THE IMPACT OF ON-SITE WASTEWATER TREATMENT SYSTEMS ON THE NITROGEN LOAD AND BASEFLOW IN STREAMS OF URBANIZING WATERSHEDS OF METROPOLITAN ATLANTA, GEORGIA

by

CHRISTOPHER WILLIAM OLIVER

(Under the Direction of Mark Risse)

ABSTRACT

The impact of on-site wastewater treatment systems (OWTSs) on the N load and baseflow in streams of urbanizing watersheds was investigated. Synoptic samples and baseflow measurements of streams affected by low (LDS) and high density (HDS) OWTSs were taken four times from 2011 to 2012. Results suggested an increase in baseflow in HDS watersheds which may off-set effects of development and maintain baseflow under drought conditions but also indicated a positive correlation between nitrate-N concentrations and OWTS density within the watershed. The Soil and Water Assessment Tool (SWAT) was calibrated to predict stream discharge in a gauged watershed of Gwinnett County, Georgia and used to quantify the influence of OWTSs on water quantity. Analysis showed a 5.9% increase in water yield due to the addition of OWTSs. Results provided data that may be used to inform users and watershed planners about the influence of OWTSs on water quality and quantity.

INDEX WORDS: On-site wastewater treatment systems, Nitrogen load, Stream water quality, Stream water quantity, SWAT, Watershed modeling.

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B.S., The University of Georgia, 2011

A Thesis Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

MASTER OF SCIENCE

ATHENS, GEORGIA

2013

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May 2013

ACKNOWLEDGEMENTS

I would like to thank my advisor, Dr. Mark Risse, for his support throughout my graduate studies in the College of Engineering at the University of Georgia. I would also like to thank Dr. David Radcliffe for his advisement throughout both my undergraduate and graduate studies at UGA and Dr. Mussie Habteselassie for his individual contribution and expertise as a committee member.

Thank you to the members of the United States Geological Survey, John Clarke, Gary Holloway, and Michael Hamrick, who helped with field work on this project. Thank you to my fellow graduate students Riaz Khan, Katie Shepherd, Hao Zhang, and Nahal Hoghooghi for being willing to help with field and lab work. Thanks also to John Rema, Rajith Mukundan, Ken Bradshaw, and Robert Mckinley for their individual contributions to various components of this project.

Thank you to a special group of friends that I have made while in graduate school at the University of Georgia. We spent many hours discussing anything from our research problems to everyday life issues. You made the long days at Driftmier worth while. I will always cherish the memories we have made, and I hope that there will be many more.

Finally, the utmost thanks to all of my family for their support in everything I do. To my loving parents, Jason and Candace Oliver. Thank you for teaching me to fear God and to work hard. To my wife and my best friend, Anna. Thank you for your unconditional love and encouragement. I am so grateful that the Lord decided to put you in my life. I am looking forward to making many memories with you.

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CHAPTER 1

INTRODUCTION

Water quality and quantity concerns have increased significantly in recent years as people continue to move into the Piedmont region of the Southeastern United States. The populations in this region, that includes Alabama, Georgia, South Carolina, and North Carolina, increased 7.5 to 18.5 percent from 2000 to 2010. Georgia, with an 18.3 percent increase, almost doubled that of the national average of 9.7 percent. Most of its growth occurred in the northern part of the state in Metropolitan Atlanta. This sixteen county district surrounding the state's capital experienced a 23.3 percent increase in population, and the growth is expected to continue in the future (US Census Bureau, 2010). The groundwater and surface water systems are well connected in the Piedmont, with each watershed acting as a unit due to the relatively impermeable underlying rock. Due to the limited availability of high yield wells, surface water withdrawals account for about 78 percent of the public water supply of Metropolitan Atlanta (Clarke and Peck, 1991; Fanning, 2001).

On-site wastewater treatment systems (OWTSs), also known as septic systems, are widely used for domestic wastewater treatment throughout the Southeast. It is estimated that more than 30 percent of the homes in Georgia are on OWTSs which is higher than the national average of 23 percent (U.S.EPA, 2002). The number of OWTSs in Metropolitan Atlanta is estimated to be 526,000 which is 26 percent of the total housing units in the district (MNGWPD, 2006). OWTSs were once considered a temporary solution to be replaced eventually by centralized wastewater collection and treatment systems, but it is now recognized that properly managed OWTSs offer several advantages over centralized wastewater treatment facilities. These include reduced construction and maintenance costs, elimination of sanitary sewer overflow and leaks, and avoidance of inter-basin water transfers (U.S.EPA, 2002). The number of OWTSs is expected to increase as populations increase in Metropolitan Atlanta because of the high costs of centralized systems to extend out to suburban populations. Therefore, as populations in Metropolitan Atlanta increase and the use of OWTSs increases, their impact on surface water quality and quantity must be investigated.

Traditional OWTSs can be potential sources of pollution for groundwater and surface water. Contaminants include nutrients such as nitrogen (N) and phosphorus (P), microbial contaminants, viruses, and hormones (DHR, 2007). Nitrogen is the primary nutrient and contaminant of concern for this study. High N concentrations can impact both human health and the environment. Elevated N concentrations in drinking water, typically in the form of nitrate-N, can be harmful to humans causing restriction of oxygen transport in the bloodstream. This can be potentially fatal for young infants or can cause problems during pregnancy as they lack the enzyme needed to correct the condition. Excess N can also cause over-stimulation of growth of aquatic plants and algae that can clog water intakes, block light to deeper waters, and use up dissolved oxygen as they decompose. This results in eutrophication that can produce fish kills and a decrease in animal and plant diversity within the watershed (USGS, 2012).

Many studies have shown groundwater in residential areas with high density OWTSs to have high nitrate-N concentrations that are up to 4 times the drinking water limit of 10 mg L^{-1} set by the U.S.EPA (Gold et al., 1990; Harman et al., 1996; Kaushal et al., 2006; Postma et al., 1992). Other studies have identified OWTSs as the dominant source of N pollution at the watershed scale in streams where the watershed is developed with neighborhoods dependent upon OWTSs (Burns et al., 2005; Hatt et al., 2004; Heisig, 2000; Kaushal et al., 2006; Reay, 2004). There have also been studies to confirm the origin of nitrate-N concentration to be from OWTSs using source tracking techniques that geochemically fingerprint the source (Aravena et al., 1993; Lu et al., 2008; McQuillan, 2004; Silva et al., 2002).

Water quality performance requirements for OWTSs are not clearly defined because of uncertainty about the processes involved in systems discharging to groundwater. Primary drinking water standards are typically addressed in code regulations only by requirements that the system be located a specified horizontal distance from a well and a specified vertical distance from the seasonally high water table (U.S.EPA, 2002). Georgia requirements state that the absorption field must be at least 100 feet from any drinking water wells and 150 feet from any perennial stream banks in a water supply watershed. There must also be a minimum of 24 inches between the bottom of the absorption field and any seasonal groundwater table, rock, or impervious soil layer (DHR, 2007). The minimum lot size for OWTSs is typically estimated in order to protect drinking water wells from exceeding the 10 mg L^{-1} drinking water standard for N. In surface waters that are sensitive to nutrient inputs, the threshold concentrations can be significantly lower than the drinking water standard. For example, in the draft EPA standards for N in Florida surface waters, the critical concentrations range from 0.51 to 1.87 mg L^{-1} depending on the use of the water resource (U.S.EPA, 2010). TMDLs (Total

Maximum Daily Loads) are also becoming important watershed management and planning strategies to minimize watershed contamination. A TMDL developed for Lake Allatoona, a large reservoir north of Atlanta, includes N limits and attributes part of the nutrient load to OWTSs in the watershed (GADNR, 2012). There is a need however, for a more accurate assessment of the N load to streams contributed from OWTSs.

While most studies investigate the impacts of OWTSs on water quality, their influence on groundwater recharge and baseflow in streams is also an important water management issue for urbanizing watersheds of Metropolitan Atlanta. Several studies have indicated that increased impervious surfaces and constructed channels due to urbanization decrease infiltration and baseflow and increase storm water runoff (Calhoun et al., 2003; Landers et al., 2007; Simmons and Reynolds, 1982). However, some studies have reported that rising groundwater levels from the combination of leaking water and waste water-supply mains and OWTSs drainage networks more than offset the effects of reduced infiltration and baseflow resulting from urbanization (Lerner, 2002; Yang et al., 1999). The specific effect of OWTSs on water quantity has been investigated by two studies, both of which found that baseflow in watersheds with high density OWTSs was significantly greater than in watersheds with low density OWTSs indicating that while suburban developments do accelerate the transport of storm water runoff into streams, OWTSs can change the expected effects of development on storm water runoff and groundwater recharge (Burns et al., 2005; Landers and Ankcorn, 2008).

A common assumption by some environmental officials in Georgia is that OWTSs can be considered consumptive use and therefore reduce the amount of water recharging surface waters in Georgia. The Metropolitan North Georgia Water Planning

District originally considered OWTSs to be 100% consumptive use (water is withdrawn and not returned to groundwater or streams) for planning purposes (MNGWPD, 2006). However, as a direct result of the study done by Landers and Ankcorn (2008), the guidelines were revised to say that the degree of consumptive use is not known, but assumed to be “more consumptive than centralized systems with surface water discharges” (MNGWPD, 2009). Therefore, there is a need to more clearly define the contribution of OWTSs to groundwater recharge and to the baseflow in the streams of Metropolitan Atlanta.

Watershed-scale models can be very useful tools for understanding and predicting the effect of OWTSs on water quality and water quantity. While there have been a number of OWTS-scale models developed to evaluate hydraulic performance and N fate and transport associated with OWTS (Beggs et al., 2011; Beggs et al., 2004; Bradshaw and Radcliffe, 2011; Hassan et al., 2008; Heatwole and McCray, 2007), there has been little modeling work done at the watershed scale due to the uncertainties introduced by the complex subsurface hydrology, spatial and temporal variation in soil and water, and lack of data regarding OWTSs (Jeong et al., 2011). However, some efforts have applied the Soil and Water Assessment Tool (SWAT) to simulate environmental effects of OWTSs at the watershed-scale (Jeong et al., 2011; Lemonds and McCray, 2003; Pradhan et al., 2005). SWAT is a watershed-scale model that was originally developed for the USDA Agricultural Research Service for the long term simulation of the impact of land management practices and land use changes on water, sediment, and agricultural chemical yields on downstream water bodies (Neitsch et al., 2011). The model is widely used in water quality modeling studies, TMDL analysis, and nonpoint-source pollution analysis (Jeong et al., 2011). In the earlier studies mentioned above (Lemonds and McCray, 2003; Pradhan et al., 2005), SWAT did not have a built-in module for simulating OWTS processes, but they could be added as a point source within the watershed. The deficiency has since been addressed by Siegrist et al. (2005), who proposed a new biozone algorithm to be incorporated in SWAT to simulate the fate and transport of domestic pollutants discharged from OWTSs. Jeong et al. (2011) tested the biozone algorithm and found that the model performed well in predicting both groundwater table levels and NO_3^- concentration in the groundwater. However, because of a wide variety of soils and geology, there is a need for further analysis of the complex water and solute transport processes associated with OWTSs at the watershed-scale.

The overall objective of this research was to determine the impact of on-site wastewater treatment systems on the nitrogen load and baseflow in streams of urbanizing watersheds of Metropolitan Atlanta, Georgia. Therefore, specific goals were as follows:

1. To examine the differences in the N load and baseflow as well as other water quality indicators in streams of watersheds impacted by high and low density OWTSs.
2. To use measured flow data from a gauged watershed in Metropolitan Atlanta, Georgia to calibrate the SWAT watershed-scale model for predicting stream discharge.
3. To use the calibrated SWAT model to predict stream discharge with and without the presence of OWTSs in order to determine their influence on water quantity.

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CHAPTER 2

LITERATURE REVIEW

CONVENTIONIONAL ON-SITE WASTEWATER TREATMENT SYSTEMS

Conventional on-site wastewater treatment systems (OWTSs) primarily consist of a septic tank, an absorption trench, and the surrounding soil treatment unit (Figure 2.1). The wastewater is discharged from the building directly into the tank where it is recommended to be retained for at least 24 hours (DHR, 2007). The primary purpose of the septic tank is to protect the soil absorption system from becoming clogged by solids suspended in the raw wastewater. It provides a place for settling and for anaerobic decomposition of solid materials. The average person uses approximately 265 L of water per day and septic tanks range from 3785 to 5678 L in size. The large size is to incorporate a margin of safety and to maintain the 24 hour retention time during peak water use (U.S.EPA, 2002). As the raw wastewater resides in the tank, the larger solids settle to the bottom while the greases, oils and other floating particles rise to the top. This forms a sludge blanket at the bottom of the tank, and a scum layer at the water surface (DHR, 2007). Total suspended solids (TSS) in raw wastewater typically range from 36 to 161 mg L^{-1} (U.S.EPA, 2002). The solids that settle are broken down by anaerobic bacteria, and their components then dissolve into the liquid phase in the septic tank. This reduces the volume of the sludge in the bottom of the tank by about 40 percent, however it is recommended to pump the tank once every three to five years to remove the accumulated solids. If this is not done, solids can re-suspend and wash out into the absorption fields where clogging can occur (DHR, 2007).

The clarified liquid from the septic tank flows to the soil absorption trenches for further treatment and disposal. This liquid contains partially degraded waste constituents, suspended solids, organic matter, ammonium, and phosphorus. It may also contain large numbers of pathogenic bacteria and viruses (DHR, 2007). The function of the absorption trenches in an OWTS is to disperse wastewater effluent over the soil treatment area and to provide additional water storage capacity during periods of peak flow. The absorption trenches act as an interface between the septic tank and the soil treatment unit where most of the treatment occurs through physical, chemical, and biological processes (U.S.EPA, 2002). The trenches are typically filled with gravel and crushed rock or an engineered material that is highly permeable for infiltration and percolation of the wastewater through the underlying soil. The wastewater effluent is distributed through a perforated pipe in the trenches by gravitational flow or by periodic dosing using a pump or a dosing siphon (Tchobanoglous and Burton, 1991). Absorption trenches are typically sized based on the soil infiltration rate and the number of bedrooms in the home that the system will serve (DHR, 2007).

Effluent discharged to the disposal field infiltrates into the soil primarily through the bottom and side walls of the trench into the vadose zone, or the unsaturated soil zone between the ground surface and the groundwater or bedrock. Flow in the vadose zone depends on the type of soil and bedrock conditions. Effluent moves over soil particle surfaces and in capillary pores in response to the force of gravity (Tchobanoglous and Burton, 1991). A biomat composed of living and dead microbes and particulate matter typically develops near the interface between the gravel and soil. The growth and formation of the biomat is important for long-term hydraulic function and enhanced purification of the septic tank effluent. Its development produces uniform infiltration and increased sorption of waste constituents to the surrounding soil and organic matter (McCray and Christopherson, 2008). Most nutrients or compounds in the effluent are partially or completely removed from the aqueous phase as it infiltrates through the vadose zone. The mechanisms for removal are different for each compound. These include sorption, aerobic biodegradation, anaerobic biodegradation, volatilization, and abiotic degradation such as hydrolysis. The soil treatment unit is the most dynamic component of an OWTS, and the removal of compounds depends on a variety of factors such as the hydraulic loading rate, the saturated and unsaturated hydraulic conductivity of the soil, the geochemical composition of the soil, the microbial population present in the soil vadose zone, and the chemical composition of the wastewater effluent (McCray et al., 2009).

NITROGEN IN AN ON-SITE WASTEWATER TREATMENT SYSTEM

In an OWTS, nitrogen (N) is present in a variety of inorganic and organic compounds. The N cycle therefore, is an important mechanism for the treatment of the wastewater effluent. Raw wastewater typically contains 3-25 mg L^{-1} organic-nitrogen, 7-40 mg L^{-1} ammonium-nitrogen (NH_4^+), and less than 1 mg L^{-1} nitrate-nitrogen (NO_3^-) (Tchobanoglous and Burton, 1991). In the septic tank, N is mostly present as organic-N and ammonium. The organic-N compounds in the wastewater that include proteins, amino acids, amides, and urea are converted to NH_4^+ in the septic tank under anaerobic conditions through the process of ammonification (DHR, 2007).

In the absorption field NH_4^+ can be adsorbed to negatively charged sites located on soil minerals and organic matter, taken up by plants, or converted to NO_3^- during autotrophic nitrification. Autotrophic nitrification is the oxidation of inorganic N compounds such as ammonium or ammonia. It is a two-stage process where NH_4^+ is first converted to nitrite (NO_2^-) by ammonia-oxidizing microbes (Equation 2.1). NO_2^- is an intermediate N species that is quickly converted into NO_3^- by nitrite-oxidizing microbes (Equation 2.2). Both of these processes occur under aerobic conditions (Klotz, 2011; McCray et al., 2009).

$$NH_4^+ + 1.5O_2 + \textit{ammonia oxidizer} \rightarrow NO_2^- + H_2O + 2H^+ \qquad \textbf{Eqn. 2.1}$$

$$NO_2^- + 0.5O_2 + \textit{nitrite oxidizer} \rightarrow NO_3^- \qquad \textbf{Eqn. 2.2}$$

In the soil, plants can then absorb NH_4^+ or NO_3^- and convert the N into plant protein. However, in the area of the absorption field only part of the NH_4^+ and NO_3^- is taken up by the plants. NO_3^- is considered quite mobile in soils and can be leached out into groundwater where drinking water wells and surface water supplies may be contaminated, or it can be denitrified by naturally occurring bacteria in soil (DHR, 2007). Denitrification is the reduction of oxidized nitrogenous compounds such as NO_2^- and NO_3^- to a gaseous phase. The process is a stepwise pathway where an oxygen atom is released in each stage as it is combined with carbon that is released from the breakdown of organics to form CO_2 (Equation 3). Under anaerobic soil conditions denitrifying bacteria oxidize organic carbon to form energy while using NO_3^- as the terminal election acceptor (McCray et al., 2009).

$$NO_3^- \rightarrow NO_2^- \rightarrow NO + N_2O \rightarrow N_2(g) \qquad \textbf{Eqn. 2.3}$$

The basis of N removal in an OWTS is sequential nitrification and denitrification processes. The end goal of an OWTS in terms of N removal is to decrease the amount of NO_3^- that is easily leached into groundwater and surface water by reducing it to N_2 gas and releasing it back into the atmosphere (Oakley et al., 2010). Several factors affect the N cycling and the type of N compounds present in the absorption field of an OWTS. These factors include the concentration and type of the initial compound, the number and type of microbes in the soil, the availability of organic carbon, pH, temperature, the surface charge of soil particles, and the soil water content (DHR, 2007).

