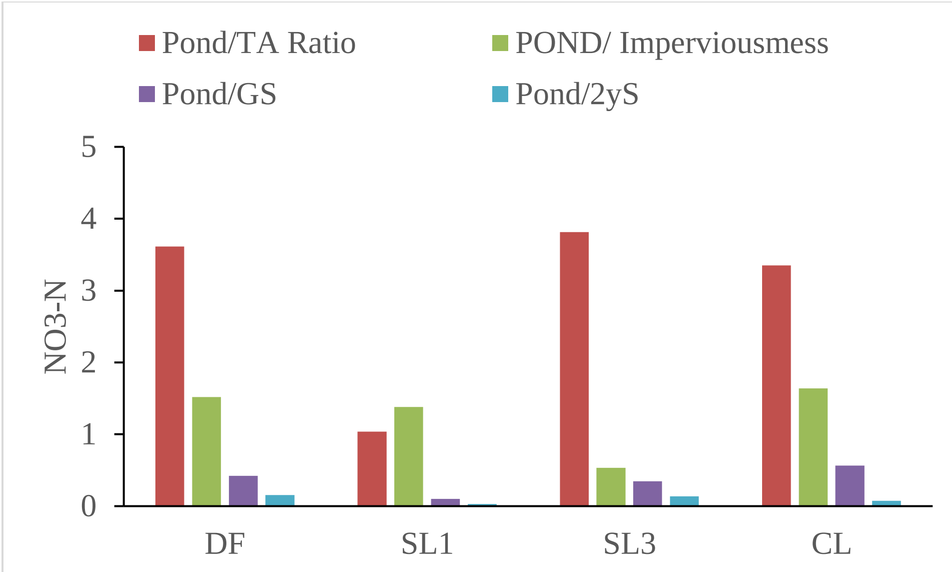

Figure 37: NO_3-N concentrations normalized by ratios of pond surface areas to other drainage area physical characteristics

The average DRP concentrations are shown in Table 40. The pond surface area to the designed storage volume for a storm with a two-year reoccurrence interval was the physical characteristic that had the most correlation with DRP concentrations (Figure 37). The lower phosphorous concentration for the Deerfield retention pond compared to the other retention ponds combined with the higher concentrations of nitrogen compounds suggests the retention pond is potentially phosphorous limited.

Table 34: Retention pond summary statistics for retention pond DRP concentrations

	DRP Concentration (mg/L)			
	DF	CL	SL1	SL3
Mean	0.057	0.141	0.109	0.199
Median	0.057	0.161	0.059	0.175
Std. Dev	0.044	0.038	0.132	0.090
Maximum	0.126	0.175	0.397	0.333
Minimum	0.001	0.088	0.014	0.097
n	6	3	6	5
Std. Error	0.018	0.022	0.054	0.040

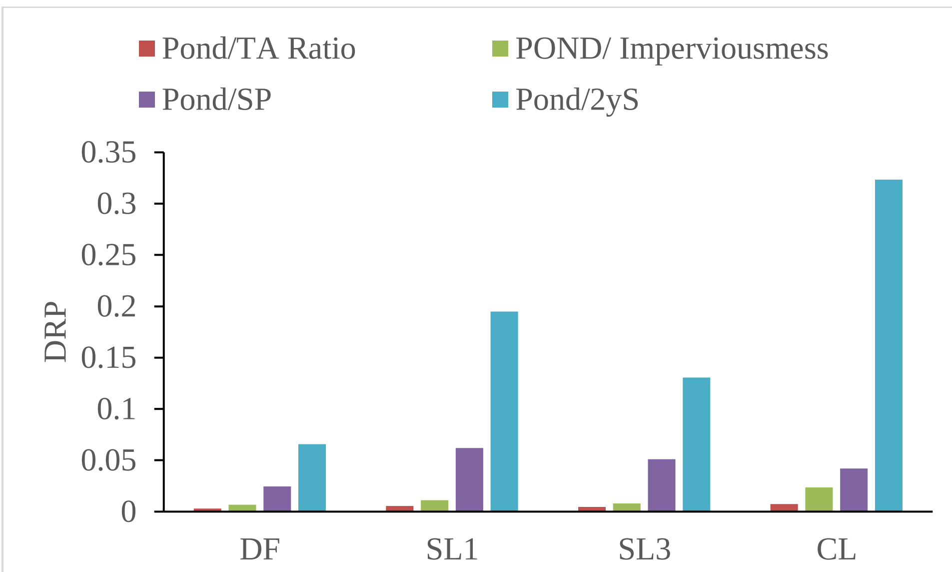

Figure 38: DRP concentrations normalized by ratios of pond surface areas to other drainage area physical characteristics

3.4 National Stormwater Database Comparison

Data from the National Stormwater Quality Database (NSD) was used to find comparable sites, providing a frame of reference for site conditions found in the study (Pitt et al. 2008). Using NSD, sites were sorted by principal land use then grouped by drainage area for direct comparisons to the study sites. The three groupings for the retention ponds were 40 to 50 acres for Deerfield and Shadow Lake 1, 15 to 20 acres for Shadow Lake 3, and 3 to 5 acres for Carrington Lakes (Figure 39). Comparing the retention ponds to NSD data, differences are seen in TSS, BOD, and metals concentrations (Figures 40-45). The only constituent with mixed results is total nitrogen. Carrington Lakes has substantially lower nitrogen concentrations then the similar sites in NSD data while Deerfield has similar concentrations and Shadow Lake 1 had higher than average concentrations (Figure 42).

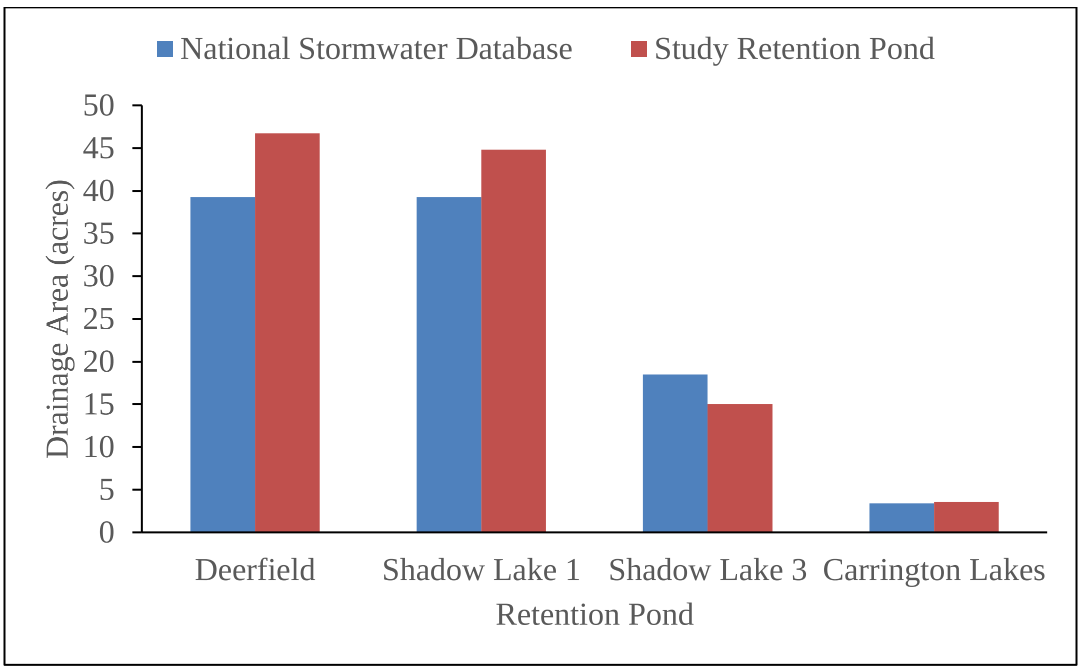

Figure 39: Drainage areas for the four study retention ponds and mean of comparable NSD sites