TRACKING NITROGEN PROCESSES IN THE ENVIRONMENT

The fate and transport of the various constituents of N in the environment has been extensively studied. Several different methods have been used to track the transformation of N in the environment. One method used is the ratio of NO_3^- to a conservative tracer such as chloride (Cl^-). Cl^- is a non-reactive solute that is not subject to transformation by microbial activity, and it has been shown to increase linearly with increasing NO_3^- concentrations. It also serves as a good indicator parameter for OWTSs impacts because it is present in all sewage (McQuillan, 2004).The NO_3^-:Cl^- ratio has been used as evidence of denitrification in an OWTS drainfield and along groundwater flow paths to streams (Bradshaw and Radcliffe, 2011; Lowrance, 1992).

Another method to track the transformation of N in the environment and identify the source of contamination is the use of stable (non-radioactive) isotopes of nitrogen through isotopic fractionation. Isotopic fractionation is the relative enrichment or depletion of one isotope over another of the same element, and it is affected by a variety of physical and biological processes. The lighter isotope of nitrogen, ^{14}N, is preferred by biological organisms over the heavier isotope, ^{15}N, for respiration and assimilation because the chemical bonds of lighter isotopes are generally broken down easier than those of heavier isotopes. As a result, ^{14}N becomes concentrated in cell mass while ^{15}N becomes concentrated in the residual N sources. Nitrate in groundwater or surface water that has been denitrified by microbes or originates from human or animal waste is enriched with ^{15}N (McQuillan, 2004). Distinct isotopic compositions have been identified to characterize N of different origin so that ^{15}N can be measured to distinguish between human and animal waste, soil organic nitrate-N, and synthetic fertilizers (Aravena et al., 1993). Isotopes are expressed in units of per mil (parts per thousand {‰}) higher or lower than that of a standard. The Greek letter delta (δ) indicates the relative enrichment of ^{15}N to atmospheric N (Silva et al., 2002). Typical $\delta^{15}N$ values range from -2 ‰ to +4 ‰ for commercial fertilizers, +3 ‰ to +8 ‰ for soil organic nitrate-N, and from 10 ‰ to 25 ‰ for human and animal wastes (Aravena et al., 1993). McQuillan (2004) reported $\delta^{15}N$ values of groundwater nitrate originating from OWTSs that ranged from +7.6 ‰ to +12.1 ‰. The enrichment or depletion of ^{15}N has been used to identify OWTSs and or leaking sewer lines to be sources of N in contaminated groundwater (Aravena et al., 1993; Silva et al., 2002).

NITROGEN LOSSES FROM ON-SITE WASTEWATER TREATMENT SYSTEMS

While OWTSs have been identified as a source of N in groundwater and streams, there is a wide range of results regarding the quantity of N that is leached from an OWTS drainfield. Harman et al. (1996) found NO_3^- concentrations in a plume caused by OWTS effluent that ranged from 20-120 mg L^{-1} with the highest concentrations closest to the drainfield. Postma et al. (1992) reported similar NO_3^- concentrations ranging from 3 to

115.5 mg L^{-1} near the drainfield of an OWTS. Wilhelm et al. (1994) found groundwater NO_3^- concentrations up to 30 mg L^{-1} near the drainfield of an OWTS with decreasing concentrations down gradient of the system. Bradshaw and Radcliffe (2011) reported NO_3^- concentrations up to 20 mg L^{-1} below the drainfield of an OWTS. Both Gold et al. (1990) and Kaushal et al. (2006) found groundwater NO_3^- concentrations in excess of 10 mg L^{-1} in areas with high density OWTSs. Cogger and Carlile (1984) also reported decreasing NO_3^- concentrations with increasing distances from an OWTS, but found relatively lower groundwater NO_3^- concentrations that ranged from less than 0.5 to 4.6 mg L^{-1}. This wide range of results indicates that nitrogen removal processes in the drainfield of an OWTS are highly dynamic and are influenced by many factors that may or may not be specific to the geographic location.

The complexity of the N fate and transport processes at the OWTS-scale is increased further at the watershed-scale. The literature investigating the impact of OWTSs on stream water quality suggest that they have a significant influence on the N load of streams. Hatt et al. (2004) sampled 15 small streams in Australia and found that there was a strong, positive correlation between total N concentrations and the density of OWTSs. Kaushal et al. (2006) estimated that between 19 and 23 percent of the annual N export from developed tributaries in Colorado was derived from OWTSs. Reay (2004) conducted field studies in the Chesapeake Bay region and reported that OWTS loadings to shallow groundwater were significant and resulted in mean shoreline N concentrations that were approximately 50 to 100 times greater than adjacent surface waters. Heisig (2000) and Burns et al. (2005) both investigated the effects of suburban development on water quality at baseflow within the Croton River basin in New York. Burns et al. (2005)

found that nitrate-N concentrations at baseflow in high and medium density residential catchments were elevated relative to undeveloped catchments. Heisig (2000) measured baseflow water quality over several seasons in 33 first and second order streams and found a strong, positive correlation between OWTS density and nitrate-N concentration.

ON-SITE WASTEWATER TREATMENT SYSTEMS AND WATER QUANTITY

The literature investigating the impact of OWTSs on stream water quantity also suggest that they have a significant influence on baseflow in streams. An increase in impervious surfaces due to urban development within a watershed has been found to decrease baseflow and increase stormwater flow due to less infiltration and more runoff (Calhoun et al., 2003; Landers et al., 2007). However, several studies have shown that OWTSs can offset the effect of urbanization on baseflow through the discharge of effluent into groundwater and eventually streams. Simmons and Reynolds (1982) analyzed 22 years of stream flow records for two watersheds in New York and found a decrease in baseflow with an increase in imperviousness and a transition from OWTSs to centralized sanitary sewer systems. However, the authors found little decrease in baseflow in two nearby watersheds that were undergoing increased urbanization but were unsewered, which suggested an increase in flow due to OWTSs. In the study by Burns et al. (2005) in New York, the authors found that baseflow during dry periods was greatest in a high density residential catchment. It was concluded that the combined effects of natural landscape features such as wetlands or human alterations such as OWTSs can change the expected effects of development within a watershed. Yang et al. (1999) reported similar results using a solute-balance and water balance approach in Nottingham, UK. The authors found that the combined influence of OWTSs, leaking water mains and sewers, and infiltration ponds resulted in rising groundwater levels and increasing baseflow.

The influence of OWTSs on groundwater recharge and baseflow in Metropolitan Atlanta, Georgia was previously investigated by Landers and Ankcorn (2008) of the United States Geological Survey. In this study, 24 small watersheds (0.181 to 8.81 km^2) in Gwinnett County, Georgia were selected based on similar geologic setting, precipitation, climate, ease of access for baseflow measurements, and the availability of spatial datasets. Of the 24 watersheds, 12 were characterized with high density OWTSs (HDS) having greater than 77 OWTSs per km^2 (200 OWTSs per mi^2), and 12 were characterized with low density OWTSs (LDS) having less than 39 OWTSs per km^2 (100 OWTSs per mi^2). The authors collected one set of synoptic field measurements of the discharge, electrical conductivity, and temperature at baseflow during extreme drought conditions in October, 2007. They found that the mean baseflow yield of the HDS watersheds was 90 percent greater than that of the LDS watersheds and that the positive correlation between density of OWTSs and baseflow yield and electrical conductivity (a common wastewater indicator) was statistically significant. They concluded that a significant factor in explaining the increased baseflow was the density of OWTSs within the watersheds but that there were unexplained variations due to a limited dataset.

MODELING ON-SITE WASTEWATER TREATMENT SYSTEMS

Models can be very useful tools in understanding, quantifying, and predicting the effect of OWTSs on the N load and baseflow of streams. Modeling the fate and transport of N associated with OWTSs at the watershed-scale has not been extensively researched due to the various uncertainties introduced beyond the drainfield of a single OWTS.

However, a new biozone algorithm proposed by Siegrist et al. (2005) was adapted to the Soil and Water Assessment Tool (SWAT 2009) in order to simulate the influence of OWTSs on water quality within a watershed (Jeong et al., 2011).

SWAT is a physically based watershed-scale model developed to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time (Neitsch et al., 2011). The model has been widely used in water quality modeling studies, the analysis of total maximum daily loads (TMDL), and nonpoint-source pollution analysis (Jeong et al., 2011). Basic user inputs for the model include a digital elevation model (DEM), land cover, and soils data. The Soil and Water Assessment Tool Users Manual provides a detailed description of the governing equations that the model uses for simulating processes within a watershed (Neitsch et al., 2011). A description of the main equations SWAT uses for simulating flow, sediment, and nutrients are described below.

In SWAT, a watershed is broken up into a number of spatially-linked subbasins that have similar climate and topography in order to describe the spatial heterogeneity and connectivity (Jeong et al., 2011) The watershed is further divided into hydrologic response units (HRUs). HRUs are lumped land areas within each subbasin that are comprised of unique land cover, soil, slope and management combinations but are not necessarily linked spatially (Neitsch et al., 2011). Watershed processes related to soil water, surface runoff, and sediment yield are computed at the HRU level and then aggregated for subsequent routing through the channel network (Jeong et al., 2011).

The hydrologic cycle of a watershed is broken up into a land phase and a routing phase. The land phase involves processes involved with the loadings of water, sediment and nutrients to the main channel in each subbasin. The routing phase describes processes involved with the movement of water, sediment, and nutrients through the channel network of the watershed (Neitsch et al., 2011).

The driving force behind the movement of sediments and nutrients in a watershed is the water balance equation:

$$SW_t = SW_0 + \sum_{i=1}^{t} R_{day} - Q_{surf} - E_a - w_{seep} - Q_{gw}$$ **Eqn. 2.4**

where SW_t is the final soil water content (mmH_2O). SW_0 is the initial soil water content on day *i* (mm H_2O). *t* is the time (days). R_{day} is the amount of precipitation on day *i* (mm H_2O). Q_{surf} is the amount of surface runoff on day *i* (mm H_2O). E_a is the amount of evapotransporation on day *i* (mm H_2O). w_{seep} is the amount of water entering the vadose zone from the soil profile on day *i* (mm H_2O). and Q_{gw} is the amount of return flow on day *i* (mm H_2O). The potential pathways of water movement simulated by SWAT at each HRU are canopy storage, infiltration, redistribution, evapotranspiration, lateral subsurface flow, surface runoff, ponds, tributary channels, and return flow (Neitsch et al., 2011). A detailed discussion of the equations used to simulate these processes can be found on pages 98-178 of the SWAT Users Manual.

Erosion and sediment yield are estimated for each HRU with the Modified Universal Soil Loss Equation (MUSLE):

$$sed = 11.8 \cdot \left(Q_{surf} \cdot q_{peak} \cdot area_{HRU}\right)^{0.56} \cdot K_{USLE} \cdot C_{USLE} \cdot P_{USLE} \cdot LS_{USLE} \cdot CFRG$$ **Eqn. 2.5**

where *sed* is the sediment yield on a given day (metric tons). Q_{surf} is the surface runoff volume (mm H_2O/ha).q_{peak} is the peak runoff rate (m^3/s). $area_{HRU}$ is the area of the HRU (ha). K_{USLE} is the USLE soil erodibility factor (m^3-metric ton cm). C_{USLE} is the USLE cover and management factor. P_{USLE} is the USLE support practice factor. LS_{USLE} is the USLE topographic factor and *CFRG* is the coarse fragment factor (Neitsch et al., 2011). The USLE factors are discussed in detail on pages 253-260 of the SWAT Users Manual.

SWAT tracks the fate and transport of nutrients such as N and P in the HRUs through the use of various pools of the specific nutrient cycle that include organic and inorganic forms. For example, N is tracked through five different pools. Two pools are inorganic forms of nitrogen: NH_4^+ and NO_3^-. The other three pools are organic forms of nitrogen: fresh organic N (crop residues and microbial biomass), active organic N, and stable organic N. Processes such as mineralization, decomposition, immobilization, nitrification, denitrification, fixation, and leaching are all simulated within the various pools (Neitsch et al., 2011). Detailed equations for the N cycle are described in detail on pages 187-200 of the SWAT Users Manual.

The biozone algorithm, developed to simulate the effects of OWTSs within a watershed, is conceptually drawn from the biozone layer (Figure 2.1), which is a biologically active layer in the soil absorption system directly below the infiltrative surface where there is growth of microorganisms feeding on the organic matter of the septic tank effluent. The biozone layer is assumed as a control volume that receives effluent from the OWTS and infiltration from the soil layer above while allowing percolation to soil layers below (Jeong et al., 2011).The mass balance equation of microorganisms in the control volume is estimated in SWAT by:

$$\frac{d(Bio)}{dt} = \alpha \cdot \left[\sum Q_{STE} \cdot C_{BOD,in} - I_p \cdot C_{BOD}\right] - R_{resp} - R_{mort} - R_{slough} \qquad \textbf{Eqn. 2.6}$$

where *Bio* is the amount of live bacteria biomass in the biozone (kg/ha), $C_{BOD, in}$ is the BOD concentration in the septic tank effluent (mg L^{-1}), C_{BOD} is the BOD concentration in the biozone, α is the ratio of live bacteria growth to BOD in the septic tank effluent, Q_{STE} is the flow rate of the septic tank effluent (mg L^{-1}), I_p is the amount of percolation out of the biozone (m^3/day), R_{resp} is the amount of respiration of bacteria (kg/ha), R_{mort} is the amount of mortality of bacteria (kg/ha), and R_{slough} is the amount of sloughed off bacteria (kg/ha; (Neitsch et al., 2011). The amount of biomass in the biozone affects the hydraulic properties of a soil including the field capacity, porosity, hydraulic conductivity, soil moisture and percolation. The equations used to simulate these processes within the biozone are described in detail on pages 397-399 of the SWAT Users Manual.

The transformation and removal of pollutants such as N in the biozone is directly related to the population of live bacteria biomass and biological processes within the biozone layer. The fate of N is estimated in SWAT by a first order reaction equation:

$$C_{end} = C_i \cdot e^{-K \cdot \Delta t} \qquad \textbf{Eqn. 2.7}$$

where C_{end} is the concentration of N in the biozone at the end of the day (mg L^{-1}), C_i is the concentration of N in the biozone at the beginning of the day (mg L^{-1}), and *K* is a first order reaction rate (day^{-1}), which is a function of the total biomass of live bacteria and a reaction rate coefficient (Neitsch et al., 2011).

The effluent from the septic tank passes through the biozone layer to the soil layers below where the constituents are then subject to the normal fate and transport processes that are expected to occur in the natural environment. The OWTSs within the watershed are aggregated to HRUs with similar soil type, average drainage area, average number of people in the house, and type of system (Jeong et al., 2011). The biozone module simulates each septic HRU based on whether or not the system is active or failing. A failing system is a system subject to hydraulic failure, and it occurs due to clogging of the biozone by suspended solids and plaque of biomass. The system fails when the soil porosity is reduced to the value for field capacity due to biozone clogging and it will remain as a failing system for a user specified number of days. During this time, the septic tank effluent migrates to the upper soil layers and eventually causes ponding on the soil surface. The nutrients in a failing system are transported with the septic tank effluent and the concentrations are estimated based on the initial concentrations and the amount of water that migrates to the surface. There are no biozone processes implemented while a system is failing. After the number of days of system failure has exceeded the designated time (simulating repair of the system), the system is reinitialized to an active system where the processes are simulated using the previously described biozone equations (Neitsch et al., 2011). A detailed discussion of the equations and processes simulated by the biozone algorithm as well as the implementation of the module into SWAT can be found on pages 394-405 of the SWAT Users Manual.

Jeong et al. (2011) evaluated the performance of the SWAT biozone algorithm in a watershed located in the Coastal Plain region of North Carolina. The model was shown to perform well in predicting both groundwater table levels (R^2 = 0.82) and NO_3^- concentration in the groundwater (R^2 = 0.76) measured in the drainfield of three homes. The model estimated that N in the domestic wastewater accounted for about 85 percent of the total N input into the soil system at each OWTS site, but at the watershed level OWTSs contributed to only 25% of the N inflow. It was also estimated that 5.2 kg N ha^{-1}

year^{-1} was lost through denitrification, and 24.7 kg N ha^{-1} year^{-1} was removed by plant uptake, which combined was 80% of the removal of total N input. The analysis suggested that while biological removal plays a significant role in N reduction through denitrification, N loading is still much greater than that of natural conditions. The authors concluded that the SWAT biozone algorithm produced reliable groundwater simulations of the N concentration in OWTSs as evidenced by the calibration and validation tests of the study.

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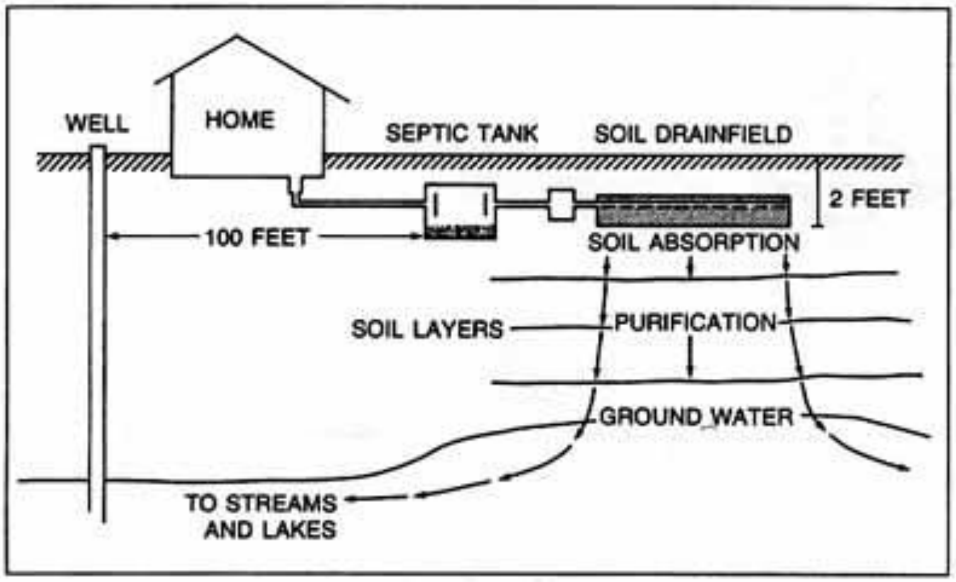

Figure 2.1. Diagram of a conventional on-site wastewater treatment system. Domestic wastewater enters from the house, undergoes physical, chemical and biological processes, and then is released to the drainfield for further purification (Jones and Yahner, 2008).

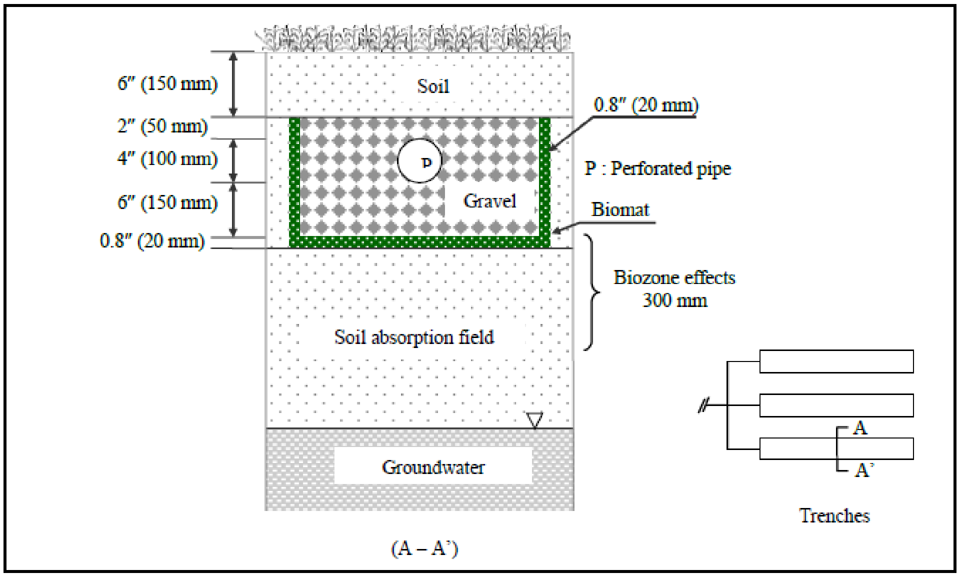

Figure 2.2. Configuration of a septic tank effluent distribution chamber and absorption system showing the formation of the biozone layer (Neitsch et al., 2011).

CHAPTER 3

THE IMPACT OF ON-SITE WASTEWATER TREATMENT SYSTEMS ON THE NITROGEN LOAD AND BASEFLOW IN STREAMS OF WATERSHEDS IN METROPOLITAN ATLANTA, GEORIGA[1]

[1] C.W. Oliver, L.M. Risse, D.E. Radcliffe, M. Habteselassie, J. Clarke. To be submitted to *Transactions of the ASABE.*

ABSTRACT

On-site wastewater treatment systems (OWTSs) are widely used in the Southeastern United States for domestic wastewater treatment. As suburban populations increase, the use of OWTSs is expected to further increase. OWTSs are often considered consumptive water use and can be potential sources of N pollution for groundwater and streams. This region heavily depends on surface waters for its water supply, therefore the impact of OWTSs on surface water quality and quantity must be determined. The overall objective of this research was to determine the impact of OWTSs on the N load and baseflow in streams of urbanizing watersheds in Metropolitan Atlanta, Georgia. This paper presents results of the differences in the N load and baseflow as well as other water quality indicators such as electrical conductivity (EC) and chloride (Cl^-) in streams of watersheds impacted by high (HDS) and low density OWTSs (LDS). Synoptic samples and discharge measurements of 24 watersheds were taken under baseflow conditions in November 2011, March 2012, July 2012, and November 2012. Mean baseflow measurements in November 2011, March 2012, and November 2012 were not statistically different between watersheds and showed no relationship with OWTS density within the watershed, but July 2012 measurements were significantly higher in the HDS watersheds and increased linearly with increasing OWTS density. EC and Cl^- concentrations increased linearly with increasing OWTS density within the watershed, and NO_3^- concentrations showed a linear increase with OWTS density above a threshold of about 100 OWTSs per sq.km. Results suggest an increase in baseflow due to the presence of OWTS effluent which may off-set the effects of impervious surfaces and maintain baseflow during drought conditions. Results also indicate a positive correlation

between NO_3^- concentration and OWTS density within the watershed above a density of about 100 OWTSs per sq.km. This study showed that OWTSs have positive and negative impacts on the water quality and quantity of urbanizing watersheds of this region. It provided data that may be used to inform users as well as watershed planners about the influence of OWTSs on the N load and baseflow in streams based on the density of OWTSs within the watershed.

INTRODUCTION

On-site wastewater treatment systems (OWTSs), also known as septic systems, are widely used for domestic wastewater treatment throughout the Southeast. It is estimated that 37 percent of the homes in Georgia are on OWTSs which is higher than the national average of 24 percent (U.S.EPA, 2002). The number of OWTSs in Metropolitan Atlanta is estimated to be 526,000 which is 26 percent of the total housing units in the district (MNGWPD, 2006). OWTSs were once considered a temporary solution to be replaced eventually by centralized wastewater collection and treatment systems, but it is now recognized that properly managed OWTSs offer several advantages over centralized wastewater treatment facilities. These include reduced construction and maintenance costs, elimination of sanitary sewer overflow and leaks, and avoidance of inter-basin water transfers (U.S.EPA, 2002) The number of OWTSs is expected to increase as populations increase in Metropolitan Atlanta because of the high costs of extending centralized systems to suburban populations. Total maximum daily loads (TMDL's) are also a driving factor in increasing the use of OWTS. Permitted surface water discharges are becoming more limited in order to meet TMDL requirements resulting in developments that are forced to use OWTSs for domestic wastewater treatment. Due to the limited availability of high yield wells, surface water withdrawals account for about 78 percent of the public water supply of Metropolitan Atlanta (Clarke and Peck, 1991; Fanning, 2001). Therefore, as the use of OWTSs in Metropolitan Atlanta increases, their impact on surface water quality and quantity must be determined.

Many studies have shown groundwater in residential areas with high density OWTSs to have high nitrate-N concentrations that are up to four times the drinking water limit of 10 mg L^{-1} set by the U.S.EPA (Gold et al., 1990; Harman et al., 1996; Kaushal et al., 2006; Postma et al., 1992; Wilhelm et al., 1994). Other studies have identified OWTSs as the dominant source of N pollution at the watershed scale in streams where the watershed contains neighborhoods dependent upon OWTSs (Burns et al., 2005; Hatt et al., 2004; Heisig, 2000; Kaushal et al., 2006; Reay, 2004). There have also been studies to confirm the origin of nitrate-N in groundwater and streams to be from OWTSs using source tracking techniques that geochemically fingerprint the source (Aravena et al., 1993; Lu et al., 2008; McQuillan, 2004; Silva et al., 2002). However, there is a need for a more accurate assessment of the N load to streams contributed from OWTSs in Metropolitan Atlanta in order to minimize watershed contamination through non-point source pollution.

While most studies investigate the impacts of OWTSs on water quality, their influence on groundwater recharge and baseflow in streams is also an important water management issue for urbanizing watersheds of Metropolitan Atlanta. Several studies have indicated that increased impervious surfaces and constructed channels due to urbanization decrease infiltration and baseflow and increase storm water runoff (Calhoun et al., 2003; Landers et al., 2007). However, some studies have reported that rising groundwater levels from the combination of leaking water and waste water-supply mains and OWTSs drainage networks can more than offset the effects of reduced infiltration and baseflow resulting from urbanization (Burns et al., 2005; Landers and Ankcorn, 2008; Simmons and Reynolds, 1982; Yang et al., 1999).

The influence of OWTSs on groundwater recharge and baseflow in Metropolitan Atlanta, Georgia was previously investigated by Landers and Ankcorn (2008) of the

United States Geological Survey. In this study, 24 small watersheds (0.181 to 8.81 km^2) in Gwinnett County, Georgia were selected based on similar geologic setting, precipitation, climate, ease of access for baseflow measurements, and the availability of spatial datasets. Of the 24 watersheds, twelve were characterized with high density OWTSs (HDS) having greater than 77 OWTSs per km^2 (200 OWTSs per mi^2), and twelve were characterized with low density OWTSs (LDS) having less than 39 OWTSs per km^2 (100 OWTSs per mi^2). The authors collected one set of synoptic field measurements of the discharge, electrical conductivity, and temperature at baseflow during extreme drought conditions in October, 2007. They found that the mean baseflow yield of the HDS watersheds was 90 percent greater than that of the LDS watersheds and that the positive correlation between density of OWTSs and baseflow yield and electrical conductivity (a common wastewater indicator) was statistically significant. They concluded that a significant factor in explaining the increased baseflow was the density of OWTSs within the watersheds but that there were unexplained variations due to a limited dataset.

The overall objective of this research was to determine the impact of OWTSs on the N load and baseflow in streams of urbanizing watersheds of the Ocmulgee and Oconee River basins in Metropolitan Atlanta, Georgia. The specific goal of this study was to conduct a more comprehensive water quantity and quality analysis of the watersheds examined by Landers and Ankcorn (2008) in order to understand how OWTS density within the watershed affects the baseflow and water quality throughout different seasons of the year. This paper presents results of the differences in the baseflow and N load as well as other water quality indicators in streams of watersheds impacted by LDS and HDS. The hypothesis was that watersheds impacted by HDS would have a higher baseflow and a higher N load when compared to watersheds with LDS. Baseflow and NO_3^- concentrations were expected to increase with increasing OWTS density within the watershed. The results from this study will provide data that will inform OWTS users, the OWTS industry, as well as local and state planners of the impacts of OWTSs and their contribution to the N load and baseflow of streams in the region based on the OWTS density within the watershed.

MATERIALS AND METHODS

The study area has been described in detail in Landers and Ankcorn (2008). The area is in the Southern Piedmont region of southeast Atlanta, Georgia and has a mean annual precipitation of about 1270 mm (National Weather Service, 2008). The small watersheds selected for the site are in the Ocmulgee and Oconee River basins, which drain to the Altamaha River and the Atlantic Ocean. The watersheds range in area from 0.181 to 8.81 km^2 with an average area of 2.49 km^2. Of the 24 watersheds selected, twelve are characterized as HDS with the remaining twelve characterized as LDS. Watersheds with less than 39 OWTSs per km^2 (200 OWTSs per mi^2) were considered LDS watersheds while watersheds with greater than 77 OWTSs per km^2 (100 OWTSs per mi^2) were considered HDS watersheds (Table 3.1). Other watershed selection criteria used were geological setting, precipitation, climate, accurate baseflow measurement locations and available spatial datasets of natural, infrastructure, and water use characteristics. The study area and watershed boundaries are shown in Figure 3.1.

Synoptic measurements of baseflow were taken concurrently with water sampling three times per year to capture the seasonal flow variations. Stream discharge measurements for the 24 sites were conducted by members of the United States Geological Survey (USGS) using the current-meter method as described by Rantz (1982). All measurements were taken under baseflow conditions within a 24-hour period with no intervening rainfall. USGS real-time stream gauges in the area were monitored to determine baseflow conditions. In addition to baseflow, basic water quality parameters such as temperature, pH, electrical conductivity (EC), and dissolved oxygen were measured using a Quanta water quality meter. Measurements and sampling events occurred in November 2011, March 2012, July, 2012, and November 2012.

Water samples were collected at the same time as the synoptic stream flow measurements and stored on ice in the field before analysis. All collected stream samples were analyzed by the University of Georgia Environmental Services Laboratory for NH_4^+, NO_3^-, total Kjeldahl N (TKN), and Cl^-. Samples were analyzed for NH_4^+ following distillation-titration according to the method of Bremner (1965). NO_3^- and Cl^- concentrations were determined using EPA Method 300.1: Determination of Inorganic Anions in Drinking Water by Ion Chromatography (U.S.EPA, 2007). TKN concentrations (the sum of the organic-N, NH_3, and NH_4^+) were determined using a modified version of the micro-Kjeldahl methods (AOAC, 1996; APHA-AWWA-WEF, 1998). All stream samples were also analyzed by the University of Georgia Analytical Chemistry Laboratory for ^{15}N using a Carlo Erba NA 1500 CHN Analyzer.

All statistical analyses were performed using a significance level of $\alpha = 0.05$ in SAS. Results for individual variables (e.g., nutrient concentrations, flow rates) were tested for normality using PROCUNIVARIATE (SAS Institute Inc, 2013a). Appropriate data transformations were made in order to achieve a normal distribution. PROCGLM was used to test the significance of the difference in the mean baseflow measurements from LDS and HDS watersheds and to estimate linear regression statistics and model parameters of regression models with baseflow or water quality parameters as a function of OWTS density within the watersheds. PROCGLM was also used to compare the regression slopes and intercepts of the regression models from LDS and HDS watersheds (SAS Institute Inc, 2013b).

Baseflow was expected to be significantly higher in HDS watersheds and to increase with increasing OWTS density to support the findings reported by Landers and Ankcorn (2008) and because groundwater and surface water systems are well connected in this region (Clarke and Peck, 1991). A positive correlation between EC and OWTS density was also expected to support the findings reported by Landers and Ankcorn (2008). EC, or specific conductance, is the measure of how well a solution conducts electricity. It is a common indicator of pollutants in water and has been shown to increase with increasing urbanization, wastewater inflows, and watershed disturbance (Dow et al., 2006; Dow and Zampella, 2000; Rose, 2007). NO_3^- concentrations were expected to increase with increasing OWTS density because it is considered to be the most mobile N species in soils and can be easily leached from a OWTS drainfield to groundwater which can contaminate surface water supplies (DHR, 2007).

Cl^- was used as a conservative tracer to detect N transformations and the effect of dilution within the watersheds. Both Cl^- and NO_3^- experience adsorption to a low degree due to the anion exchange capacity of Southern Piedmont soils (Gupte et al., 1996), but Cl^- is not subject to transformation by microbial activity, and it has been shown to increase linearly with increasing NO_3^- concentrations. The NO_3^-:Cl^- ratio has been used as evidence of denitrification in an OWTS drainfield and along groundwater flow paths to streams (Bradshaw and Radcliffe, 2011; Lowrance, 1992). Cl^- also serves as a good indicator parameter for OWTS impacts because it is present in all sewage (McQuillan, 2004). Therefore, Cl^- concentrations were expected to increase with increasing OWTS density.

The amount of ^{15}N, a stable isotope of N, was also determined to aid in identifying the sources of N within the watersheds. Biological organisms preferentially use the lighter isotope of nitrogen, ^{14}N, rather than the heavier isotope, ^{15}N, for respiration and assimilation because the chemical bonds of lighter isotopes are generally broken down easier than those of heavier isotopes. As a result, ^{14}N becomes concentrated in cell mass while ^{15}N becomes concentrated in the residual N sources. NO_3^- in groundwater that has been denitrified by microbes or originates from human or animal waste is enriched with ^{15}N (McQuillan, 2004). Distinct isotopic compositions have been identified to characterize N of different origin so that ^{15}N can be measured to distinguish between human and animal waste, soil organic nitrate-N, and synthetic fertilizers (Aravena et al., 1993). N isotopes are expressed as $\delta^{15}N$ in units per mil higher or lower than that of atmospheric ^{15}N (Silva et al., 2002). Typical $\delta^{15}N$ values range from -2 ‰ to +4 ‰ for commercial fertilizers, +3 ‰ to +8 ‰ for soil organic nitrate-N, and from 10 ‰ to 25 ‰ for human and animal wastes (Aravena et al., 1993). McQuillan (2004) reported $\delta^{15}N$ values of groundwater NO_3^- originating from OWTSs that ranged from +7.6 ‰ to +12.1 ‰. Therefore, δ15N values in the HDS watersheds were expected to be significantly higher than the LDS watersheds and in the range of values associated with human wastes.

RESULTS AND DISCUSSION

Electrical Conductivity and Chloride

Figure 3.2 shows EC and Cl^- concentrations as a function of OWTS density within the watershed for all four sampling events. Analysis revealed a linear increase in EC and Cl^- concentrations with OWTS density within the watershed (R^2=0.5211 for EC and R^2=0.5596 for Cl^-). Results imply the presence of OWTS effluent in streams of watersheds with HDS and an increase in the presence of OWTS effluent with an increase in OWTS density. While Cl^- can originate from other sources that include agricultural runoff, landfill leachates, industrial effluents, water softeners, pool salts, or road de-icing (World Health Organization, 1996), the source of Cl^- in the watersheds of this region was assumed to be from OWTS effluent. However, further analysis is needed to correctly identify the dominate sources of Cl^- within these watersheds.

Baseflow

Average baseflow yield (discharge per unit watershed area) measurements in November 2011, March 2012, and November 2012 were not statistically different between LDS and HDS watersheds, but July 2012 measurements were significantly higher in streams of watersheds impacted by HDS (Table 3.2 and Figure 3.3). Figure 3.4 and Figure 3.5 show the regression analysis of baseflow yield as a function of OWTS density with the watershed for all sampling events. Analysis revealed a weak correlation between baseflow yield and OWTS density in November 2011 (R^2=0.0527), March 2012 (R^2=0.0121), and November 2012 (R^2=0.0194), but July 2012 measurements indicated a linear increase in baseflow yield with OWTS density (R^2=0.2877).

Baseflow results imply the presence of OWTS effluent in HDS watersheds despite the lack of significant difference between mean values in the watersheds in three out of the four sampling events. An important characteristic of the HDS watersheds is that they are significantly higher in impervious surfaces than the LDS watersheds (Table 3.1). As mentioned above, studies have shown that watersheds with a greater percentage of impervious surfaces are expected to have a lower baseflow than watersheds with a low percentage of impervious surfaces due to less infiltration, less groundwater recharge, and more runoff. However, the results show no significant difference in the baseflow yield in November 2011, March 2012, and November 2012 and a significantly higher baseflow yield in July 2012 implying an increase in baseflow in the HDS watersheds due to OWTS effluent which may offset the effects of impervious surfaces.

In July 2012 the mean baseflow yield was approximately 6 times greater in the HDS watersheds (Table 3.2). The large difference for this sampling event may be explained by the water deficit (precipitation minus the evapotranspiration losses) during the time of the sampling. The month prior to the July sampling had the largest water deficit of all sampling periods (Table 3.3) indicating very dry conditions. The results imply that the influence of OWTSs on baseflow within the watersheds of this region is greatest under drought conditions.

The baseflow yield varied significantly (p-value <0.001) between sampling periods that may or may not be explained by seasonality. Variations could have been caused by point sources within the watershed such as leaking water supply lines or by measurement errors under extremely low flow conditions. Further analysis is needed to correctly identify reasons for the variability in order to accurately determine the effect of OWTSs on the baseflow in this region.

Nitrogen

Figure 3.6 shows NO_3^- concentrations as a function of OWTS density within the watershed for all four sampling events. Separate linear regression lines were fit to the data from both the LDS and HDS watersheds, and the slope and intercept were shown to be significantly different (p-value <0.0001). Analysis of the NO_3^- concentrations in the LDS watersheds revealed a linear decrease in concentration with OWTS density within the watershed (R^2=0.2807) while analysis of the NO_3^- concentrations in the HDS watersheds revealed a linear increase in concentration with OWTS density within the watershed (R^2=0.6508). Results imply an increase in NO_3^- concentration with OWTS density within the watershed above a threshold of about 100 OWTSs per square kilometer. Below this threshold, NO_3^- concentrations were more variable and decreased with OWTS density possibly due to a transition from less agricultural land use to more urban land use.