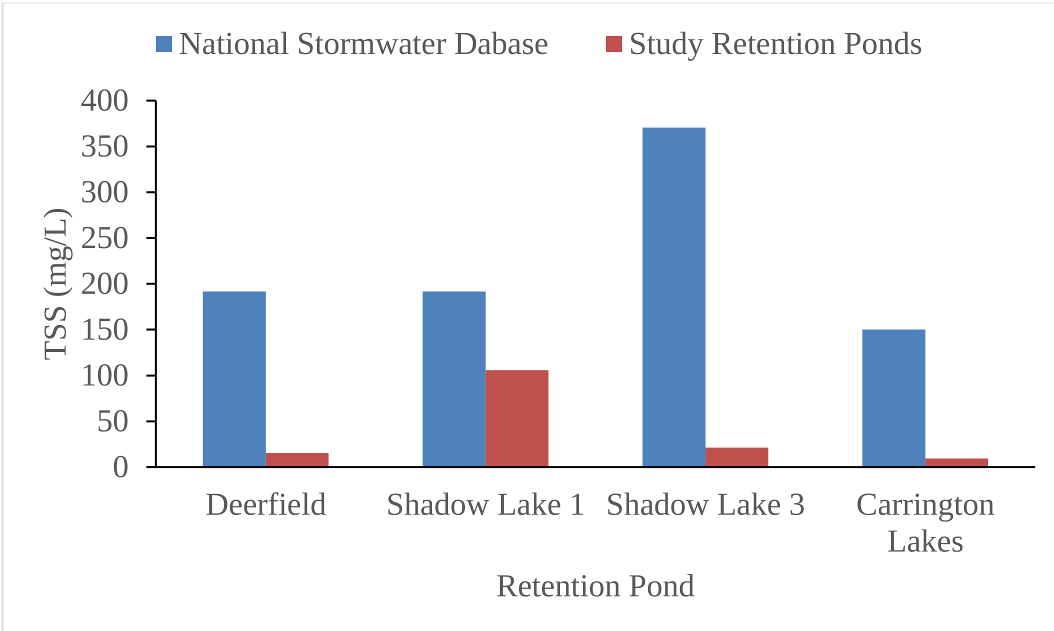

Figure 40: Mean TSS concentration comparison between NSD and study retention ponds

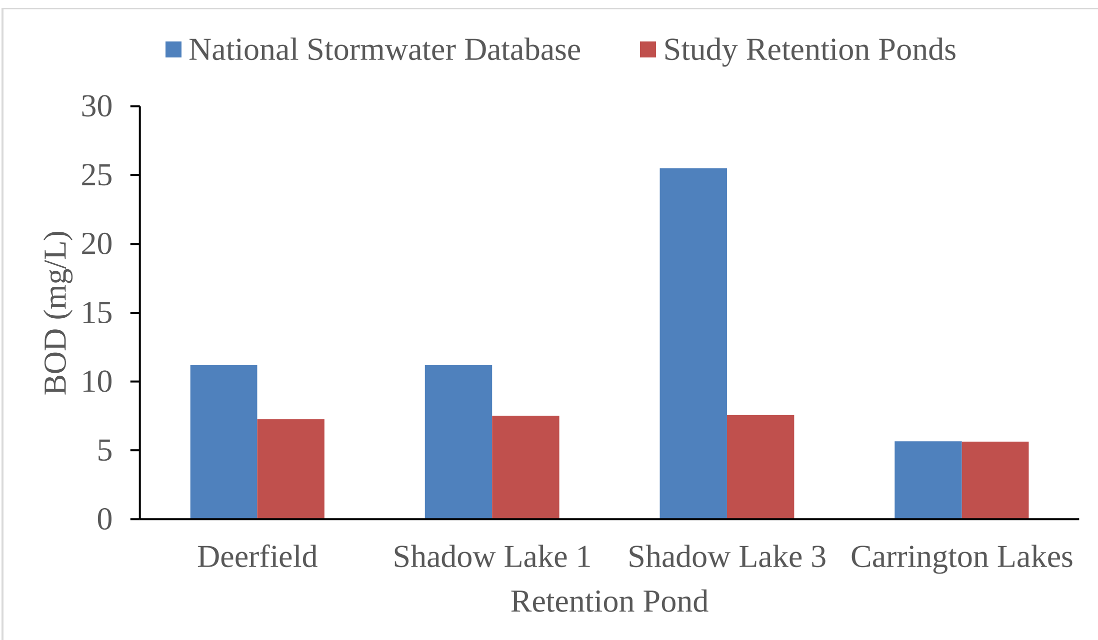

Figure 41: Mean BOD concentration comparison between NSD and study retention ponds

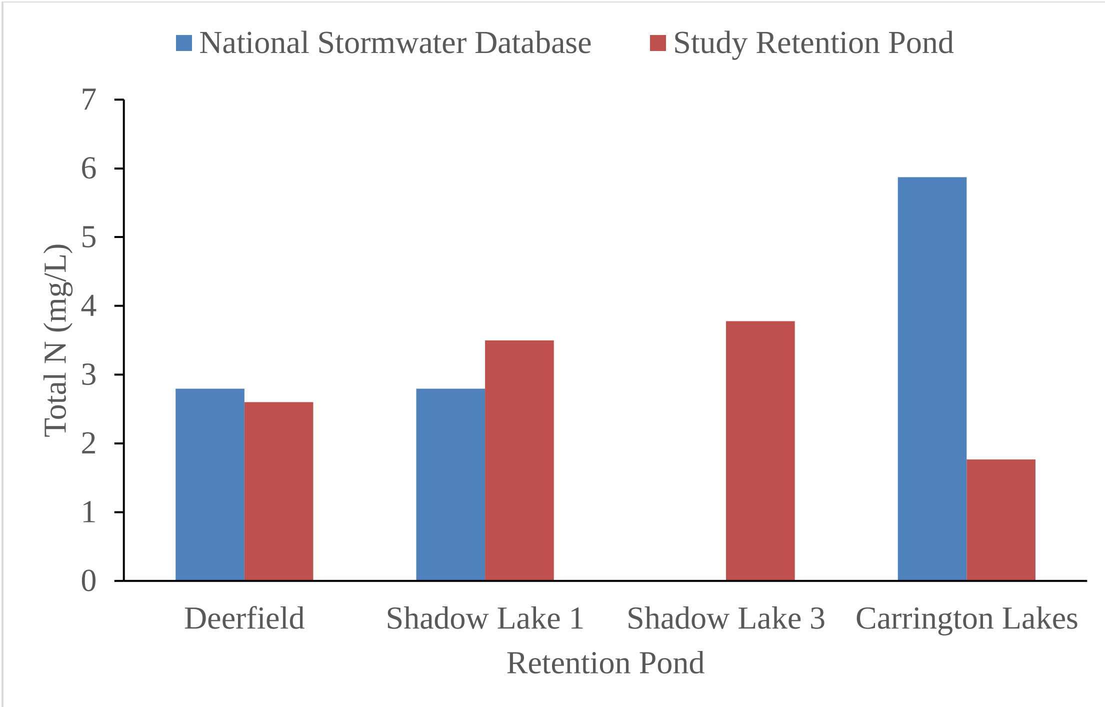

Figure 42: Mean TN concentration comparison between NSD and study retention ponds