Variability of the NO_3^- concentrations in the LDS watershed may be explained by the presence of sources other than OWTSs. Some LDS watersheds appeared to have high NO_3^- concentrations that may have originated from agricultural runoff, animal wastes, or leaking sewer lines. $\delta^{15}N$ values in these watersheds were in the range reported for human and animal wastes (+7.6 ‰ to 25 ‰) which imply the presence of NO_3^- in some LDS watersheds that was derived from human or animal wastes or from other sources that have undergone denitrification (Figure 3.7a).

Figure 3.7b shows $\delta^{15}N$ as a function of OWTS density within the watershed. Linear regression lines were fit to the data from both the LDS and HDS watersheds, and the slope and intercept were shown to be significantly different (p-value <0.0001). Analysis of the $\delta^{15}N$ values in the LDS watersheds revealed a linear decrease with OWTS density within the watershed ($R^2=0.2829$) while analysis of the $\delta^{15}N$ values in the HDS watersheds revealed a linear increase with OWTS density within the watershed ($R^2=0.5603$). Results imply an increase in NO_3^- originating from OWTS effluent with an increase in OWTS density within the watershed above a threshold of about 100 OWTSs per square kilometer. Below this threshold, results indicate variability due to the presence of non-OWTS sources of NO_3^-. Further analysis of potential point and non-point sources of N within the watersheds of this region is needed to account for the variability and to develop a more accurate assessment of the effect of OWTSs on the N load.

CONCLUSIONS

Synoptic samples and discharge measurements of streams affected by low and high density OWTSs were taken four times in November 2011, March 2012, July 2012, and November 2012 under baseflow conditions. EC and Cl^- results indicated the presence of OWTS effluent in streams of watersheds with HDS and an increase in the presence of OWTS effluent with an increase in OWTS density within the watershed.

Baseflow results suggested an increase in discharge in the HDS watersheds due to the presence of OWTS effluent which may off-set the effects of impervious surfaces. Results also imply that during drought conditions, effluent from OWTSs can significantly influence the baseflow in small streams. However, further analysis is needed to correctly identify reasons for variability in order to accurately determine the effect of OWTSs on the baseflow in this region.

Analysis of the NO_3^- concentrations in the LDS watersheds revealed a linear decrease in concentration with OWTS density within the watershed while analysis of the NO_3^- concentrations in the HDS watersheds revealed a linear increase in concentration with OWTS density. Results imply that above a threshold of about 100 OWTS per square kilometer, NO_3^- concentrations increased linearly with OWTS density. Below this threshold, NO_3^- concentrations decreased with OWTS density possibly due to a transition from less agricultural land use to more urban land use and were more variable due to the presence of non-OWTS sources of NO_3^- within the watershed.

This study showed that OWTSs have positive and negative impacts on the water quality and quantity of urbanizing watersheds of this region. It provided data that may be used to inform users as well as watershed planners about the influence of OWTSs on the N load and baseflow in streams based on the density of OWTSs within the watershed. Future research goals are to quantify the contribution of OWTSs to the discharge and N load through the use of hydrologic models in order to better asses their impact on water quality and quantity.

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Table 3.1. The characteristics of the watersheds in the study area in Gwinnett County, Georgia (Landers and Ankcorn, 2008)

Watershed ID	HDS or LDS	Drainage area (km^2)	Count of OWTSs	Density of OWTSs (per km^2)	Median distance to stream (m)	Watershed imperviousness (percent)	Mean slope (percent)
1	LDS	8.39	70	8	162.76	4.2	8.8
2	LDS	1.55	15	10	126.49	3.3	10.6
3	LDS	2.67	37	14	162.76	4.3	8.5
4	LDS	0.62	22	35	171.60	11.6	7.3
5	LDS	1.48	30	20	85.65	5.4	5.8
6	LDS	5.28	82	16	107.59	4.1	6.5
7	LDS	1.11	20	18	89.92	6.3	10.6
8	LDS	1.27	22	17	94.18	3.0	9.2
9	LDS	2.95	81	27	159.11	7.8	7.7
10	LDS	4.40	152	35	118.57	7.3	8.3
11	LDS	4.20	105	25	119.48	7.6	7.8
15	LDS	1.68	62	37	140.21	15.2	4.6
12	HDS	3.29	378	115	104.85	12.3	9.1
13	HDS	8.81	779	88	116.74	13.2	8.0
14	HDS	1.74	245	141	103.94	16.1	8.5
16	HDS	2.59	486	188	99.36	26.4	5.7
17	HDS	1.68	384	228	138.38	20.1	7.5
18	HDS	0.98	302	307	150.57	18.4	7.4
19	HDS	0.18	72	397	105.46	20.3	7.8
20	HDS	0.54	159	292	83.21	18.3	6.0
21	HDS	1.14	246	216	63.40	17.5	8.6
22	HDS	1.94	304	156	62.79	19.9	7.0
23	HDS	0.52	120	232	64.92	18.4	7.3
24	HDS	0.67	173	257	54.56	20.0	7.6
Mean	**LDS**	**2.97**	**58**	**22**	**128.19**	**6.7**	**8.0**
Mean	**HDS**	**2.01**	**304**	**218**	**95.68**	**18.3**	**7.5**

Table 3.2. Mean baseflow yield in streams of watersheds with low density (LDS) and high density (HDS) on-site wastewater treatment systems in November 2011, March, 2012, July 2012, November 2012, and from results reported by Landers and Ankcorn (2008).

	Baseflow Yield (10^{-3}) (m^3 s^{-1} km^{-2})			
Sampling Event	LDS	HDS	HDS/LDS	P-value†
Oct, 2007*	1.96	3.83	1.95	0.01
Nov, 2011	2.68	2.88	1.07	0.46
Mar, 2012	6.22	6.95	1.12	0.26
July, 2012	0.99	6.12	6.18	<0.001
Nov, 2012	2.10	2.56	1.22	0.54

*Results reported by Landers and Ankcorn (2008)

†P-value <0.05 indicates the mean baseflow yield in the LDS and HDS watersheds were significantly different.

Table 3.3. Precipitation and evapotranspiration data for one month prior to each synoptic sampling event. Information obtained from the Georgia Automated Environmental Monitoring Network (2011).

Sampling Event	Precipitation (cm.)	Evapotranspiration (cm.)	Water Balance (cm.)
Oct, 2007*	1.57	2.64	-1.07
Nov, 2011	7.54	7.59	-0.05
Mar, 2012	7.90	6.43	1.47
July, 2012	16.2	18.6	-2.4
Nov, 2012	3.25	3.73	-0.48

*Sampling period of results reported by Landers and Ankcorn (2008)

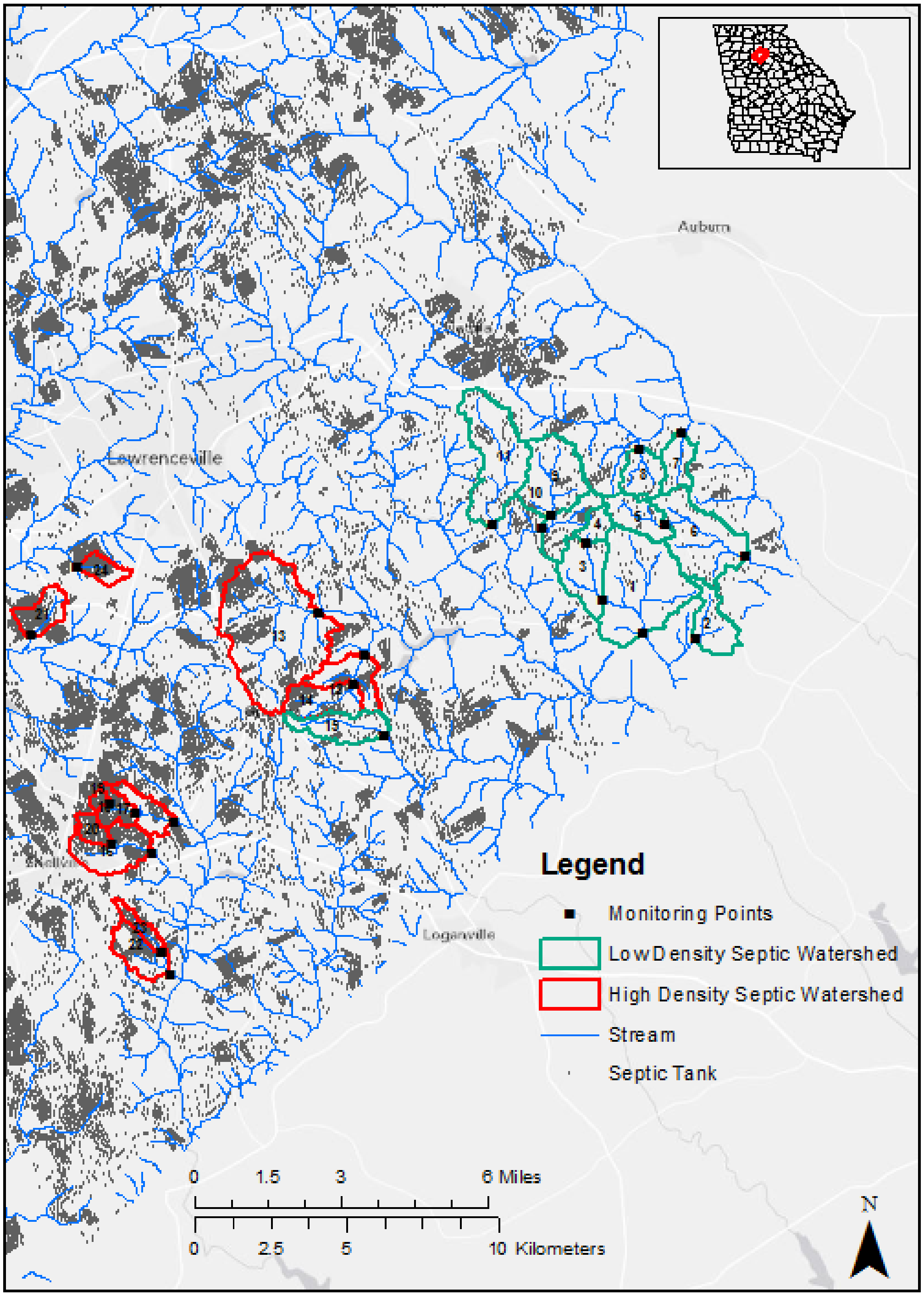
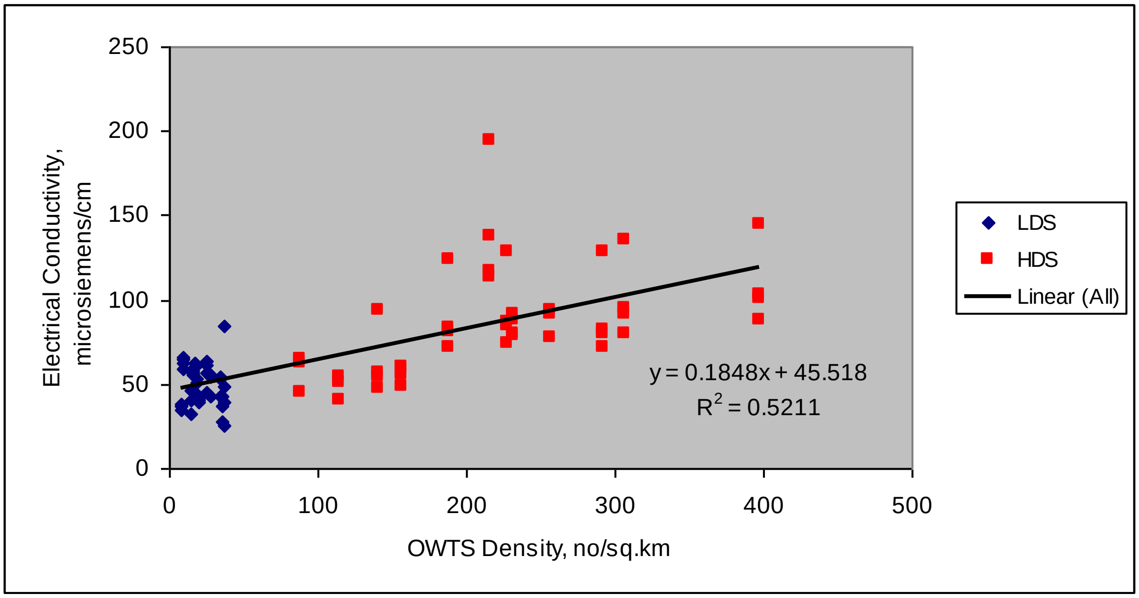


Figure 3.1. Location of the study area, 24 watershed boundaries, sampling sites, and on-site wastewater treatment systems, Gwinnett County GA (Landers and Ankcorn, 2008)

a)



b)

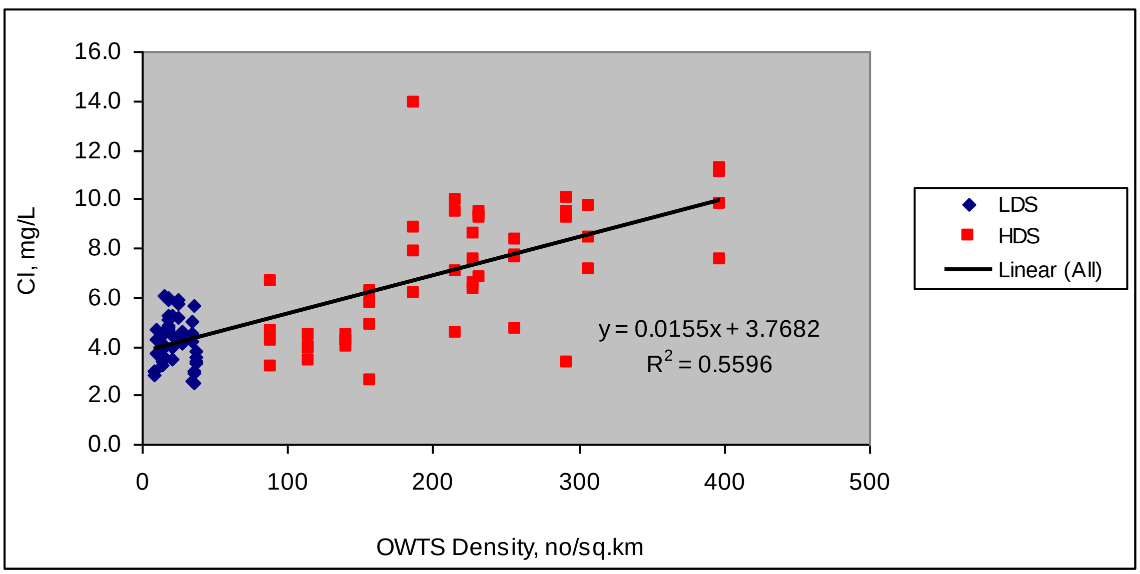
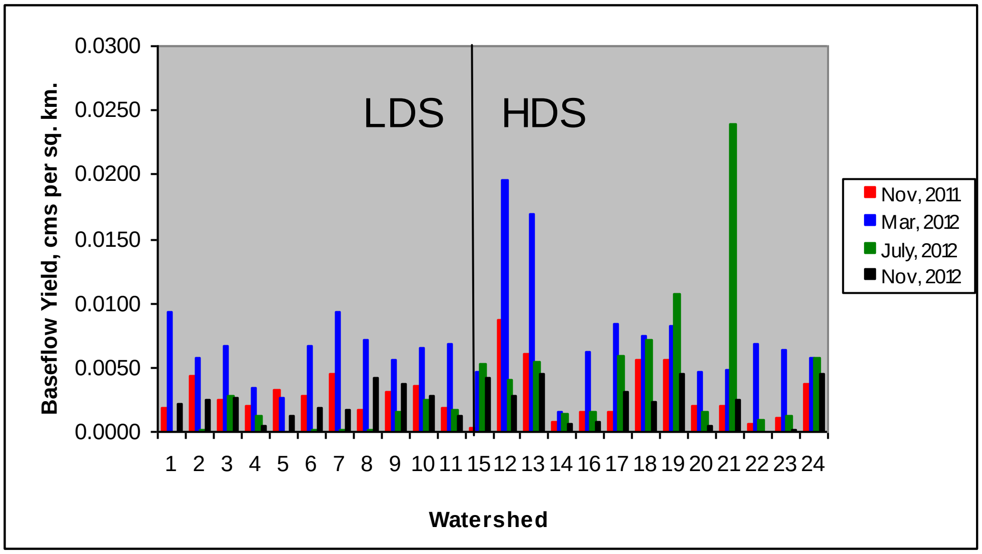


Figure 3.2. Electrical conductivity (EC; a) and chloride (Cl^-; b) concentration as a function of on-site wastewater treatments system (OWTS) density within the watersheds in November 2011, March, 2012, July 2012, and November 2012

a)



b)

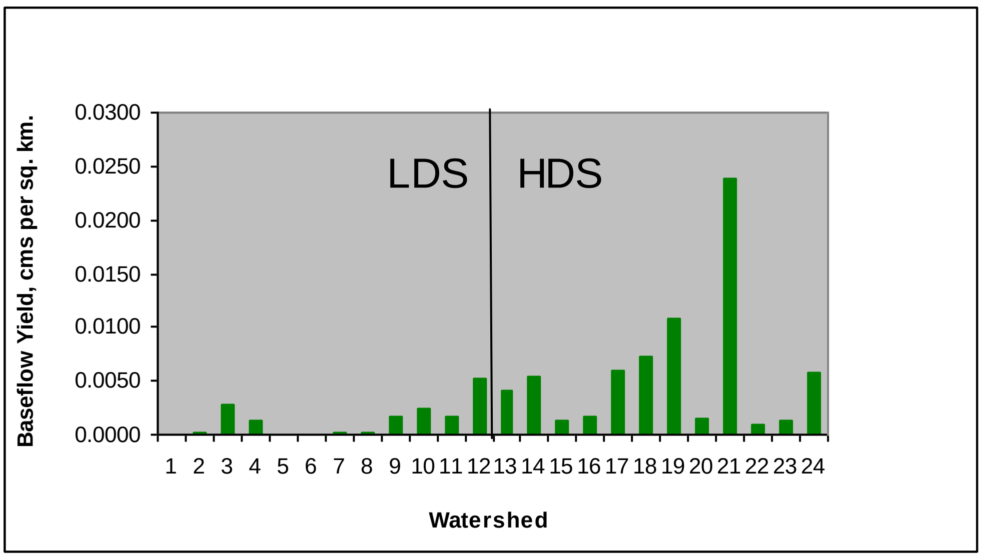
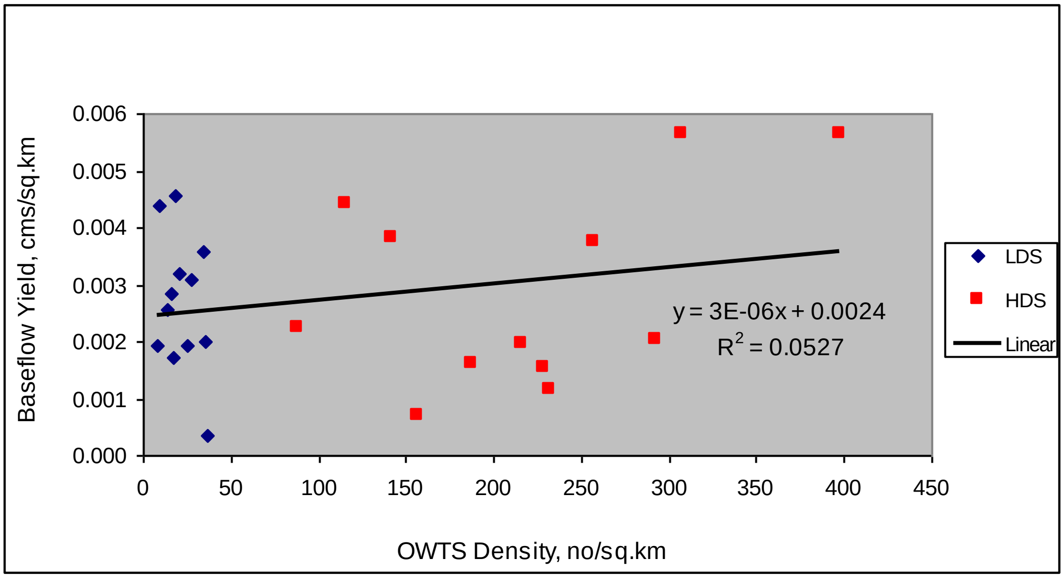


Figure 3.3. Baseflow yield (discharge per unit area) in streams of watersheds with low density (LDS) and high density (HDS) on-site wastewater treatment systems for all sampling events (a; November 2011, March 2012, July 2012, and November 2012) and in the July 2012 sampling event (b).

a)



b)

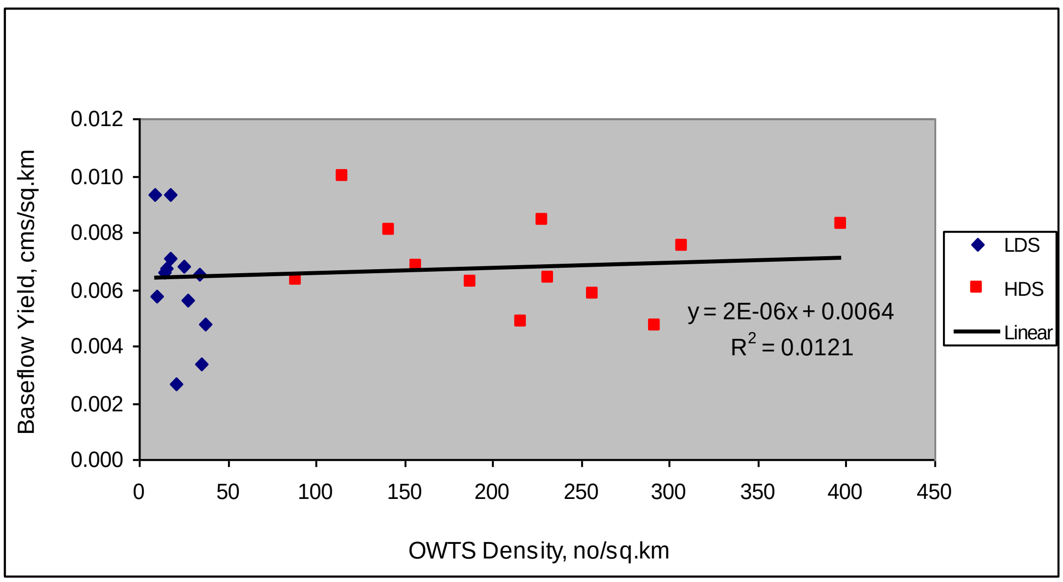
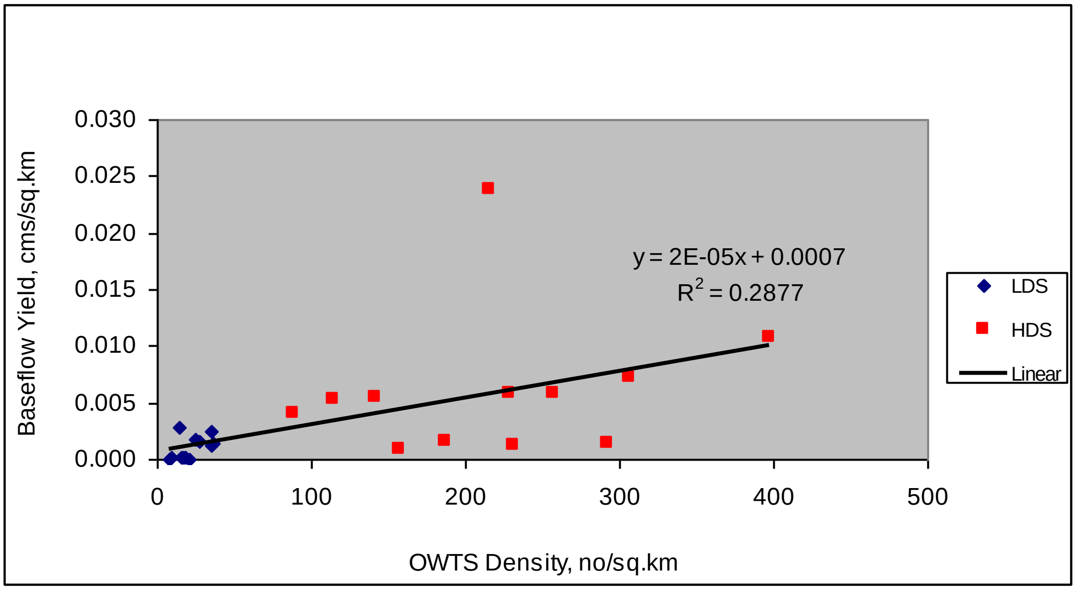


Figure 3.4. Baseflow yield (discharge per unit watershed area) as a function of on-site wastewater treatment system (OWTS) density within watersheds in November 2011 (a) and March 2012 (b)

a)



b)

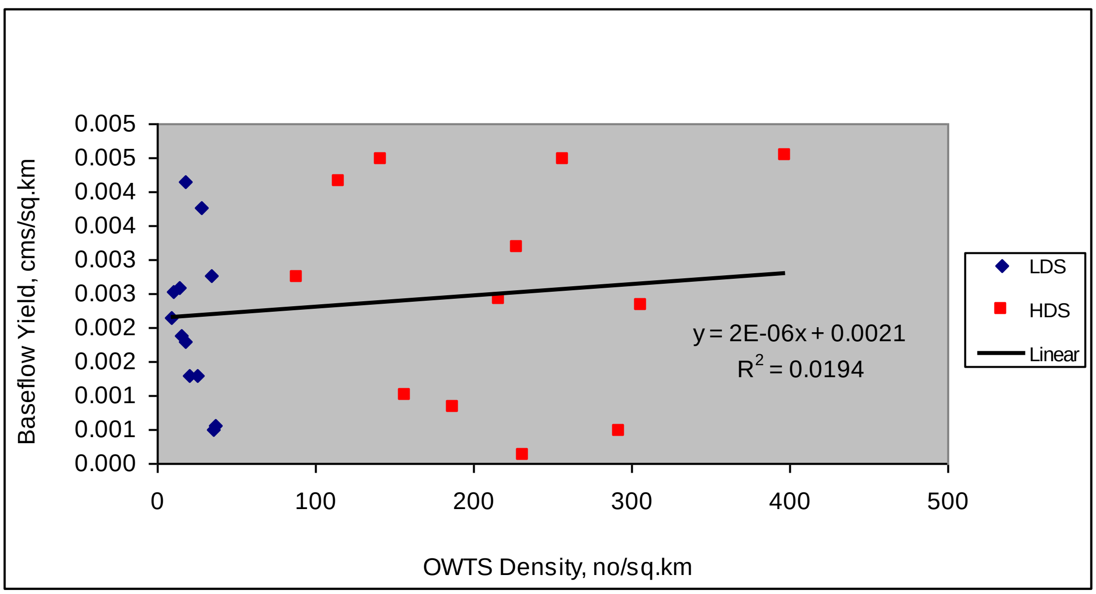


Figure 3.5. Baseflow yield (discharge per unit watershed area) versus on-site wastewater treatment system (OWTS) density within the watersheds in July 2012 (a) and November 2012 (b)

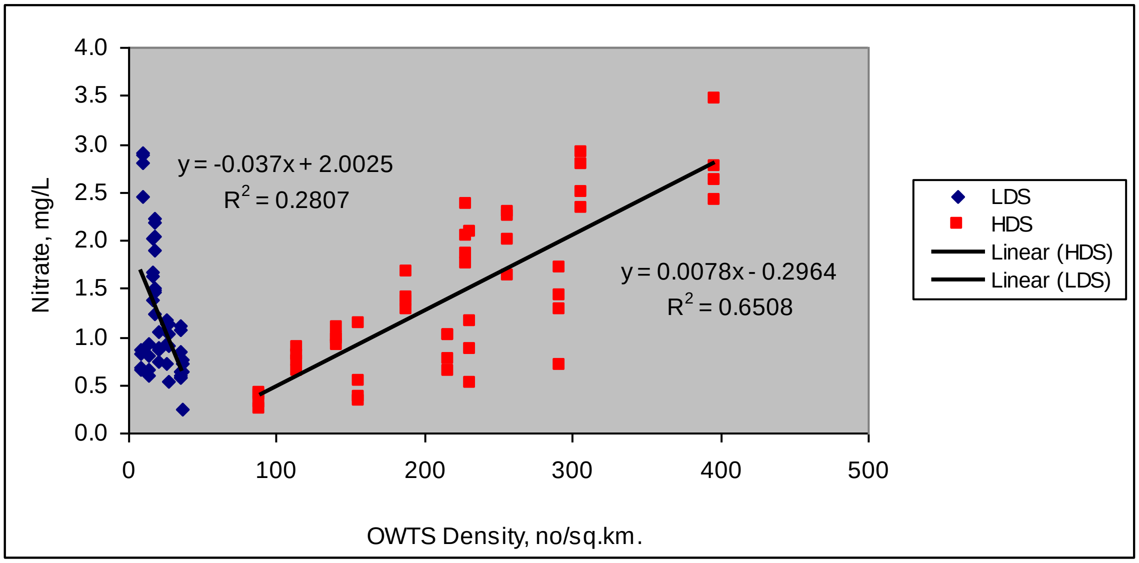
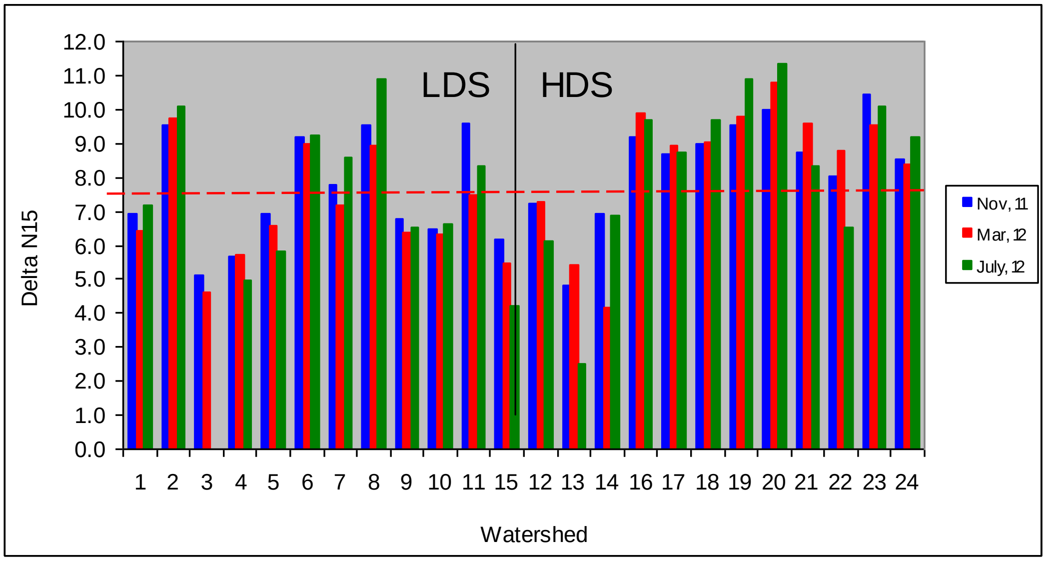


Figure 3.6. Nitrate-N concentration as a function of on-site wastewater treatment system (OWTS) density within the watersheds in November 2011, March, 2012, July 2012, and November 2012

a)



b)

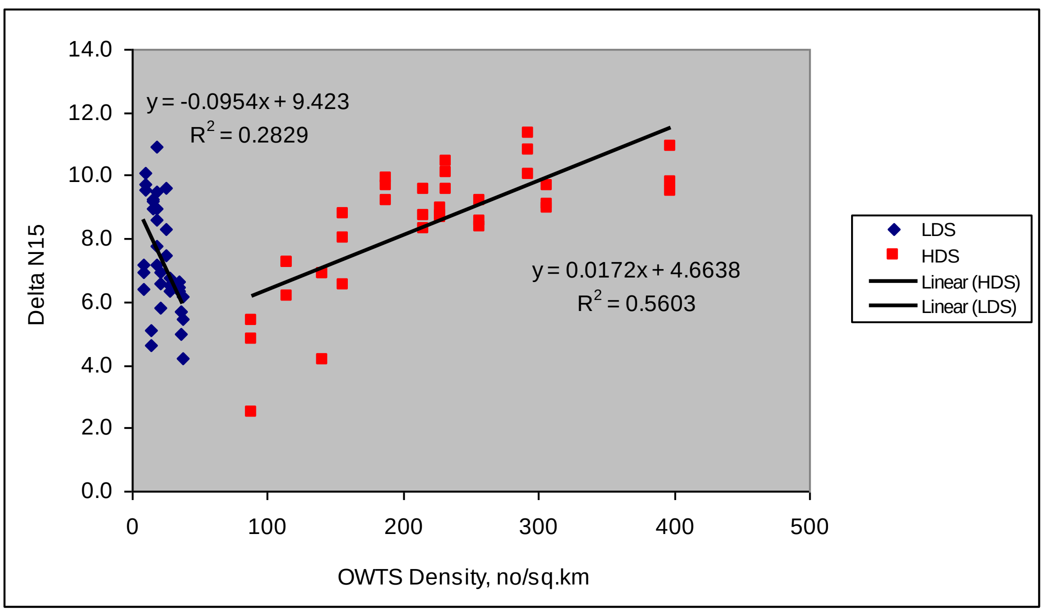


Figure 3.7. δ^{15}N in streams of watersheds with low density (LDS) and high density (HDS) on-site wastewater treatment systems (OWTSs; a) and δ^{15}N as a function of OWTS density within the watershed (b) in November 2011, March 2012, July 2012, and November 2012

CHAPTER 4

QUANTIFYING THE CONTRIBUTION OF ON-SITE WASTEWATER TREATMENT SYSTEMS TO STREAM DISCHARGE USING THE SWAT MODEL[2]

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ABSTRACT

In the Southeastern United States, on-site wastewater treatment systems (OWTSs) are widely used for domestic wastewater treatment. The degree to which OWTSs represent consumptive water use has been questioned in Georgia. The goal of this study was to estimate the effect of OWTSs on stream flow in a gauged watershed in Gwinnett County, Georgia using the Soil and Water Assessment Tool (SWAT) watershed-scale model, which includes a new OWTS algorithm. Stream discharge was modeled with and without the presence of OWTSs in order to quantify their influence on water quantity. The model was calibrated using data from January 1, 2003 to December 31, 2006 and validated from January 1, 2007 to December 31, 2010 using the auto-calibration tool, SWAT-CUP 4. The daily and monthly flow NS coefficients were 0.49 and 0.71, respectively for the calibration period and 0.37 and 0.68, respectively for the validation period indicating a satisfactory fit. Analysis of water balance output variables between simulations with and without the presence of OWTSs showed a 3.1% increase in total water yield at the watershed-scale and a 5.9% increase at the subbasin-scale. The percent change in water yield between simulations was the greatest in dry years implying that the influence of OWTSs on the water yield within the watershed is greatest under drought conditions. Mean OWTS water use was approximately 5.7% consumptive, contrary to common assumptions by water planning agencies in Georgia. Results from this study may be used by OWTS users as well as watershed planners in order to understand the influence of OWTSs on water quantity within watersheds of Metropolitan Atlanta, Georgia.

INTRODUCTION

In the Southeastern United States on-site wastewater treatment systems (OWTSs), commonly known as septic systems, are widely used for domestic wastewater treatment. For example, the number of OWTSs in Metropolitan Atlanta, Georgia is estimated to be 526,000 which is 26 percent of the total housing units in the district (MNGWPD, 2006). As suburban populations continue to increase in this region, the use of OWTSs is expected to increase due to the costs associated with extending sewer systems to the outlying communities. The Metropolitan North Georgia Water Planning District originally considered OWTSs to be 100% consumptive use (water is withdrawn and not returned to streams) for planning purposes (MNGWPD, 2006). However, as direct result of a USGS study done by Landers and Ankcorn (2008), the guidelines were revised to say that the degree of consumptive use was not known, but assumed to be “more consumptive than centralized systems with surface water discharges” (MNGWPD, 2009). Due to the limited availability of high yield wells, surface water withdrawals account for about 78 percent of the public water supply of Metropolitan Atlanta (Clarke and Peck, 1991; Fanning, 2001). Droughts and conflicts with Florida and Alabama over water use by Metropolitan Atlanta have compounded the problem (Appel, 2007). Therefore, as the use of OWTSs in this region increases, their impact on surface water quantity must be determined.

The literature on the impact of OWTSs on stream water quantity suggests that they have a significant influence on baseflow in streams. An increase in impervious surfaces due to urban development within a watershed has been found to decrease baseflow and increase stormwater flow due to less infiltration and more runoff (Calhoun et al., 2003; Landers and Ankcorn, 2008; Landers et al., 2007). However, several studies have shown that OWTSs can offset the effect of urbanization on baseflow through the discharge of effluent into groundwater and eventually streams. Simmons and Reynolds (1982) analyzed 22 years of stream flow records for two watersheds in New York and found a decrease in baseflow with an increase in imperviousness and a transition from OWTSs to centralized sanitary sewer systems. However, the authors found little decrease in baseflow in two nearby watersheds that were undergoing increased urbanization but were unsewered, which suggested an increase in flow due to OWTSs. In a study by Burns et al. (2005) in New York, the authors found that baseflow during dry periods was greatest in a high density residential catchment. It was concluded that the combined effects of natural landscape features such as wetlands or human alterations such as OWTSs can change the expected effects of development within a watershed. Yang et al. (1999) reported similar results using a solute-balance and water balance approach in Nottingham, UK. The authors found that the combined influence of OWTSs, leaking water mains and sewers, and infiltration ponds resulted in rising groundwater levels and increasing baseflow.