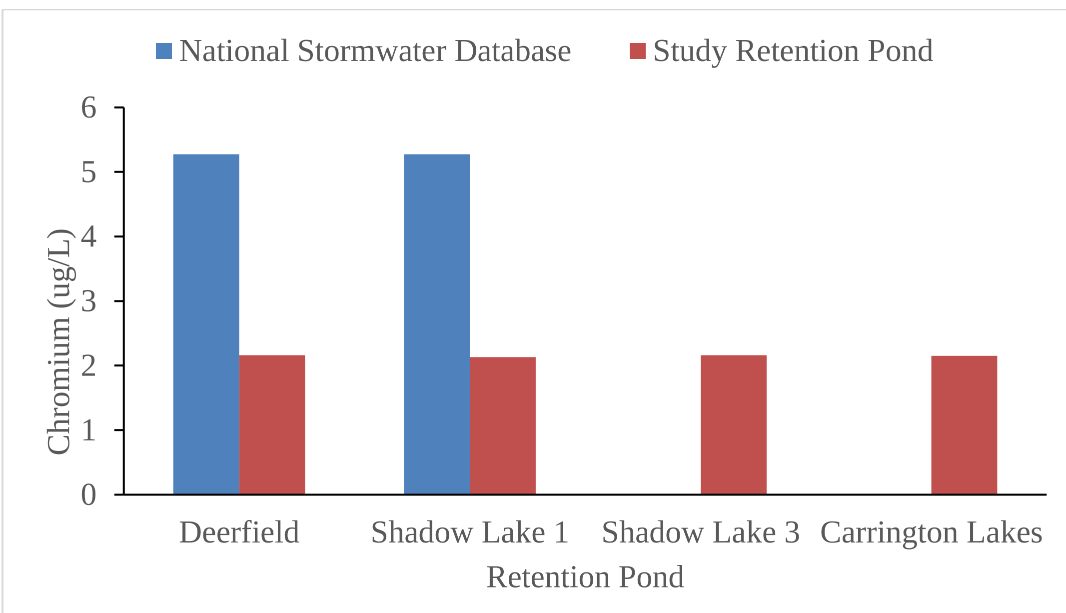

Figure 43: Mean Cr concentration comparison between NSD and study retention ponds

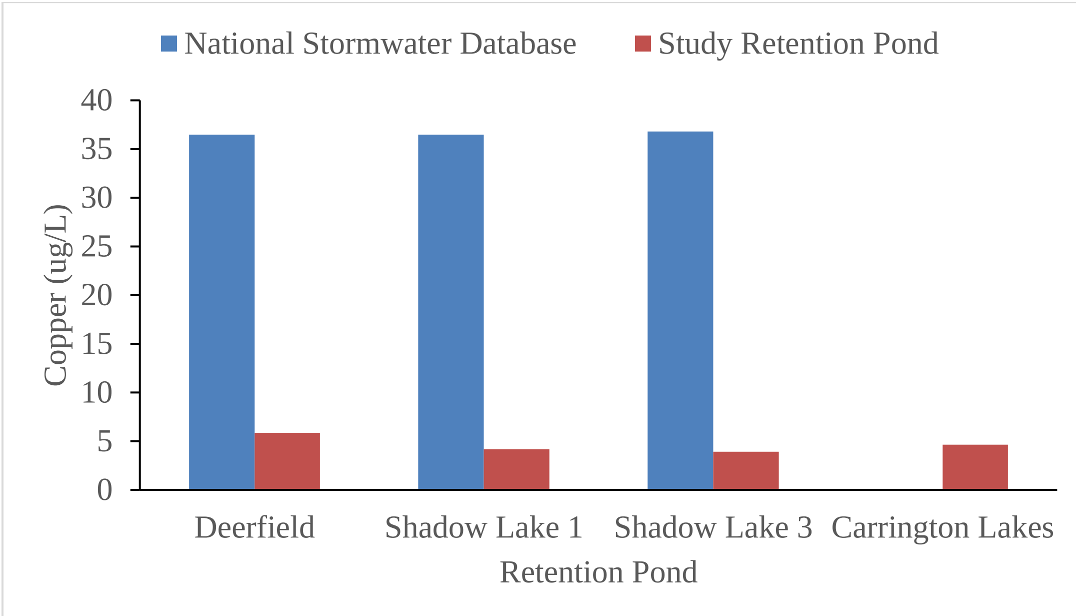

Figure 44: Mean Co concentration comparison between NSD and study retention ponds

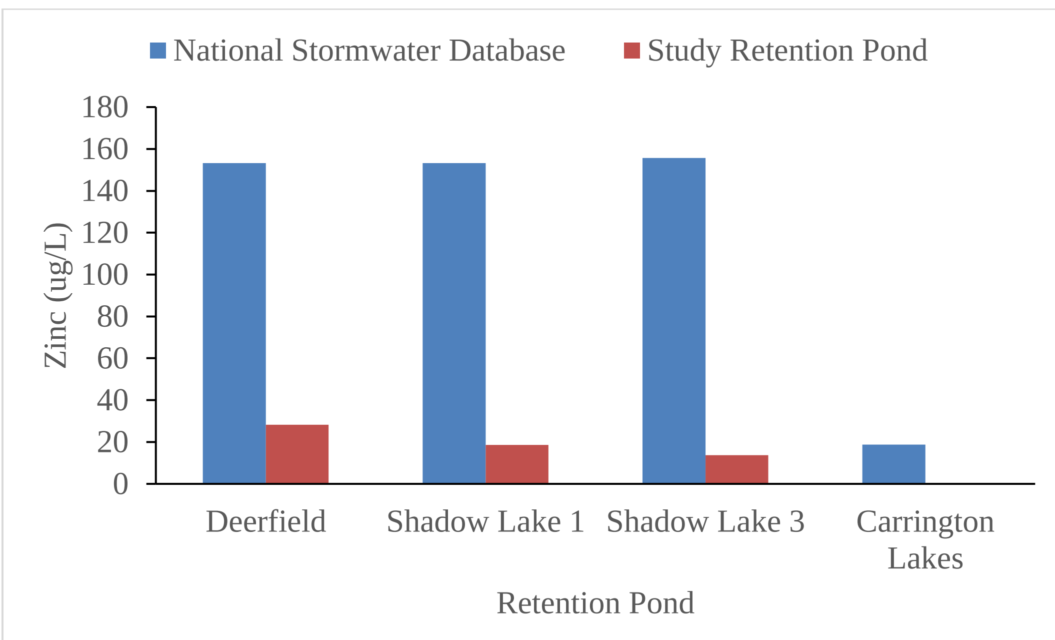

Figure 45: Mean Zn concentration comparison between NSD and study retention ponds

4 Conclusions

The collection analysis of stormwater runoff samples and water quantity data for a traditional development basin with an adjacent LID BMPs development basin provides an opportunity for a direct comparison of localized stormwater management effectiveness. Furthermore, while not directly downstream from the paired watersheds, collecting and analyzing samples from nearby retention ponds provides a similar comparison of effectiveness.

In these studies, data from a total of 42 storm events were evaluated during the 21-month study. Peak discharge and runoff duration were both significantly different for the LID BMPs development than the traditional stormwater development. This study has shown LID BMPs, even in extreme weather environments like Oklahoma, provide an effective means for stormwater control. The increased stormwater management control is achieved through increasing runoff staying in the localized area, reducing the magnitude of the runoff surge into local water bodies, and increasing the time runoff takes to leave the localized area. Improved stormwater management at the localized level reduces requirements and stress of stormwater management downstream.