The influence of OWTSs on groundwater recharge and baseflow in Metropolitan Atlanta, Georgia was previously investigated by Landers and Ankcorn (2008) of the United States Geological Survey. In this study, 24 small watersheds (0.181 to 8.81 km^2) in Gwinnett County, Georgia were selected based on similar geologic setting, precipitation, climate, ease of access for baseflow measurements, and the availability of spatial datasets. Of the 24 watersheds, twelve were characterized with high density (HDS) OWTSs having greater than 77 OWTSs per km^2 (200 OWTSs mi^2), and twelve were characterized with low density (LDS) OWTSs having less than 39 OWTSs per km^2 (100 OWTSs per square mile). The authors collected one set of synoptic field measurements of the discharge, electrical conductivity, and temperature at baseflow conditions during extreme drought conditions in October, 2007. They found that the mean baseflow yield of the HDS watersheds was 90 percent greater than that of the LDS watershed and that the relationship between density of OWTSs and baseflow yield and electrical conductivity (a common wastewater indicator) was statistically significant. They concluded that a significant factor in explaining the increased baseflow was the density of OWTSs within the watersheds.

Watershed-scale models can be very useful tools for understanding and predicting the effect of OWTSs on water quality and water quantity. Modeling the effects of OWTSs at the watershed-scale has not been extensively researched due to the various uncertainties introduced beyond the drainfield of a single OWTS. However, a new biozone algorithm proposed by Siegrist et al. (2005) was adapted to the Soil and Water Assessment Tool (SWAT) in order to simulate the influence of OWTSs on water quality within a watershed (Jeong et al., 2011). Jeong et al. (2011) used a watershed in the Coastal Plain region of North Carolina to show the model performed well in predicting both groundwater table levels and NO_3^- concentrations in the groundwater, but it was an un-gauged watershed, so comparisons with stream flow were not possible. Because of a wide variety of soils and geology, there is a need for further analysis of the complex water and solute transport processes associated with OWTSs at the watershed-scale.

The objective of this research was to determine the impact of OWTSs on the discharge of a stream in an urbanized watershed in Metropolitan Atlanta, Georgia.

Measured flow data from the gauged watershed was used to calibrate the SWAT model for predicting stream discharge. The stream discharge within the watershed was modeled with and without the presence of OWTSs in order to quantify their influence on water yield.

MATERIALS AND METHODS

The Soil and Water Assessment Tool (SWAT)

The Soil and Water Assessment Tool was used to simulate hydrologic processes with and without the presence of OWTSs in a gauged watershed of Gwinnett County, Georgia. SWAT is a physically based watershed-scale model that was developed to predict the impact of land management practices on water, sediment, and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time (Neitsch et al., 2011). The model has been widely used in water quality modeling studies, the analysis of total maximum daily loads (TMDL), and nonpoint-source pollution analysis (Jeong et al., 2011). The Soil and Water Assessment Tool Users Manual provides a detailed description of the governing equations that the model uses for simulating processes within a watershed (Neitsch et al., 2011). A description of the main equations SWAT uses for simulating flow, sediment, and nutrients are described in Chapter 2.

Study Area

The study area was the Big Haynes Creek watershed located in Gwinnett County, Georgia (Figure 4.1). The watershed drains an area of approximately 44.0 km^2, and the mean elevation is 297 m. The land cover consists of approximately 38% medium density residential development, 28% low density residential development, and 24% forest. The average annual rainfall of the region is about 1270 mm. Observed flow data at the outlet of the watershed was obtained from the United States Geological Survey (USGS) gauge number 02207385: Big Haynes Creek at Lenora Road.

SWAT Model Data

The ArcGIS-ArcView extension and graphical user input interface (ArcSWAT) was used for the model set-up and development. Basic user inputs for the model include land cover, a digital elevation model (DEM), soils data and OWTS use. Land cover information was downloaded from the National Land Cover Database developed by the Multi-Resolution Land Characteristics Consortium (Fry, 2011).

Information on OWTSs within Gwinnett County was obtained from the Gwinnett County Geographic Information Systems (GIS) database (Gwinnett County Board of Commissioners, 2013). The Gwinnett County Water Resources Office digitized paper OWTS records and added them to the Gwinnett GIS database to show the location of septic tanks and drain fields. The most recent OWTS layer (2011) was downloaded as a point feature class and then converted to a raster with a grid size of 10 meters by 10 meters to represent the average size of a OWTS drainfield (100 m^2). The OWTS raster was then merged with the NLCD land use map, and a new "septic" land use was defined in the land use layer.

The digital elevation model (DEM) used in the SWAT model was developed by the USGS Earth Resources Observation and Science (EROS) Data Center. The National Elevation Dataset (NED) was downloaded for Gwinnett County at 1:24,000-scale with a spatial resolution of 30-meters (U.S. Geological Survey, 2013).

The SSURGO soils database from the Natural Resources Conservation Service (NRCS) was used for soils information in the SWAT model. The SSURGO dataset for Gwinnett County was downloaded from the NRCS Soil Data Mart at a scale of 1:12,000. In order to use the SSURGO data in the model, a SWAT/SSURGO soils database containing all of the SWAT soil attributes associated with SSURGO soil values in the United States was downloaded from the SWAT website (Texas A&M University and USDA-ARS, 2013).

Watershed Delineation and Hydrologic Response Unit (HRU) Distribution

The Big Haynes Creek watershed was delineated from the DEM into subbasins using the automatic delineation tool in the ArcSWAT interface, and a watershed outlet was manually added corresponding to the location of the USGS gauging station. A total of 35 subbasins were delineated for the watershed based on topographic and stream network data (Figure 4.1).

SWAT uses land use, soils, and slope data to determine the hydrologic response unit (HRU) distribution in each sub-watershed. Three slope classifications were defined: 0-5%, 6-10%, and >10%, respectively. The sub-watersheds were divided into one or more HRUs based on unique combinations of land use (including OWTS drainfields), soils, and slope in order to reflect the spatial variability and account for the differences in the processes that affect the water balance (Neitsch et al., 2011). A threshold value of 10% over the sub-basin area was applied for land use, and a threshold value of 30% over the sub-basin area was applied for soils and slope classes. The thresholds eliminated minor land uses, soil types, or slope classes in order to create a reasonable number of HRUs. The septic land use category was added as an exempt land use due to the relatively small percentage of the subbasin that OWTS drainfields cover. A total of 263 HRUs were created.

Weather Data

Daily weather data was obtained from the National Centers for Environmental Prediction (NCEP) Climate Forecast System Reanalysis (CFSR). CFSR data for daily precipitation, wind speed, relative humidity, and temperature was downloaded in SWAT file format using a website tool by Texas A&M University (2013). Data from one weather station was used for simulations in SWAT. The station is located approximately 7.2 km to the west of the main reach of the watershed at -84.0625 W, 33.8768 N.

SWAT Model Calibration

Model parameters are set to default values in SWAT when the model is set up for a watershed. The model may be calibrated by adjusting the parameters governing flow to obtain a best possible fit between the simulated model output and the observed data. Calibration of the model in this study was performed using SWAT Calibration and Uncertainty Programs Version 4 (SWAT-CUP 4), which is an auto-calibration tool that links several procedures to SWAT that allow for sensitivity analysis, calibration, validation, and uncertainty analysis of the model (Abbaspour, 2011). The Sequential Uncertainty Fitting Version 2 (SUFI-2) algorithm was used to estimate the SWAT parameters related to flow and to evaluate the goodness-of-fit of the model. SUFI-2 combines calibration and uncertainty analysis to find parameter uncertainties that result in prediction uncertainties bracketing most of the measured data, while producing the smallest possible prediction uncertainty band (Rotstamian et al., 2008). SUFI-2 accounts for all sources of uncertainties such as uncertainty in driving variables (e.g. rainfall), conceptual model parameters, and measured data. This is quantified by the P-factor, which is the percentage of measured data bracketed by the 95% prediction uncertainty (95PPU). The 95PPU is calculated at the 2.5% and 97.5% levels of the cumulative distribution of an output variable obtained from Latin hypercube sampling (Abbaspour, 2011). SUFI-2 initially assumes a large parameter uncertainty and then decreases this uncertainty in steps while monitoring the P factor and R-factor. The R-factor is the average thickness of the 95PPU band divided by the standard deviation of the measured data. In each step, previous parameter ranges are updated by calculating the sensitivity matrix, 95% confidence intervals of the parameters, and the correlation matrix. The new parameter ranges are always smaller than the previous ranges and are centered around the best simulation (Abbaspour, 2011). The value for P-factor ranges between 0 and 1 while the value for R-factor ranges between 0 and infinity. A P-factor of 1 and R-factor of zero corresponds to a perfect fit between measured and predicted values. A larger P-factor can be achieved at the expense of a smaller R-factor; therefore a balance must be reached between the two values (Abbaspour, 2011).

The goodness of fit and uncertainty of the model stream flow in this study was assessed by the Nash-Sutcliff (NS) model efficiency coefficient and the R^2 value. The NS model efficiency coefficient is determined by the following equation:

$$NS = 1 - \frac{\sum\left(Q_m - Q_{m,avg}\right)^2 - \sum\left(Q_m - Q_p\right)^2}{\sum\left(Q_m - Q_{m,avg}\right)^2} \qquad \textbf{Eqn. 4.1}$$

where, Q_m is the measured stream flow, $Q_{m,avg}$ is the arithmetic average of the measured stream flow, and Q_p is the predicted stream flow. The NS coefficient is the sum of the deviations of the observations from a linear regression line with a slope of 1. It is expected to be between 0 and 1, where a value of 1 would be a perfect fit between measured and predicted values. If the NS is negative, predictions are poor, and the average value of the output is a better estimate than the model prediction (Nash and Sutcliffe, 1970; S.Grunwald and Frede, 1999). The R^2 value describes how well a linear regression line fits the dataset. The value ranges from 0 to 1 where a value of 1 would be a perfect fit (Ott and Longnecker, 2001).

SWAT Calibration and Input Parameters

The parameters in SWAT affecting watershed hydrology that are most commonly used in calibration procedures were identified through literature review (Arnold et al., 2012; Cibin et al., 2010; Feyereisen et al., 2007; Moriasi et al., 2007; Reungsang et al., 2007; Zhang et al., 2010). A general description of the parameters selected for calibration and their range of values are shown in Table 4.1.

The runoff curve number (CN2), soil evaporation compensation factor (ESCO), surface runoff lag coefficient (SURLAG), and the available soil water capacity (SOL_AWC) are parameters associated with surface water response. CN2 directly impacts surface runoff and is a function of the soil's permeability, land use and antecedent soil water conditions. ESCO controls the soil evaporative demand that is to be met from different soil depths. SURLAG is the surface runoff storage feature that lags a portion of the surface runoff release to the main channel. In large subbasins with a time of concentration greater than 1 day, only a portion of the surface runoff will reach the main channel on the day it is generated. SSURLAG controls the fraction of total available water that will be allowed to enter the reach on any one day. SOL_AWC is estimated by subtracting the fraction of the water present at field capacity from the fraction of the water present at wilting point (Arnold et al., 2011).

Parameters associated with groundwater flow are the groundwater re-evaporation coefficient (GW_REVAP), the shallow aquifer depth threshold value (GWQMN), the groundwater delay time (GW_DELAY), the bank storage factor (ALPHA_BNK), and the baseflow recession constant (ALPHA_BF). GW_REVAP controls the water movement from the shallow aquifer into the overlying unsaturated zone. GWQMN is the threshold depth of water in the shallow aquifer required for return flow to the reach to occur. GW_DELAY is the time for water leaving the bottom of the root zone to reach the shallow aquifer. ALPHA_BNK is the exponential decay factor for bank storage which characterizes the bank storage recession curve. ALPHA_BF is the exponential decay factor for groundwater flow to the stream (Arnold et al., 2011). The value for ALPHA_BF was estimated using measured stream flow and an automated baseflow filter program developed by Arnold et al. (1995).

Parameters affecting the flow of water in the channel network of the watershed are the effective hydraulic conductivity of the alluvium in the main channel (CH_K2) and the Manning's roughness coefficient of the main channel (CH_N2; (Arnold et al., 2011).

The input parameters associated with water use by OWTSs within the watershed are shown in Table 4.2. The data contained in the septic input file are: type of OWTS, geometry of biozone, characteristics of biomass, and bio-physical reaction coefficients occurring in the biozone (Arnold et al., 2011). All default parameter values were used except for the initial septic HRU operational condition (ISEP_OPT) and the number of permanent residents in the house on the system (SEP_CAP). For simulations with the presence of OWTSs in the watershed, ISEP_OPT was defined as “active”, and for simulations without the presence of OWTSs in the watershed, ISEP_OPT was defined as “non-septic.” According to the United States Census Bureau (2013) the average number of persons per household in Gwinnett County, Georgia is three. Therefore, the value for SEP_CAP was set to three for simulations with the presence of OWTSs in the watershed.

None of the OWTS parameters were used in the calibration procedure because they are not available in SWAT-CUP 4. The OWTS algorithm in SWAT simulates OWTS failure due to clogging of the biomat, in which case OWTS effluent rises to the soil surface and becomes part of runoff. This typically happens when an OWTS reaches an age of 10-25 years in the simulation (Jeong et al., 2011). Age of the OWTS is an input variable and for our study we set the initial age to zero for all of the OWTS so that there were no failing OWTSs. The Metropolitan North Georgia Water Management District has estimated (using records of repairs) that 1% of the OWTSs are in failure at any given time (MNGWPD, 2006).

SWAT Model Simulations

The model was first calibrated with the presence of OWTSs on a daily time step for the four-year period from January 1, 2003 to December 31, 2006. It was allowed to warm-up for two years prior to the starting simulation date to allow groundwater and soil storage pools to equilibrate. After calibration, the model was simulated for another four-year period from January 1, 2007 to December 31, 2010 for validation. The model output was compared to the observed data to make sure NS and R^2 values were still reasonable.

In order to observe the effect of OWTSs on water quantity within the watershed, model simulations were run with and without the presence of OWTSs for the eight year period from January 1, 2003 to December 31, 2010. Model output values associated with the water balance at the watershed-scale and subbasin-scale were evaluated for each simulation. The water balance output values observed were: the actual evapotranspiration (ET, mm), the percolation past the root zone (PERC, mm), the surface runoff contribution to the stream (SURQ, mm), the lateral flow contribution to the stream (LATQ, mm), the groundwater contribution to the stream (GWQ, mm), and the water yield or the total amount of water leaving the watershed or subbasin (WYLD, mm).

Watershed-scale output variables for simulations with and without the presence of OWTSs were observed at the outlet of the watershed. Subbasin-scale output variables were observed at the outlet of subbasin #13 because it had the highest septic land use density in the watershed. The subbasin had a total area of 3.28 km^2 and a septic land use area of 0.053 km^2, which is equivalent to approximately 162 OWTSs per km^2. Figure 4.2 shows the distribution of OWTSs within the watershed and subbasins.

Percent consumptive water use by OWTSs was calculated at the watershed-scale and at the subbasin-scale by using the following equation:

$$\left[1-\left(\frac{WYLD_{Septic}-WYLD_{Without}}{OWTS_{\inf low}}\right)\right]\times 100 \qquad \textbf{Eqn. 4.2}$$

where $WYLD_{septic}$ (mm) is the total water yield at the outlet of the watershed or subbasin in the simulation with OWTSs, $WYLD_{without}$ (mm) is the total water yield at the outlet of the watershed or subbasin in the simulation without the presence of OWTSs, and $OWTS_{inflow}$ (mm) is the inflow from all OWTSs within the watershed or subbasin. SWAT assumes a septic inflow of 0.227 m^3/person/day based on a study of the literature by McCray et al. (2005).

RESULTS AND DISCUSSION

Model Calibration

Final values of the parameters adjusted to calibrate flow and the corresponding p-values for sensitivity are shown in Table 4.1. The most sensitive parameters were (in order of decreasing sensitivity): ALPHA_BNK, CH_N2, CH_K2, SOL_K, CH_K1, CN2, and GW_DELAY. Figure 4.3 shows the plot of the observed flow, the model best estimation of the flow, and the 95% prediction uncertainty bands for the simulation, and Table 4.3 gives the simulation NS coefficients and R^2 values. For daily flow, the NS coefficient and R^2 value were 0.49 and 0.50, respectively. For monthly flow, the NS coefficient and R^2 value were 0.71 and 0.74, respectively. The calibration P-factor and R-factor for daily flow were 0.90 and 0.72, respectively. For the validation period daily flow, the NS coefficient and R^2 value were 0.37 and 0.46, respectively. For monthly flow, the NS coefficient and R^2 value were 0.68 and 0.69, respectively (Table 4.3). According to Moriasi et al. (2007), a model fit for stream flow can be considered satisfactory if the monthly flow NS is greater than 0.5. They also indicated that it is expected that the daily NS will be less than the monthly NS, and that the validation NS will be less than the calibration NS.

The annual precipitation for each year of the simulation is shown in Table 4.4. Both the calibration and validation periods included wet (2003, 2005, and 2009) and dry (2006 and 2007) years. But the validation period included the wettest year (2009 with 1897 mm) and the driest year (2007 with 868 mm) so the extreme conditions probably contributed to the lower NS in the validation period. The main limit on accuracy was probably the fact that precipitation came from only one location, which was outside of the watershed. There were clearly days when the observed streamflow responded to a rainfall event, but no precipitation was recorded at the weather station.

Analysis of the Water Balance

The calibrated SWAT model was simulated for the eight-year period from January 1, 2003 to December 31, 2010 with and without the presence of OWTSs in order to determine the effect on water quantity within the watershed. For both simulations the daily flow NS coefficient and R^2 value were 0.44 and 0.47, respectively. The monthly flow NS coefficient and R^2 value were 0.72 and 0.73, respectively (Table 4.3).

Figure 4.4a shows the water balance output variables at the watershed-scale for the simulations with and without OWTSs. Analysis showed a 3.1% increase in the total water yield with the addition of OWTSs in the watershed (Figure 4.4b). The increase in water yield was mostly contributed by the variables associated with groundwater hydrology of the watershed due to the addition of OWTS effluent in the groundwater portion of the water balance. The average annual groundwater contribution to the stream had the greatest percent increase beween simulations (Figure 4.4b). Overall, changes in the total water yield between simulations with and without OWTSs at the watershed-scale were small because septic HRU's represent only 0.88% of the total watershed area.

Figure 4.5a shows the water balance output variables at the subbasin-scale for the simulations with and without OWTSs in the high-density subbasin #13. Analysis showed a 5.9% increase in the total water yield with the addition of OWTSs in the subbasin (Figure 4.5b). Results were consistent with the watershed-scale simulation in that the increase in water yield was mostly contributed by the variables associated with groundwater hydrology of the subbasin due to the addition of OWTS effluent in the groundwater portion of the water balance. The average annual groundwater contribution to the stream had the greatest percent increase between simulations (Figure 4.5b). Changes were higher than at the watershed-scale because septic HRU's make up a larger portion of the subbasin area (1.62%).

Figure 4.6 shows the percent change in water yield at the watershed and subbasin-scale in each year of the simulation, and Table 4.4 shows the average annual precipitation in each year of the simulation. Analysis showed that the percent change in the water yield between simulations was the greatest in relatively dry years. The percent increase in water yield at the watershed-scale ranged from 5.2% in 2007 to only 1.8% in 2003. The percent increase in water yield at the subbasin-scale ranged from 9.9% in 2007 to only 4.7% in 2009. Results imply that the influence of OWTSs on the water yield within the watershed is greatest during drought conditions.

Figure 4.7 shows the annual percent consumptive water use by OWTSs at the watershed-scale and subbasin-scale for the eight-year simulation. The mean percent consumptive water use for the simulation was 5.6% at the watershed-scale and 5.7% at the subbasin-scale with losses attributed to evapotranspiration and deep aquifer recharge. Water use by OWTSs ranged from approximately 1.5% consumptive in 2004 to 15.6% consumptive in 2007. The higher percent consumptive use in 2007 may be explained by the drought conditions when evapotranpiration losses are the greatest (Table 4.4). Results imply that OWTSs can return approximately 84.4 to 98.5% of the water withdrawn back to ground water and streams for reuse.

Figure 4.8 shows the baseflow fraction of the total water yield versus the percent of impervious surfaces for each subbasin in simulations with and without OWTSs. Both simulations showed a decrease in baseflow with increasing impervious surfaces (R^2 = 0.2143 and 0.2595 for simulations with and without OWTSs, respectively) which is consistent with findings from the study by Calhoun et al. (2003). However, the regression line for the simulation without OWTSs had a steeper slope than the regression line for the simulation with OWTSs indicating that effluent from OWTSs can offset the effects of urbanization on baseflow depending on the the density within the watershed.

CONCLUSIONS

Results show that the SWAT model with the new OWTS algorithm satisfactorily predicted stream discharge in an urbanized watershed containing a large number of OWTSs. At the outlet of the 44 km^2 watershed, OWTSs caused a relatively small increase in total water yield of 3.1%, but at the outlet of a high-density OWTS subbasin, the increase was 5.9%. These results show the importance of considering OWTSs even though they represent a very small percentage of the land use (0.88 and 1.62% of the entire basin and high density subbasin, respectively).

The percent change in water yield between simulations was the greatest in relatively dry years impling that the influence of OWTSs on the water yield within the watershed is greatest during drought conditions. The results support our findings reported in Chapter 3 that there is a small increase in the baseflow in streams of watersheds with high density OWTSs, especially under drought conditions. Contrary to the common assumptions by water planning agencies in Georgia, mean OWTS water use was approximately 5.7% consumptive with the highest consumptive use occurring under drought conditions (15.6 % in 2007).

Despite the advantage of OWTSs in replenishing groundwater and surface water resources, there are water quality concerns in regard to nitrogen and bacteria. This study is part of a larger project examing the contribution of OWTSs in 24 small watersheds surrounding the Big Haynes Creek watershed. The calibrated flow parameters obtained from this model will be used in a future study along with measured N and Cl^- data to calibrate the SWAT model for predicting load in four un-gauged watersheds impacted by high and low density OWTSs. Results from this study may be used by OWTS users as well as watershed planners in order to understand the influence of OWTSs on the water quantity within a watershed.

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Table 4.1. Parameters for calibration in the SWAT model. [R] indicates a relative increase or decrease in the parameter value.

SWAT Parameter	Description	Range	Calibrated Value	P-Value for Sensitivity
CN2	Curve number	35-98	- 0.0921[R]	0.0049
ESCO	Soil evaporation compensation factor	0-1	0.9159	0.0545
SOL_AWC	Available soil water capacity (mm H_2O/mm soil)	0-1	+0.1622[R]	0.3121
SOL_K	Soil saturated hydraulic conductivity (mm/hr)	0-2000	+ 0.7397[R]	< 0.001
SURLAG	Surface runoff lag coefficient	0.05-24	0.9042	0.7930
GW_REVAP	Groundwater re-evaporation coefficient	0.02-0.2	0.0727	0.2627
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur (mm)	0-5000	1.003	0.6845
GW_DELAY	Time for water leaving the bottom of the root zone to reach the shallow aquifer (days)	0-500	215.5	0.0211
ALPHA_BNK	Baseflow alpha factor for bank storage (days)	0-1	0.8623	< 0.001
ALPHA_BF	Baseflow recession constant (days)	0-1	0.0228	N/A
CH_K2	Main channel hydraulic conductivity (mm/hr)	0-500	19.46	< 0.001
CH_N2	Manning's "n" value for the main channel	0.0-0.3	0.0197	< 0.001

Table 4.2. SWAT input parameters associated with water use by on-site wastewater treatment systems (OWTSs).

Input Parameter	**Description**	**Default Value**
ISEP_TYP	The type of OWTS	1 (Generic type conventional system)
ISEP_IYR	Year the OWTS became operational.	0 (Beginning of simulation)
ISEP_OPT	Initial septic HRU operational condition	0 (non-septic)
SEP_CAP	Number of residents in the house	2.5
BZ_AREA	Avg. drainfield area on an OWTS (m^2)	100
ISEP_TFAIL	Time until failing system gets fixed (days)	70
BZ_Z	Depth to the top of the biozone layer from the ground surface (mm)	500
BZ_THK	Thickness of biozone layer (mm)	50
BIO_BD	Density of biomass (kg m^{-3})	1000

Table 4.3.Daily and monthly Nash-Sutcliffe (NS) model efficiency coefficients and R^2 values for three simulation periods

Simulation Period	Daily Flow		Monthly Flow	
	NS	R^2	NS	R^2
2003-2006 (Calibration)	0.49	0.50	0.71	0.74
2007-2010 (Validation)	0.37	0.46	0.68	0.69
2003-2010	0.44	0.47	0.72	0.73

Table 4.4. Annual precipitation in model simulations from January 1, 2003 to December 31, 2010.

Year	Annual Precipitation (mm)
2003	1761
2004	1285
2005	1590
2006	1051
2007	868
2008	1158
2009	1897
2010	1131

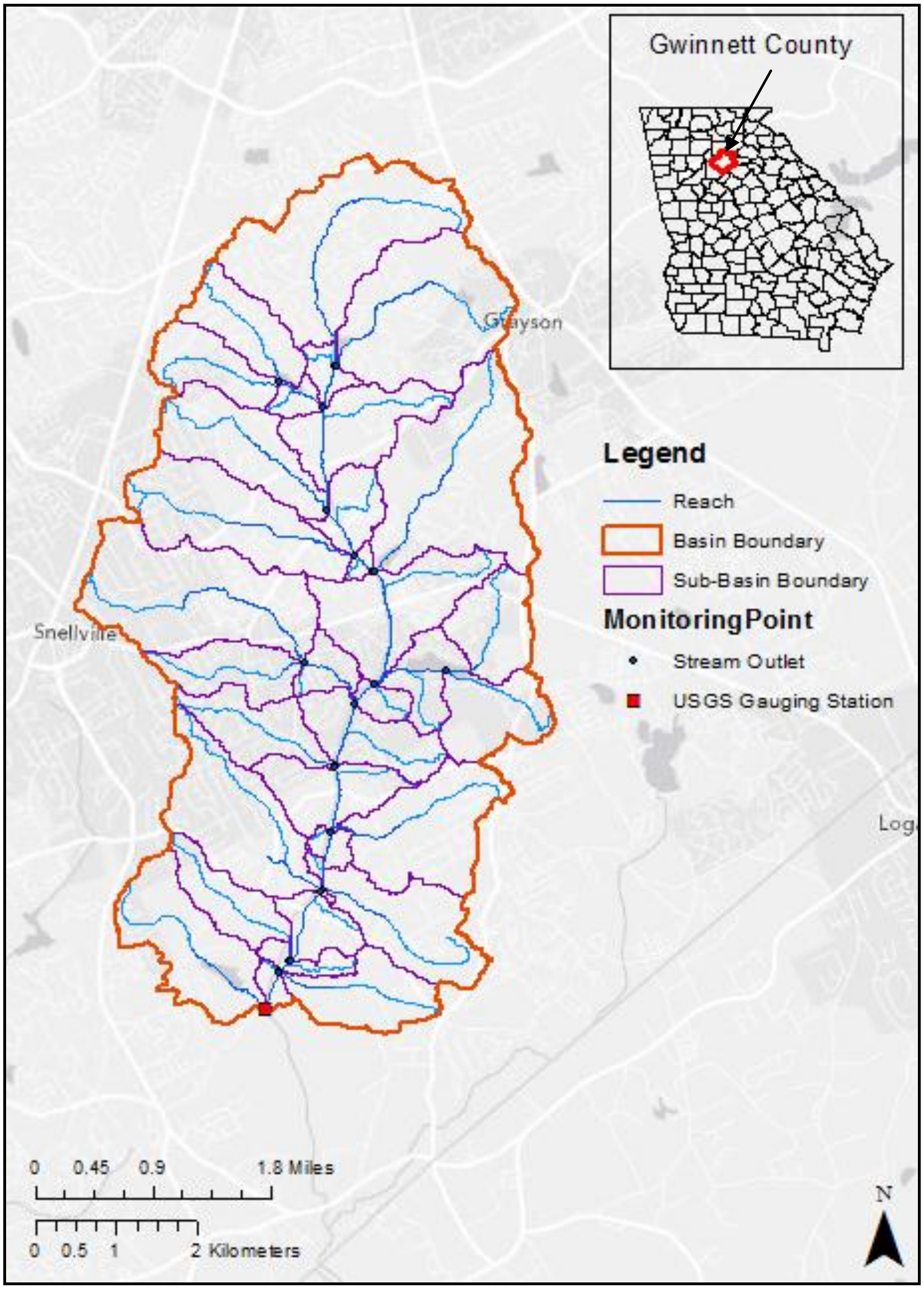


Figure 4.1. Location of the Big Haynes Creek watershed and subbasin boundaries located in Gwinnett County, Georgia

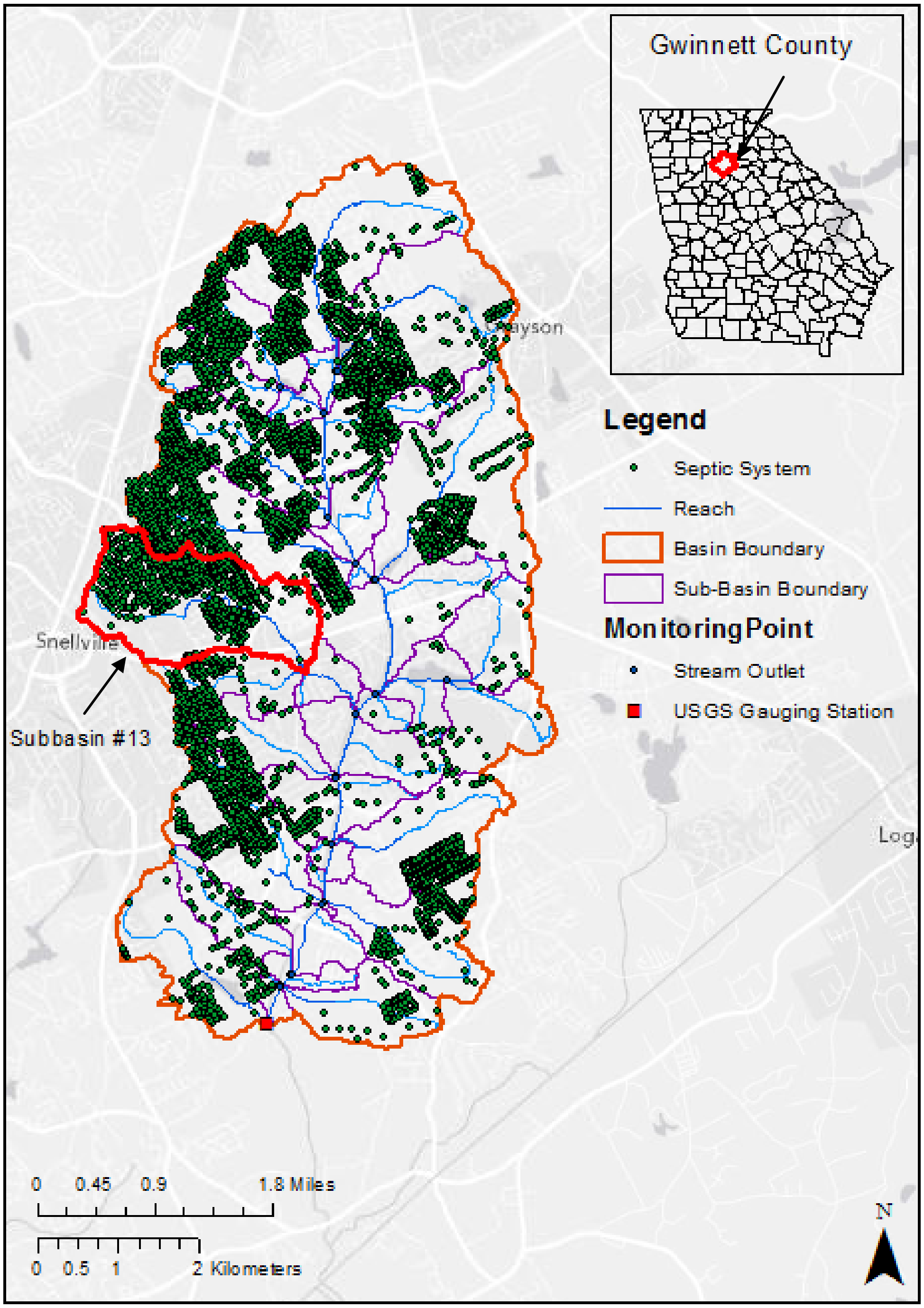


Figure 4.2. Distribution of on-site wastewater treatment systems (OWTSs) within the Big Haynes Creek watershed

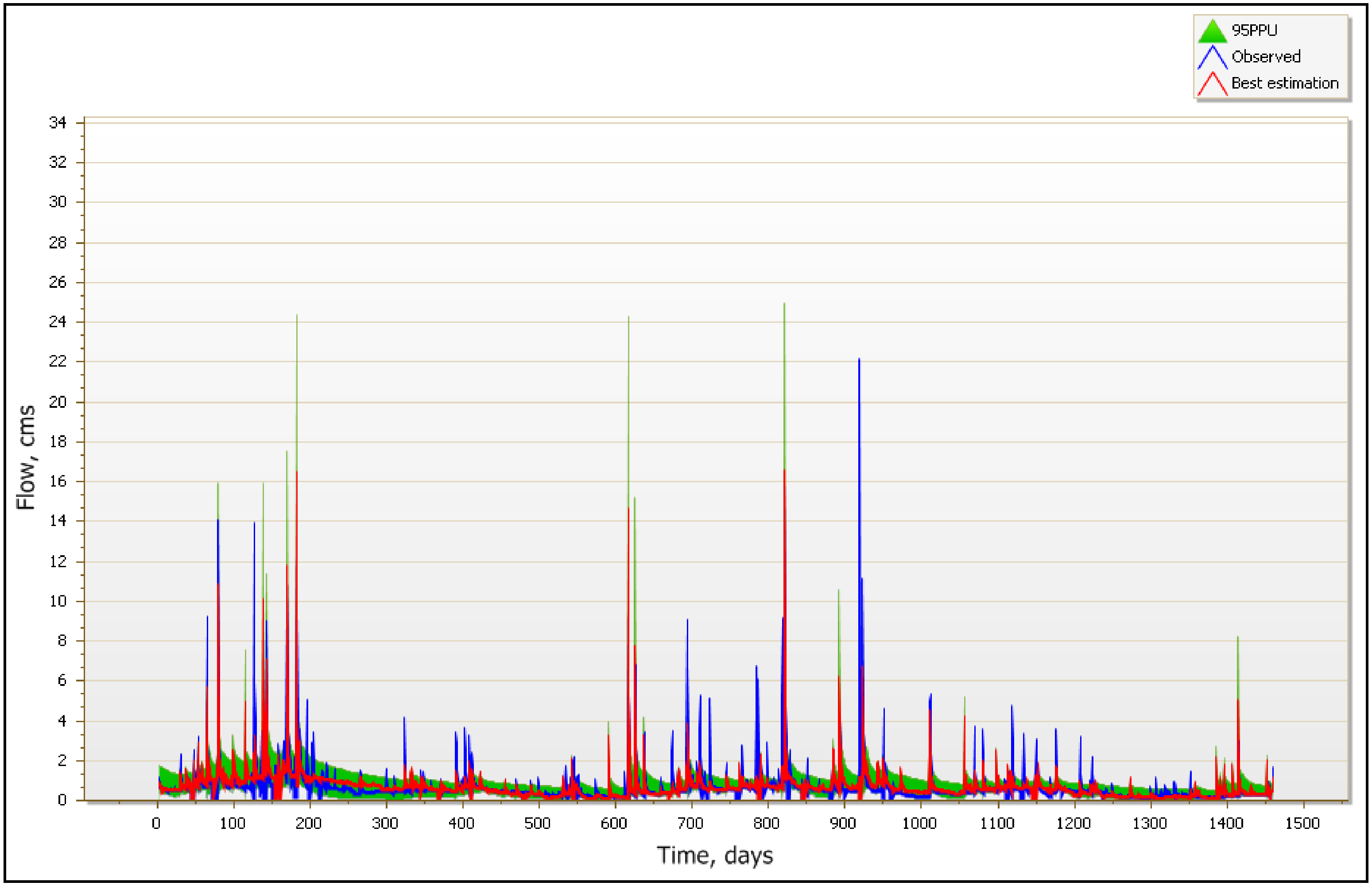


Figure 4.3. Plot of the observed flow, the SWAT model best estimation, and the 95% prediction uncertainty bands for the Big Haynes Creek Watershed from January 1, 2003 to December 31, 2006

a)