The LID BMPs improved water quality as well, by showing significantly different ($p < 0.05$) TSS concentrations. The other nutrients did not show statistical significant differences but the average concentrations of traditional stormwater runoff were higher. The one exception was DRP with a lower concentration for the traditional development than the LID BMPs development, which had a significant difference, and an average increase of over 200%. The cause of increased phosphorous was most likely associated with the media used in the rain gardens or possibly anaerobic conditions in the permeable pavement.

Analyzed metals concentrations showed similar LID BMPs effectiveness. Of the 14 metals analyzed, all except two (Ca and Mg) had lower average concentrations in the LID watershed runoff than the traditional development runoff. The trace metals Cr, Cu, and Zn are some of the common metals harmful to aquatic ecosystems, prevalent in automobile pollution, and commonly removed by absorption or cation exchange. The statistical significance between the concentrations for Cr, Cu, and Zn combined with increased runoff concentrations of Mg and Ca suggests rain garden soil environmental conditions are promoting cation change.

The LID BMPs showed improved water quality performance under certain storm event characteristics. LID BMPs total runoff volume reductions and reduction percentages were both greater at higher rainfall intensities and for events with longer times since the previous storm event. LID BMPs decreased peak discharge rates to a larger degree for higher intensity and higher storm totals, with little to no effect based on the time since the previous storm event.

LID BMPs removed statistically significant amounts of TN and TSS while releasing statistically significant concentrations of DRP for rainfall totals above 1.18 inches. TSS was more effectively removed for storms with intensities below 1 in/hr, while more DRP was released in storms with intensities greater than 1 in/hr. The only statistically significant difference in time since previous event was for events greater than 6 days in TN and DRP concentrations.

The stormwater retention ponds were effective in providing water quality improvements. TSS, BOD, TP, Co, Cr, and Zn concentrations were substantially lower for all four retention ponds than the average concentrations for similar drainage areas in the NSD. The one water quality parameter for which the retention ponds had similar concentrations was TN. Carrington Lakes had much lower concentrations, but Deerfield had only slightly lower concentrations, while Shadow Lake 1 had higher concentrations than their respective similar sites. The NSD comparable sites for Shadow Lake 3 did not show TN data so it is unknown how its concentrations would compare. Besides comparing the retention pond concentrations to the NSD, the concentrations were also compared using ratios of the pond surface area to total drainage area, pond surface area to total imperviousness area, pond surface area to green space area, and pond surface area to designed storage volume of a storm event with a two-year reoccurrence interval. The correlation between physical parameters and concentrations varied between constituents and ponds. TSS was not correlated to any of the physical parameters while nitrogen compounds were correlated to pond surface to total drainage area, and phosphorous concentrations were correlated to pond surface area to imperviousness.

The combination of LID BMPs and retention ponds in series would provide the best chance of decreasing nutrient concentrations in the Lake Thunderbird watershed to the desired levels. While retention ponds are currently built for stormwater quantity management, the implementation of LID BMPs would reduce the required storage size of the retention ponds and therefore costs. Additionally, taking water quality impacts into account during retention pond design would further decrease nutrient loads into Lake Thunderbird.

5 Bibliography

Ahiablame, L., B. Engel, and I. Chaubey, 2012. “Effectiveness of Low Impact Development Practices: Literature Review and Suggestions for Future Research.” Water, Air & Soil Pollution 223(7): 4253-4273.

Ackerman, D. and E.D. Stein. 2008. Evaluating the effectiveness of best management practices using dynamic modeling. J. Environ. Eng. 134(8): 628 – 639.

Anderson D.M, Glibert P.M, Burkholder J.M., 2002 “Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences.” Estuaries. 25(4b): 562–584

Arnold, C and Gibbons, C., 1996 “Impervious Surface Coverage- The Emergence of a Key Environmental Indicator.” Journal of the American Planning Association 62(2): 243-258

Athanasiadis, Konstantinos and Horn, H & Helmreich, Brigitte., 2010. “A field study on the first flush effect of copper roof runoff.” Corrosion Science. 52: 21-29.

Barbosa, Ana and Fernandes, Joao & David, Luís., 2012. “Key issues for sustainable urban stormwater management.” Water Research. 46.

Barrett, M. E. 2004 “Retention Pond Performance: Examples from the International Stormwater BMP Database.” Critical Transitions in Water and Environmental Resources Management: Proceedings World Water and Environmental Resources Congress: Salt Lake City, Utah, June 27–July 1

Barret, M, 2008. “Comparison of BMP Performance Using the International BMP Database” Journal of Irrigation and Drainage Engineering 134 (5): 556-661.

Bean, E.Z., Hunt, W.F., Bidelspach, D.A. and Burak, R.J., 2004. “Study on the Infiltration Rate of Permeable Pavements,” in Proceedings of 1st Water and Environment Specialty Conference, Canadian Society for Civil Engineering, Saskatoon.

Bedan, E.S., and Clausen, J.C., 2009. “Storm water runoff quality and quantity from traditional and low impact development watersheds,” Journal of the American Water Resources Association, 45(4): 998-1008.

Braswell AS, et al. J Environ Manage., 2018. “Hydrologic and water quality performance of permeable pavement with internal water storage over a clay soil in Durham, North Carolina” Journal of Environmental Management. 224: 277-287

Brett M.T., Arhonditsis G.B., Mueller S.E., Hartley D.M., Frodge J.D. and Funke D.E., 2005 “Non-point-source impacts on stream nutrient concentrations along a forest to urban gradient.” Environmental Management. 35: 330–342.

Brezonik, P.L., and Stadelmann, T.H., 2002 “Analysis and Predictive Models of Stormwater Runoff Volumes, Loads and Pollutant Concentrations from Watersheds in the Twin Cities Metropolitan Area, Minnesota, USA.” Water Research. 36: 1743-1757.

Brown, R.A., and Borst, M., 2014. “Evaluation of surface infiltration testing procedures in permeable pavement systems.” Journal of Environmental Engineering. 140(3)

Brown, R.A., and Hunt, W.F., 2011. “Impacts of media depth on effluent water quality and hydrologic performance of undersized bioretention cells.” Journal of Irrigation Drainage Engineering. 137: 132 – 143

Burns M.J., Fletcher T.D., Walsh C.J., Ladson A.R., and Hatt, B.E., 2012. “Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform.” Landscape Urban Planning. 105: 230–240.

Coffman, L., 2000. “Low-impact development design strategies – an integrated design approach,” EPA 841-B-00-003, 150 pp.

Coffman, R., 2014. “Trailwoods neighborhood best management practices,” Prepared for Oklahoma Conservation Commission, 40 pp.

Chapman, C., and Horner., R.R., 2010. “Performance assessment of a street-drainage bioretention system.” Water Environmental Resources 82(2): 109 – 119.

Charters, F.J., Cochrane, T.A. and O'Sullivan, A.D. 2015. “Particle size distribution variance in untreated urban runoff and its implication on treatment selection.” Water Research. 85: 337-345.

Chen, J., and Adams, B.J., 2006 "Urban Stormwater Quality Control Analysis with Detention Ponds." Water Environment Research. 78(7): 744-53

Chen, X. L., Peltier, E., Sturm, B. S. M., and Young, C. B. 2013. “Nitrogen removal and nitrifying and denitrifying bacteria quantification in a stormwater bioretention system.” Water Research. 47: 1691–1700.