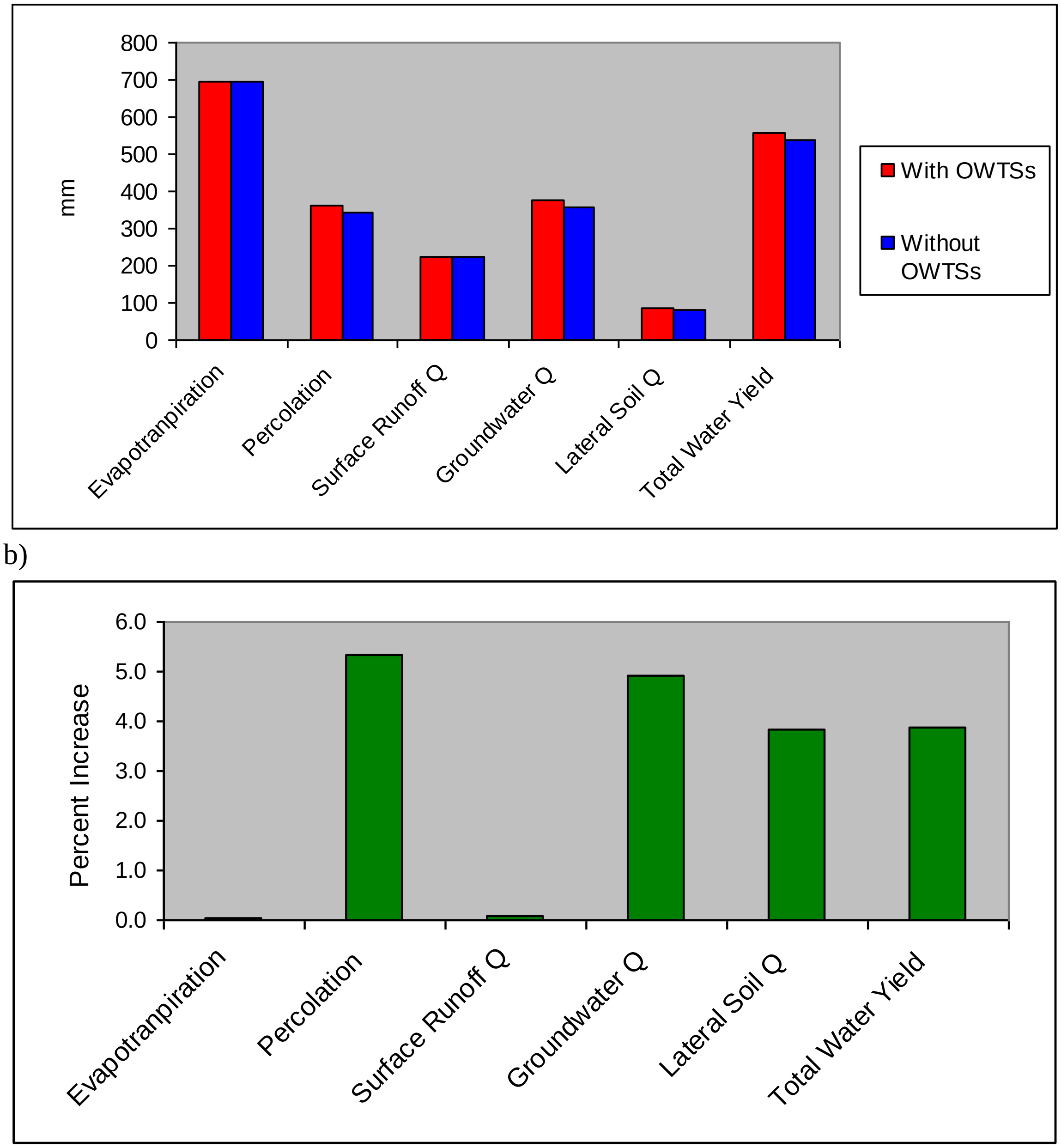
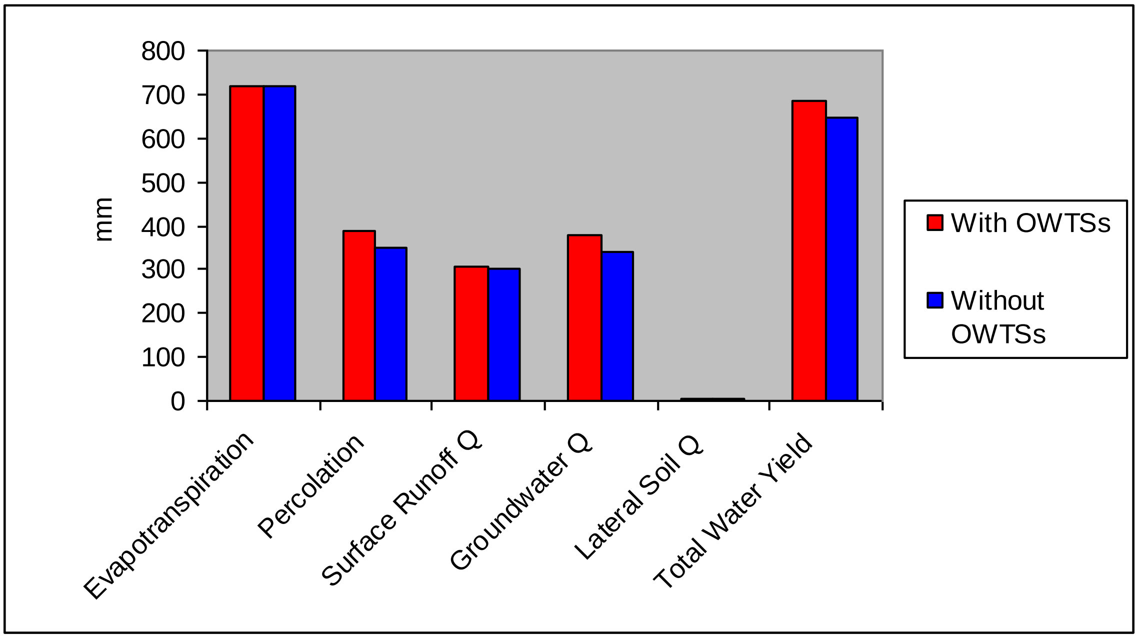


Figure 4.4. Watershed-scale water balance output variables (a) and the percent increase between model simulations (b) with and without the presence of on-site wastewater treatment systems (OWTSs) from January 1, 2003 to December 31, 2010

a)



b)

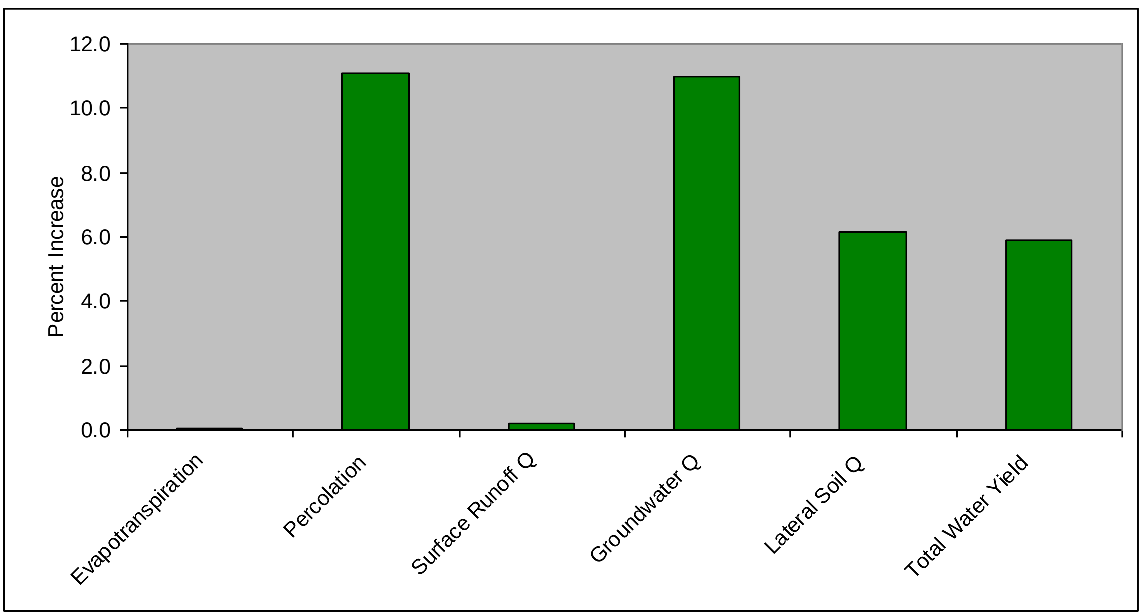


Figure 4.5. Subbasin-scale water balance output variables (a) and the percent increase between model simulations (b) with and without the presence of on-site wastewater treatment systems (OWTSs) from January 1, 2003 to December 31, 2010

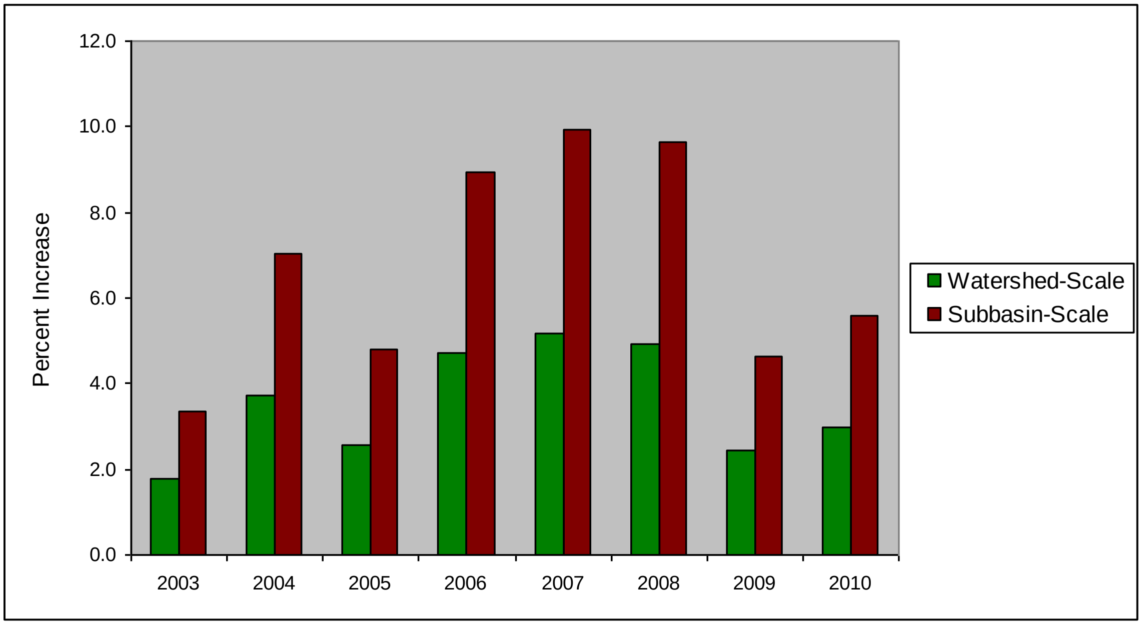


Figure 4.6. Plot of the annual percent incease in the total water yield between model simulations with and without on-site wastewater treatment systems at the watershed and subbasin-scale from January 1, 2003 to December 31, 2010

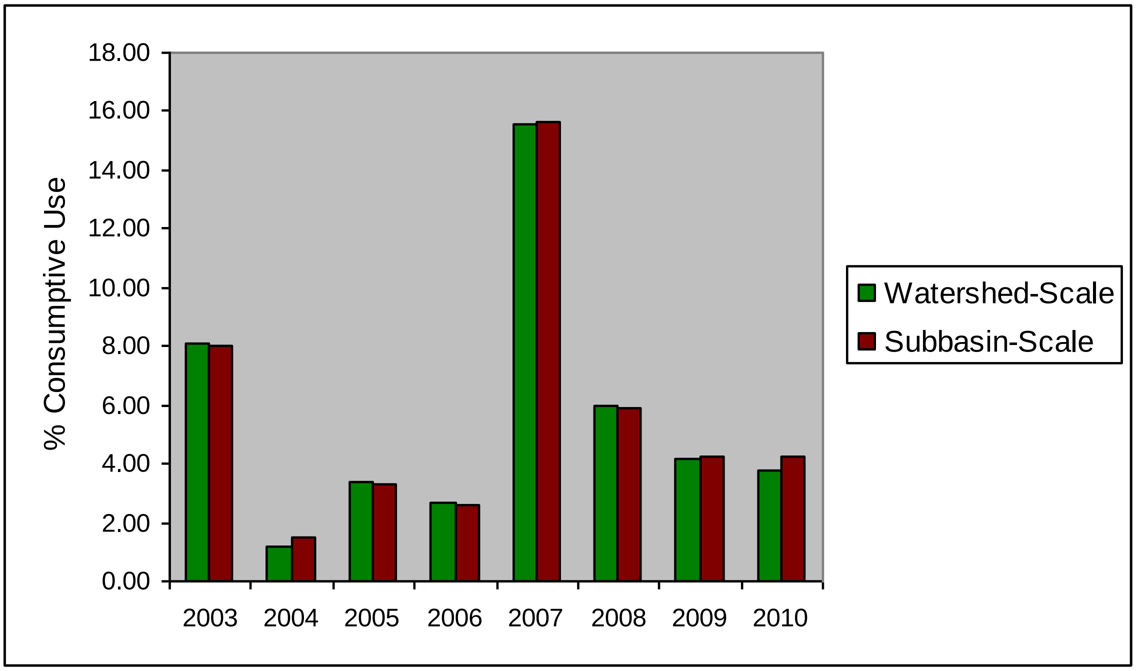


Figure 4.7. The annual percent consumptive water use by on-site wastewater treatment systems at the watershed-scale and the subbasin-scale in model simulations from January 1, 2003 to December 31, 2010.

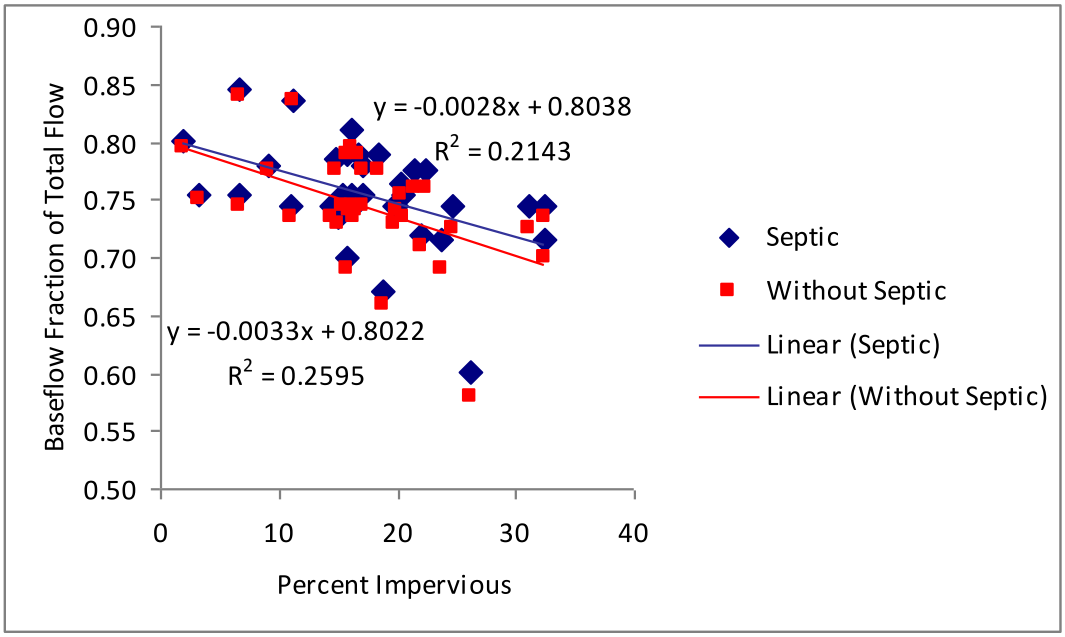


Figure 4.8. The baseflow fraction of the total water yield versus the percent of impervious surfaces for each subbasin in simulations with and without on-site wastewater treatment systems

CHAPTER 5

SUMMARY AND CONCLUSION

The overall objective of this research was to determine the impact of on-site wastewater treatment systems (OWTSs) on the nitrogen (N) load and baseflow in streams of urbanizing watersheds in Metropolitan Atlanta, Georgia. The first goal was to examine the differences in the N load and baseflow as well as other water quality indicators in streams of watersheds impacted by low (LDS) and high density (HDS) OWTSs. Synoptic samples and discharge measurements of streams affected by low and high density OWTSs were taken four times in November 2011, March 2012, July 2012, and November 2012 under baseflow conditions. Mean baseflow measurements in November 2011, March 2012, and November 2012 were not statistically different between watersheds and showed no relationship with OWTS density within the watershed, but July 2012 measurements were significantly higher in the HDS watersheds and increased linearly with increasing OWTS density. EC and Cl^- concentrations increased linearly with increasing OWTS density within the watershed, and NO_3^- concentrations showed a linear increase with OWTS density above a threshold of about 100 OWTSs per square kilometer. The results suggest an increase in baseflow due to the presence of OWTS effluent which may off-set the effects of impervious surfaces and maintain baseflow during drought conditions. Results also indicate a positive correlation between NO_3^- concentration and OWTS density within the watershed above a density of about 100 OWTSs per sq.km.

The second goal of this research was to use measured flow data from a gauged watershed in Metropolitan Atlanta, Georgia to calibrate the SWAT watershed-scale model for predicting stream discharge. The model was calibrated using data from January 1, 2003 to December 31, 2006 and validated from January 1, 2007 to December 31, 2010 using the auto-calibration tool, SWAT-CUP 4. The daily and monthly flow NS coefficients were 0.49 and 0.71, respectively for the calibration period and 0.37 and 0.68, respectively for the validation period indicating a satisfactory fit.

The third goal of this research was to use the calibrated SWAT model to predict stream discharge with and without the presence of OWTSs in order to determine their influence on water quantity within the watershed. Analysis of water balance output variables between simulations with and without the presence of OWTSs showed a 3.1% increase in total water yield at the outlet of the watershed and a 5.9% increase at the outlet of a high-density OWTS subbasin. These results show the importance of considering OWTSs even though they represent a very small percentage of the land use (0.88 and 1.62% of the entire basin and high density subbasin, respectively).

The percent increase in water yield between simulations was the greatest in relatively dry years impling that the influence of OWTSs on the water yield within the watershed is greatest during drought conditions. The results support our findings from the first study that there is a small increase in the baseflow in streams of watersheds with high density OWTSs, especially under drought conditions. According to our simulations, OWTS water use was approximately 5.7% consumptive use, contrary to common assumptions by water planning agencies in Georgia.

As populations in Metropolitan Atlanta, Georgia continue to increase, the use of OWTSs is expected to increase. Properly managed OWTSs offer several advantages over centralized wastewater collection and treatment and are important for the growth of communities. However, because this region heavily relies on surface water withdrawals for the public water supply, it is important to understand the impact of OWTSs on surface water quantity and quality. Overall, this research showed that OWTSs contribute to the discharge in streams of urbanizing watersheds of this region which may off-set the effects of development and maintain baseflow under drought conditions. However the results also showed a positive correlation between NO_3^- concentration and OWTS density within the watershed above a density of about 100 OWTSs per square kilometer. The goal of future research will be to understand and quantify the contribution of OWTSs to the N load through the use of hydrologic models. Results from this research provided data that may be used to inform users as well as watershed planners about the positive and negative impacts of OWTSs on stream water quality and quantity.