Cheng, M., Coffman, L., Zhang, Y., and Licsko, J., 2005. “Hydrologic responses from low impact development compared to conventional development,” Stormwater Management for Smart Growth. 337-357 pp.

Comings, K., Booth, D., and Horner, R., 2000. “Storm Water Pollutant Removal by Two Wet Ponds in Bellevue, Washington.” Journal of Environmental Engineering. 126:4(321).

Damodaram, C., Giacomoni, M.H., Khedum, C.P., Holmes, H., Ryan, A., Saour, W., Zechman, E.M., 2010. “Simulation of combined best management practices and low impact development for sustainable stormwater management.” JAWRA. 46(5): 907-918

Dams, J., Batelaan, O., Nossent, J., and Chormanski, J., 2008. “A strategy towards improved hydrological model parameterisation in urbanized catchments using remote sensing derived impervious surface cover maps.” 15-19.

Davis, A.P., M. Shokouhian, M., Sharma, H., Minami, C., and D. Winogradoff, D., 2003. “Water quality improvement through bioretention: lead, copper, and zinc removal.” Water Environmental Resources. 75: 73 – 82.

Davis, A.P., and McCuen, R.H., 2005. “Stormwater Management for Smart Growth.” Stormwater Management for Smart Growth. Pp 361-368.

Davis, A.P., Shokouhian, M., Sharma, H., and Minami, C., 2006. “Water quality improvement through bioretention media: nitrogen and phosphorus removal.” Water Environmental Resources. 78: 284 – 293.

Davis, A.P., Hunt, W.F., Traver, R.G., and M. Clark, M., 2009. “Bioretention technology: Overview of current practice and future needs.” Journal of Environmental Engineering. 135(3): 109 – 117.

DeBusk, K. M., and Wynn, K. M., 2011. “Storm-water bioretention for runoff quality and quantity mitigation.” Journal of Environmental Engineering. 137: 800–808.
Deletic A., 1990 “The first flush load of urban surface runoff.” Water Resources. 32(8): 2462–70.

Deletic, A., and Fletcher, T. D., 2006. “Performance of grass filters used for stormwater treatment - A field and modelling study. Journal of Hydrology. 317(3-4): 261 – 275.

Dietz, M.E. and Clausen, J.C., 2005. “A field evaluation of rain garden flow and pollutant treatment.” Water Air Soil Pollution. 167(1-4): 123 – 138.

Dietz, M., 2007. “Low Impact Development Practices: A Review of Current Research 28 and Recommendations for Future Directions.” Water, Air and Soil Pollution 186(1-4): 351- 363.

Djukic, A., Lekić, B., Rajaković-Ognjanović, V., Veljovic, D., Vulić, T., Đolić, M., Naunovic, Z., Despotovic, J., and Prodanovic, D., 2016. “Further insight into the mechanism of heavy metals partitioning in stormwater runoff.” Journal of Environmental Management. 168. 104-110.

Drake, J., Bradford, A., and Van Seters, T., 2014. “Water effluent quality from partial-infiltration permeable pavement systems.” Journal of Environmental Engineering 140.

Dreelin, E. A., Fowler L., and Carroll, R.C., 2006. “A test of porous pavement effectiveness on clay soils during natural storm events.” Water Resources. 40: 799 - 805.

Driscoll, E.D., 1983, "Performance of detention basins for control of urban runoff quality: Lexington, University of Kentucky," 1983 International symposium in Urban Hydrology, Hydraulics and Sediment Sontrol, 46 p.

Duncan, H. P., 1999. "Urban Stormwater Quality: A Statistical Overview. Melbourne, Australia," Cooperative Research Centre for Catchment Hydrology. Retrieved September 19, 2018, (https://www.researchgate.net/publication/228939940_Stormwater_Flow_and_Quality_and_the_Effectiveness_of_Non-proprietary_Stormwater_Treatment_Measures_A_Review_and_Gap_Analysis).

Environmental Protection Agency, 2000. "Low impact development (LID): a literature review," EPA 841-B-00-005. 41 pp.

Environmental Protection Agency, 2007. "Reducing storm water costs through low impact development (LID) strategies and practices," EPA 841-F-07-006. 37 pp.

Gilroy, K.L. and McCuen.R.H., 2009. "Spatio-temporal effects of low impact development practices." Journal of Hydrology. 367: 228 – 236.

Gnecco, I., Berretta, C., Lanza, L.G., and Barbera, P., 2005 "Quality of stormwater runoff from paved surfaces of two production sites." 10th Int. Conf. On Urban Drainage, Copenhagen, Denmark.

Goonetilleke, A., Thomas, E., Ginn, S., and D. Gilbert, D., 2005. "Understanding the role of land use in urban stormwater quality management." Journal of Environmental Management. 74: 31-42

Gowland, D. and Younos, T.," Feasibility of Rainwater Harvesting BMP for Stormwater Management." VWRRC Special Report SR38-2008

Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X. and Briggs, J.M., 2008. "Global change and the ecology of cities." Science, 319(5864): 756-760.

Grimmond, S.U.E., 2007. "Urbanization and global environmental change: local effects of urban warming." Geographical Journal. 173(1): 83-88.

Harper, H.H., Baker, D.M., 2007. Evaluation of current stormwater design criteria within the State of Florida. Environmental Research and Design, Inc., Orlando.

Hatt, B.E., Fletcher, T.D., and Deletic, A. 2009. "Pollutant removal performance of field scale stormwater biofiltration systems." Water Science and Technology 59: 1567-1576.

Hawley, R.J. and Bledsoe, B.P., 2013. "Channel enlargement in semi-arid suburbanizing watersheds: A southern California case study." Journal of Hydrology. 496: 17–30.

Hawley R.J., MacMannis K.R., and Wooten M.S., 2013. "Bed coarsening, riffle shortening, and channel enlargement in urbanizing watersheds, northern Kentucky, USA." Geomorphology. 201: 111–126

Herngren, L., Goonetilleke, A. and Ayoko, G. A., 2005. "Understanding heavy metal and suspended solids relationships in urban stormwater using simulated rainfall." Journal of Environmental Management. 76:149-158

Hirsch, R.M., Walker, J.F., Day, J.C., and Kallio, R., 1990 'The Influence of Man on Hydrological Systems'. In M.G. Wolman and H.C. Riggs (eds.) Surface Water Hydrology. Vol. 0-1. Geological Society of America, Boulder, CO. pp. 329–359.

Hood, M. J., Clausen, J. C. and Warner, G. S., 2007, "Comparison of Stormwater Lag Times for Low Impact and Traditional Residential Development." Journal of the American Water Resources Association. 43: 1036-1046

Horowitz A.J., 2008 "Determining annual suspended sediment and sediment-associated trace element and nutrient fluxes," Science Total Environment 400(1-3):315-43

Hunt, W. F., Jarrett, A.R., and Smith, J.T., 2006. "Evaluating Bioretention Hydrology and Nutrient Removal at Three Field Sites in North Carolina." Journal of Irrigation and Drainage Engineering.132(6): 600-608

Hunt, W.F., Smith, J.T., Jadlocki S.J., Hathaway J.M., Eubanks P.R., 2008. "Pollutant removal and peak flow mitigation by a bioretention cell in urban Charlotte, NC". Journal of Environmental Engineering. 134: 403-408

Jacobson, C.R., 2011." Identification and quantification of the hydrological impacts of imperviousness in urban catchments: a review." Journal of Environmental Management. 92(6):1438-48

Jennings, D.B. and Taylor Jarnagin, T.S., 2002 "Changes in anthropogenic impervious surfaces, precipitation and daily streamflow discharge: a historical perspective in a mid-Atlantic sub watershed" Landscape Ecology. 17: 471

Jennings, A., and Adeel, A., Hopkins, A., Litofsky, A., Wellstead, S., 2013. "Rain Barrel–Urban Garden Stormwater Management Performance." Journal of Environmental Engineering. 139: 757-765.

Jones, M., and Hunt, W., 2010. "Performance of rainwater harvesting systems in the southeastern United States." Resources, Conservation and Recycling. 54:623-629.

Kayhanian M., C. Stransky, C., Bay, S., Lau, L., and M. Stenstrom, M.K., 2008. Toxicity "Evaluation of Urban Highway Runoff with Respect to Storm Duration." Journal of Science of the Total Environment. 389(2-3): 386-406.

Kloss, C., and Calarusse, C., 2006. “Roo1ops to rivers: Green strategies for controlling stormwater and combined sewer overflows.” Natural Resources Defense Council

Lawler, J. J., White, D., Neilson, R.P., and Blaustein. A.R., 2006. “Predicting climate-induced range shifts: model differences and model reliability.” Global Change Biology. 12:1568– 1584.

Lawler, D.W., Foster, I.D.L., Petts, G., Harper, S., and Morrissey, I.P., 2006 “Suspended sediment dynamics for June storm events in the urbanized River Tame, UK” Sediment Dynamics and the Hydromorphology of Fluvial Systems (Proceedings of a symposium held in Dundee, UK, July 2006). IAHS Publ. 306, 2006.

Lee, J.H., Bang, K.W., Ketchum Jr., L.H., Choe, J.S., and Yu, M.J., 2002 “First flush analysis of urban storm runoff,” Science of The Total Environment. 293(1–3): 163-175

Leopold, B. L., 1968. “Hydrology for urban land planning – a guidebook on the hydrologic effects of urban land use,” U.S. Geological Survey Circular. 554: 18

Li L.Q., Yin, C.Q., He, Q.C., K, L.L., 2007 First flush of storm runoff pollution from an urban catchment in China. Journal of Environmental Sciences. 19(3):295–299.

Line, D. E., and White, N. M., 2015. “Runoff and pollutant export from a LID subdivision in North Carolina.” Journal of Environmental Engineering. 142(1).

Lucas, T., and Nichols, P., 2015 “The pollution removal and stormwater reduction performance of street-side bioretention basins after ten years in operation,” Science of The Total Environment. 536:784-792,

Mangangka, I. R., Liu, A., Egodawatta, P., and Goonetilleke, A., 2015. Performance characterisation of a stormwater treatment bioretention basin. Journal of Environmental Management. 150: 173–178

Martin, E.H., and Smoot, J.L., 1986, “Constituent-load changes in urban stormwater runoff routed through a detention pond-wetlands system in central Florida” U.S. Geological Survey Water-Resources Investigations Report 85-4310, 74 p.

Martin-Mikle, J.C., de Beurs, M.K., Julian, P.J., and Mayer, M.P., 2015. “Identifying priority sites for low impact development (LID) in a mixed-use watershed,” Journal of Landscape and Urban Planning. 140: 29-41.

McCarthy, D. T., Hathaway, J. M., Hunt, W. F., and Deletic, A. 2012. “Intra-event variability of Escherichia coli and total suspended solids in urban stormwater runoff.” Water Research. 46(20): 6661-6670

McPhillips, L. E., and Walter, M. T. 2015. “Hydrologic conditions drive denitrification and greenhouse gas emissions in stormwater detention basins.” Ecological Engineering 85: 67–75

Miguez, M., Bahiense, J.,Rezende, O., and Verol, A., 2012. “New urban developments: flood control and LID—a sustainable approach for urban drainage systems.” 469-480

Mullane, J., Flury, M., Iqbal, H., Freeze, P., Hinman, C., Cogger, C., and Shi, Z., 2015. “Intermittent rainstorms cause pulses of nitrogen, phosphorus, and copper in leachate from compost in bioretention systems.” Science of The Total Environment. 537: 294-303

Niemczynowicz, J., 1999. “Urban Hydrology and Water Management – Present and Future Challenges.” Urban Water. 1: 1-14

Nunn, J., 2014. “Low Impact Development practices: traditional vs. LID storm water management comparison,” Prepared for Orange County Board of County Commissioners. 271 pp.

Olivera, F and DeFee, B.B., 2007. “Urbanization and its effect on runoff in the white oak bayou watershed, Texas” Journal of American Water Resources. 41:1

Oklahoma Climatological Survey. 2015. “Climate of Oklahoma,” Retrieved August 2014– May 2015 from (http://climate.ok.gov/index.php/climate/category/oklahoma_climate).

Oklahoma Climatological Survey. 2018. “Climate of Oklahoma,” Retrieved July 2018 from (http://climate.ok.gov/index.php/climate/category/oklahoma_climate).

Oklahoma Conservation Commission, 2010. “Watershed based plan for the Lake Thunderbird watershed” EPA 319(h) Grant C9-996100-13, Project 2, Output 2.2.7. 34 pp.

Oklahoma Conservation Commission, 2015. “Lake Thunderbird watershed implementation project: Trailwoods demonstration site monitoring – quality assurance project plan,” FY 2008 §319(h) EPA Grant C9-99100-14, Project 10, Output 10.3.2a, Q-Trak 12-351. 33 pp. 119

Pappas E.A., Smith D.R., Huang C., Shuster W.D., Bonta J.V, 2008. “Impervious surface impacts to runoff and sediment discharge under laboratory rainfall simulation.” Catena. 72:146-152.

Park, D. and Roesner, L., 2013. “Effects of Surface Area and Inflow on the Performance of Stormwater Best Management Practices with Uncertainty Analysis.” Water Environment Research. 85: 782-792.

Pitt, Robert and Maestre, A and Hyche, H and Togawa, N., 2008. "The Updated Stormwater Quality Database (NSQD), Version 3." Proceedings of the Water Environment Federation. 1007-1026. 10.2175/193864708788735015. Version 4.02 - March 17 2015

Prince George's County, 1999 "Programs and Planning Division. Low Impact Development Design Strategies: An Integrated Approach." Environmental Protection Agency

Qin, H., Kang-mao, H., and Fu, Guangtao, F., 2016. "Modeling middle and final flush effects of urban runoff pollution in an urbanizing catchment." Journal of Hydrology. 534

Ryan, P.A 1991. "Environmental effects of sediment on New Zealand streams: A review, New Zealand" Journal of Marine and Freshwater Research. 25(2): 207-221,

Roesner, L.A., Bledsoe, B.P. and Brashear, R.W., 2001. "Are best-management-practice criteria really environmentally friendly?." Journal of Water Resources Planning and Management. 127(3):150-154

Roig, N., Sierra, J., Moreno-Garrido, I., Nieto, E., Gallego, E.P., Schuhmacher, M. and Blasco, J., 2016 "Metal bioavailability in freshwater sediment samples and their influence on ecological status of river basins," Science of The Total Environment. 540:287-296

Rushton, B., 2001. "Low-impact parking lot design reduces runoff and pollutant loads." Journal of Water Resources Planning Management. 127(3): 172 – 179

Saget, A., Chebbo, G., and Bertrand-Krajewski, J., 1996, "The first flush in sewer systems," Water Science and Technology. 33(9): 101-108

Sansalone, J. and Buchberger, S., 1997. "Characterization of solid and metal element distributions in urban highway stormwater." Water Science and Technology. 36: 155-160

Sansalone, J., Kuang, X., Ying, G. and Ranieri, V. 2012. "Filtration and Clogging of Permeable Pavement Loaded by Urban Drainage." Water Research. 46(20): 6763-6774

Selbig, W.R. and Bannerman R.T., 2008. "A Comparison of Runoff Quantity and Quality from Two Small Basins Undergoing Implementation of Conventional and Low-Impact-Development (LID) Strategies: Cross Plains, Wisconsin, Water Years 1999-2005." U.S. Geological Survey Scientific Investigations Report 2008-5008. 57pp.

Serrano, L., and DeLorenzo, M. E., 2008. "Water quality and restoration in a coastal subdivision stormwater pond." Journal of Environmental Management. 88(1), 43–52

Stanley, D.W., 1996. "Pollutant removal by a stormwater dry detention pond." Water Environmental Resources. 68:1076 – 1083

Tixier, G., Lafont, M., Grapentine, L., Rochfort, Q., and Marsalek, J., 2011. "Ecological risk assessment of urban stormwater ponds: Literature review and proposal of a new conceptual approach providing ecological quality goals and the associated bioassessment tools." Ecological Indicators. 11:1497-1506.

Thorpe, T. and Lloyd, B., 1999 "The macroinvertebrate fauna of St. Lucia elucidated by canonical correspondence analysis" Hydrobiologia. 400: 19

Tortorelli, R.L., Rea, A., and Asquith, W.H., 2005. "Depth-Duration Frequency Of Precipitation for Oklahoma" USGS/WRIR 99-4232
Stream – Corridor – Restoration (Federal Interagency Stream Restoration Working Group: www.nrcs.usda.gov/technical/stream_restoration/newgra.html). 2017

Valtanen, M., Sillanpää, N., and H. Setälä, 2015. "Key factors affecting urban runoff pollution under cold climatic conditions." Journal of Hydrology. 529(3): 1578–1589

Vietz, G., Walsh, C.J., and Fletcher, T. D., 2015. "Urban hydrogeomorphology and the urban stream syndrome: Treating the symptoms and causes of geomorphic change." Progress in Physical Geography. 40(3):480-492

Vitousek, P.M., and Farrington, H., 1997. "Nutrient limitation and soil development: experimental test of a biogeochemical theory," Biogeochemistry. 37:63-75

Vorreiter, L. and Hickey, C., 1994. "Incidence of the first flush phenomenon in catchments of the Sydney region." National Conference Publication - Institution of Engineers, Australia. 3:359-364

Walker Jr., W., 1987 "Phosphorus removal by urban runoff detention basins," Lake and Reservoir Management. 3(1): 314-326

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M. and Morgan, R.P., 2005. "The urban stream syndrome: current knowledge and the search for a cure." Journal of the North American Benthological Society. 24(3):706-723.

Walsh, C.J., Fletcher, T.D., and Burns, M.J., 2012. "Urban stormwater runoff: a new class of environmental flow problem." PLoS One. *7*(9):45814.

Wei, Z., Simin, L., and Fengbing, T., 2013. "Characterization of urban runoff pollution between dissolved and particulate phases." The Scientific World Journal. 964737.

White, M.D and Greer, K., 2006. "The effects of watershed urbanization on the stream hydrology and riparian vegetation of Los Peñasquitos Creek, California," Landscape and Urban Planning. 74(2):125-138

Whitney, J.W., Glancy, P., Buckingham, S., and Ehrenberg, A., 2015. “Effects of rapid urbanization on streamflow, erosion, and sedimentation in a desert stream in the American Southwest.” Anthropocene.

Williams, C., Frost, P., and Xenopoulos, M., 2013. “Beyond best management practices: Pelagic biogeochemical dynamics in urban stormwater ponds.” Ecological Applications. 23(6): 1384-1395.

Winston, R. J., Dorsey, J. D., and Hunt, W. F. 2016., “Quantifying volume reduction and peak flow mitigation for three bioretention cells in clay soils in northeast Ohio.” Science of the Total Environment. 553: 83-95

Witter, A., Minh, E. and Nguyen, H., 2016. "Determination of oxygen, nitrogen, and sulfur-containing polycyclic aromatic hydrocarbons (PAHs) in urban stream sediments,” Environmental Pollution, 209:186-196

Zimmerman, M.J., Waldron, M.C., Barbaro, J.R., and Sorenson, J.R., 2010. “Effects of Low-Impact-Development (LID) practices on streamflow, runoff quantity, and runoff quality in the Ipswich River basin, Massachusetts: A summary of field and modeling studies,” U.S. Geological Survey Circular. 1361:44

6 Appendix A: Storm Event Characteristics

Event Date	Five-Minute Max Intensity (in/hr)	Total Rainfall (in)	Antecedent Dry Period (days)
9/12/2013	0.60	0.34	26
9/17/2013	0.72	0.21	4
9/19/2013	1.32	0.41	1
10/14/2013	2.28	1.53	8
10/18/2013	0.24	0.12	2
10/20/2013	2.52	0.76	2
10/26/2013	0.36	0.22	4
10/30/2013	0.84	0.54	3
11/22/2013	0.12	0.23	6
12/12/2013	0.25	0.37	6
2/3/2014	0.48	0.19	23
3/4/2014	0.36	0.41	2
4/7/2014	0.24	0.49	9
4/13/2014	0.48	0.10	5
4/21/2014	0.48	0.18	6
4/27/2014	0.60	0.14	3
5/8/2014	0.24	0.13	8
5/23/2014	0.60	0.71	9
6/2/2014	0.24	0.12	5
6/8/2014	0.84	1.27	3
7/30/2014	0.72	1.44	12
8/9/2014	1.80	0.76	2
8/18/2014	1.80	0.51	8
9/6/2014	0.84	0.66	3
10/6/2014	1.56	0.26	2

10/10/2014	2.16	1.46	3
10/13/2014	1.32	1.01	1
10/22/2014	0.24	0.14	8
11/4/2014	0.72	1.32	10
2/16/2015	0.12	0.06	14
2/24/2015	0.12	0.05	6
3/1/2015	0.12	0.28	4
3/5/2015	0.12	0.07	2
3/9/2015	0.24	0.52	3
3/13/2015	0.12	0.39	2
3/18/2015	0.48	0.57	4
3/25/2015	2.76	0.47	5
4/13/2015	2.16	1.52	1
4/18/2015	1.08	0.22	3
4/22/2015	0.48	0.39	2
4/27/2015	0.48	1.43	3
5/11/2015	4.55	10.16	7

Appendix B: Monthly Rainfall Comparison

	Monthly Total Rainfall (Inches)	Rainfall >0.01 Days	Rainfall> 0.1 Days	Greatest 24 hour Rainfall (inches)	Historical Total Rainfall Averages (1981-2010)
Sept	2.39	7	4	1.39	3.94
Oct	3.84	10	9	0.81	3.95
Nov	2.52	4	10	1.82	2.47
Dec	0.25	7	0	0.08	2.19
Jan	0.1	3	0	0.05	1.59
Feb	0.26	5	1	0.16	1.97
March	2.05	6	5	1.1	3.2
April	1.01	10	4	0.3	3.4
May	0.96	9	3	0.29	5.18
June	4.58	9	7	1.47	5.01
July	3.76	6	5	1.39	2.74
Aug	1.34	5	2	0.76	3.06
Sept	0.96	4	3	0.62	3.94
Oct	2.98	9	6	1.43	3.95
Nov	3.52	6	2	2.12	2.47
Dec	0.97	12	3	0.34	2.19
Jan	1.64	6	4	0.69	1.59
Feb	0.17	5	0	0.05	1.97
March	2.42	12	7	0.51	3.2
April	4.1	14	7	1.52	3.4
May	23.39	19	15	4.67	5.18

Appendix C: Peak Discharge

Peak Discharge (CFS)				
Date	Control	Treatment	Reduction	Percent
9/12/2013	0.46	0.21	0.26	55.84
9/17/2013	0.38	0.26	0.13	32.86
9/19/2013	1.08	0.61	0.46	43.16
10/14/2013	4.04	1.68	2.36	58.46
10/18/2013	0.09	0.07	0.01	16.27
10/20/2013	4.70	2.63	2.07	43.98
10/26/2013	0.16	0.13	0.03	19.49
10/30/2013	1.53	0.89	0.64	41.63
11/22/2013	0.26	0.25	0.02	5.74
12/12/2013	0.25	0.37	-0.13	-51.52
2/3/2014	0.15	0.12	0.03	20.56
3/4/2014	0.31	0.40	-0.09	-27.46
4/7/2014	0.25	0.14	0.12	45.64
4/13/2014	0.15	0.08	0.07	43.70
4/21/2014	0.11	0.12	-0.01	-8.51
4/27/2014	0.22	0.23	-0.02	-7.17
5/8/2014	0.20	0.14	0.06	28.24
5/23/2014	0.74	0.67	0.07	9.58
6/2/2014	0.19	0.18	0.00	2.22
6/8/2014	0.78	0.50	0.28	36.22
7/30/2014	0.65	0.48	0.17	26.09
8/9/2014	10.38	6.48	3.91	37.63
8/18/2014	3.70	1.75	1.95	52.78
9/6/2014	0.69	0.53	0.17	23.93
10/2/2014	0.12	0.13	-0.01	-12.02
10/6/2014	1.56	0.99	0.58	36.83
10/10/2014	2.75	1.48	1.27	46.26

10/13/2014	1.24	0.74	0.50	40.62
10/22/2014	0.19	0.15	0.04	20.04
11/4/2014	0.58	0.41	0.17	29.43
2/16/2015	0.15	0.13	0.02	15.36
2/24/2015	0.06	0.07	0.00	-3.48
3/1/2015	0.12	0.13	-0.01	-9.17
3/5/2015	0.14	0.21	-0.06	-45.97
3/9/2015	0.15	0.10	0.05	31.43
3/13/2015	0.13	0.28	-0.15	-111.03
3/18/2015	0.43	0.32	0.10	24.20
3/25/2015	4.27	1.96	2.31	54.11
4/13/2015	2.25	1.04	1.22	53.98
4/18/2015	0.93	0.64	0.29	31.30
4/22/2015	0.20	0.14	0.06	28.24
4/27/2015	0.93	0.51	0.42	44.79
5/11/2015	16.89	8.20	8.69	51.43

Appendix D: Peak Discharge Sampled Events

Date	Peak Q (CFS)		Difference	Percent
	CW	TW	Peak CW - Peak TW	%
10/14/2013	4.04	1.68	2.362	58.46
11/22/2013	0.262	0.247	0.015	5.72
12/12/2013	0.079	0.118	-0.040	-50.25
4/27/2014	0.216	0.231	-0.015	-7.17
5/27/2014	0.247	0.205	0.042	17.02
6/6/2014	0.779	0.530	0.249	31.97
6/7/2014	0.779	0.497	0.282	36.22
6/19/2014	2.703	1.592	1.111	41.11
6/23/2014	0.688	0.434	0.253	36.81
7/9/2014	0.706	0.448	0.258	36.49
7/30/2014	0.649	0.480	0.169	26.08
9/6/2014	0.697	0.530	0.167	23.94
10/10/2014	2.751	1.479	1.273	46.26
10/12/2014	1.241	0.737	0.504	40.62
11/4/2014	0.583	0.412	0.172	29.43
3/25/2015	4.273	1.961	2.312	54.11
4/13/2015	2.252	1.036	1.216	53.98
Mean	1.35	0.74	0.608	45.02
Median	0.71	0.50	0.209	29.62
Std. Dev.	1.30	0.56	0.736	56.61
Maximum	4.27	1.96	2.312	54.11
Minimum	0.08	0.12	-0.040	-50.25
Sample size	19	19		
Std. Error	0.30	0.13		
T-Test	0.048			

Appendix E: Total Runoff Sampled Events

Date	Total Q (CF)				Intensity	Time since rainfall	Total rainfall	Rainfall duration
	Control	Treat.	Diff.	Percent	in/hr	Days	Inches	Minutes
10/14/2013	4476	3751	705	15.85	2.28	8	1.53	225
11/25/2013	2182	1949	233	10.68	0.12	17	0.18	255
12/12/2013	998	2174	-1176	-117.80	0.12	14	0.04	135
4/27/2014	382	321	60	15.83	0.12	3	0.06	5
5/27/2014	1598	1876	-278	-17.40	0.12	2	0.38	185
6/6/2014	1641	1984	-344	-20.97	0.48	3	0.50	205
6/7/2014	6795	6691	103	1.52	0.60	1	1.19	770
6/19/2014	2888	2634	254	8.78	0.24	6	0.33	195
6/23/2014	2303	2959	-657	-28.52	1.08	2	0.58	155
7/9/2014	2415	1972	443	18.35	0.84	15	0.68	235
9/6/2014	2651	3207	-556	-20.97	0.72	3	0.66	550
10/6/2014	1508	1398	110	7.29	1.56	5	0.52	175
10/12/2014	7097	5977	1120	15.78	1.80	2	1.01	905
11/4/2014	6670	6225	445	6.67	1.80	13	1.32	705
3/25/2015	3788	1940	1848	48.78	0.84	6	0.47	40
4/13/2015	9536	5522	4014	42.10	1.92	2	1.52	500
Mean	4994	4065	929	18.60	0.97	5.6	0.77	414
Median	3558	3161	397	11.15	0.91	6.4	0.69	328
Std. Dev.	2533	2404	129	5.09	0.72	5.21	0.47	262.80
Maximum	2546	1864	682	27				
Minimum	9536	6691	2844	30				
Sample size	382	321	60	16				
Stand error	637	466						
T-Test	0.206							

Appendix F: Total Suspend Solids

	TSS Concentration (mg/L)		
	Control	Treatment	Difference
10/14/2013	4.10	8.58	-4.48
11/25/2013	11.94	5.97	5.97
3/15/2014	26.12	52.61	-26.49
4/6/2014	60.82	82.46	-21.64
4/27/2014	84.67	71.00	13.67
5/27/2014	83.58	7.84	75.75
6/6/2014	48.08	42.97	5.11
6/12/2014	38.52	21.32	17.20
6/19/2014	18.52	26.30	-7.78
7/9/2014	84.98	38.66	46.32
9/6/2014	110.00	10.47	99.53
10/13/2014	18.12	36.52	-18.40
11/4/2014	32.85	8.80	24.05
11/22/2014	99.60	22.40	77.20
3/25/2015	698.80	565.60	133.20
4/13/2015	112.22	40.40	71.82
Mean	95.81	65.12	30.69
Median	54.45	31.41	23.04
Std. Dev.	159.60	131.10	28.50
Maximum	698.80	565.60	133.20
Minimum	4.10	5.97	-1.87
Sample size	16.00	16.00	-
Std. Error	39.90	32.77	7.12
T Test	0.011	-	-